Shellfisheries, Seabed Habitats and Interactions in Northumberland

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Abstract

A push for a more evidence based approach to management has resulted in the need for robust evidence of fisheries impacts (or lack thereof), including further research into fine-scale impacts of potting for which little evidence is available. The present work focusses on the inshore pot fishery in Northumberland, UK.

Potting effort distribution maps - a combination of fishing vessel sightings recorded during routine patrols and fishing effort by the Northumberland Inshore Fisheries and Conservation Authority (NIFCA) – showed changes in potting effort across large sections of the study area between years (2004 – 2014). Temporal changes in fishers' habitat selection were investigated using recently collected habitat data. Space-time clustering suggested fishers were actively targeting habitats of interest. Compositional analysis of habitat, showed that fishers preferred rocky habitats over sediment habitats when using both EUNIS level 3 and 6 habitat maps. Information on habitat use and fishing pressure provided the basis for investigation of long-term impacts of parlour potting on epibenthos and habitat within the Berwickshire & North Northumberland Coast European Marine Site (BNNC EMS) through analysis of historical videographic monitoring data between 2002 – 2011. Analysis of biotope change between years (a method recommended for monitoring purposes) showed that at the scales investigated here, change had not occurred. Weaknesses of this analysis for the use in robust ecological research are discussed. A more detailed analysis of community composition and diversity change between years showed that there was little evidence of change. However, changes in species composition and richness of 'Faunal and algal crusts on exposed to moderately wave-exposed circalittoral rock' were observed between years, in most models and between fishing pressures.

Finally, quantification of direct impacts through *in-situ* experimental fishing was investigated using a BACI design. The high experimental fishing intensity, in small experimental areas, coupled with high levels of sampling and replication, provided

robust evidence that current levels of potting are unlikely to have a direct physical impact on epibenthos in Faunal and algal crust, and *Laminaria* spp dominated habitats in Northumberland.

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Acronyms and Symbols

Before, after, control, impact	BACI
Berwickshire and North Northumberland Coast European Marine Site	BNNC EMS
Catch per unit effort	CPUE
Coquet to St Mary’s recommended Marine Conservation Zone	CQSM MCZ
Department for Environment, Food & Rural Affairs	Defra
Distance	*D*
Ecosystem based fisheries management	EBFM
European habitat classification system	EUNIS
European Marine Site	EMS
Faunal and algal crusts on exposed to moderately wave-exposed circalittoral rock	FaAlCr
Geographic information system	GIS
Habitats Regulations Assessment	HRA
Inshore Fisheries and Conservation Authorities	IFCA
Kernel density estimation	KDE
Laminaria hyperborea park and foliose red seaweeds on moderately exposed lower infralittoral rock	Lhyp.Pk
Landings per unit effort	LPUE
Marine conservation zone	MCZ
Marine Management Organisation	MMO
Marine protected area	MPA
Multibeam echosounder	MBES
Multi-dimensional scaling plot	MDS
No take zone	NTZ
Northumberland Inshore Fisheries and Conservation Authorities	NIFCA
Patrol effort	*PE*
Percentage volume contours	PVC
Principal Coordinates Analysis	PCO
Sample number	*n*
Special Areas of Conservation	SAC
Special Protection Area	SPA
Species richness	*S*
The Centre for Environment, Fisheries and Aquaculture Science	Cefas
The International Council for the Exploration of the Sea	ICES
Underwater visual census	UVC
Vessel monitoring schemes	VMS

Chapter 1 . Impacts and Management of Fishing on Habitat and Benthic Communities

1.1. Introduction

Fishing is an important source of food and livelihoods globally, although management of fisheries worldwide has had varying success (Pauly *et al.*, 2002). The traditional model of fisheries management focusses on single species bio-economic modelling, although this method is increasingly perceived as incomplete (Caddy and Cochrane, 2001). More recently, habitat quality and health have been recognised as an integral part to management of sustainable fish stocks as part of an ecosystem based approach to fisheries management (EBFM) (Pikitch *et al.*, 2004; Armstrong and Falk-Petersen, 2008; Howarth *et al.*, 2011; Salomidi *et al.*, 2012). On temperate reefs – some of the most biodiverse and productive habitats in the world - epifauna are key ecosystem components (Hiscock and Tyler-Walters, 2006) providing a source of food and shelter for other species, including commercially important ones (Lambert *et al.*, 2011; Howarth *et al.*, 2015). However, a growing number of scientific publications provide evidence that long-term changes in marine ecosystems may occur due to fishing impacts (Kaiser *et al.*, 1996; MacDonald *et al.*, 1996; Kaiser *et al.*, 2006), although knowledge on fishing impacts is still far from complete, especially for static gear fisheries (MMO, 2012).

Information of fishing gear impacts is needed for effective EBFM management. This is reflected in European and UK policy and legislation that require an evidence base and are changing the way in which marine resources are managed (Woolmer, 2009; MMO, 2012). In the UK, the Department for Environment, Food & Rural Affairs (Defra) recently revised fisheries management in European Marine Sites (EMSs)(MMO, 2012) stating that fishing activities within these protected areas would only be allowed if they do not adversely affect site integrity or undermine the achievement of conservation objectives (MMO, 2012). Consequently, each current fishing activity within EMSs must undergo an Habitat Regulations Assessment (HRA) in accordance with Article 6 of the Habitats Directive (Council Directive 92/43/EEC), to assess potential impacts on the designated features of each site. In addition, with the designation of 27 Marine Conservations Zones (MCZs) across the UK in 2016 and further sites recommended for designation, activities impacting features for which these were designated must be better understood in order to allow effective management.

This review discusses some of the methods and information required for effective implementation of EBFM. Impacts of commonly used fishing gears in the UK on benthic communities and habitats will be discussed, with a particular focus on the little studied environmental impacts of potting. Scientific publications on direct physical impacts of potting and indirect impacts such as ecosystem changes will be critically appraised. Assessment methodologies employed to detect marine benthic change as part of current monitoring practices will also be reviewed with critical appraisal focused on hierarchical classification systems and videography methods. Habitat and fishing effort mapping and its uses in management of the marine environment will be discussed. Finally, current legislation protecting habitats of interest in the UK, the study area and the aims of this research are described.

1.2. Sensitivity of organisms

Fishing impacts have been extensively studied over the past 20 years due to growing concern over the ecological impacts that can arise from the removal or damage of benthos and habitat (Shester and Micheli, 2011). Sensitivity of organisms and habitats play a crucial role in determining the impacts of fishing.

Biological and ecological traits of species, as well as the nature of the fishing gear, will determine sensitivity to fishing impacts (Roberts *et al.*, 2010; Shester and Micheli, 2011). Benthic epifauna and flora, and sedentary infauna are not able to avoid fishing gears, weights, anchors or ropes and the degree of damage will depend on the robustness of the organism. For example, flexible organisms may not be damaged by the passage of a trawl whereas brittle and inflexible species such as echinoderms may be crushed (MacDonald *et al.*, 1996). Larger species are considered more vulnerable than small species to towed gear, as smaller organisms may pass under ropes or be pushed out of the way by pressure waves created at the front of towed gears (Bergman and van Santbrink, 2000). Body size is also linked to life history, as larger organisms are often slower growing, longer living, have a lower reproductive output and lower natural mortalities (Roberts *et al.*, 2010). This means that larger species' populations are predicted to be more affected by high fishing mortalities.

However, the nature of epibenthic communities is predominately determined by their surrounding abiotic conditions (Connor *et al.*, 2004) and as such fishing impacts will also vary depending on habitat type (Hiscock and Tyler-Walters, 2006). Generally, areas that naturally have high levels of disturbance from wave action, sedimentation and currents will be composed of small and robust organisms that will be resistant and resilient to disturbances (Kaiser *et al.*, 2006). In contrast, assemblages in stable, physically undisturbed habitats, will tend to be structurally more complex, with larger, long-lived species that are intolerant to disturbances (Kaiser *et al.*, 2006).

1.3. General fishing gear impacts

Fishing activities will not always have detectable effects. Impacts to the benthos from fishing can range from no or minimal physical damage to major redistribution of sediment and high mortality of benthic species (MacDonald *et al.*, 1996). The impacts of mobile gear on benthic habitats have been well studied and have been found to reduce complexity, species diversity, and productivity (Watling and Norse, 1998; Johnson, 2002) (Table 1.1). In contrast, the physical impacts of fishing gear on benthic habitats by static gears have been less studied and results from experimental work differ (Shester and Micheli, 2011; Coleman *et al.*, 2013). However, the relatively simple technology used, the limited area disturbed and the capability for effective local governance (Jacquet and Pauly, 2008) are expected to lead to low ecological impacts for static gear (Shester and Micheli, 2011) (Table 1.1).

The lack of research into benthic impacts of static gears may be explained in several ways. Firstly, in industrialised parts of the world, large-scale commercial fishing vessels often use mobile gears which cover large areas, whereas static gears impact smaller areas and are often used in artisanal or small-scale fisheries (Shester and Micheli, 2011) (Table 1.1). Thus, mobile gears are viewed as more destructive than static gears and are therefore seen as a priority for research (Coleman *et al*., 2013). Secondly, mobile gears are often deployed on soft sediment habitats that can be easily sampled for scientific investigation with grabs and corers, whereas static gears are often deployed on hard surfaces where scientific sampling techniques can be more costly, time consuming and destructive depending on the sampling technique used (Davies *et al.*, 2001) (see section 4 for further detail).

Table 1.1. Summary of impacts of different fishing gears on habitat and benthic communities.

GEAR TYPE	DESCRIPTION OF GEAR AND FISHING TECHNIQUES	SCALE	HABITAT TYPE	MAIN IMPACTS	• REFERENCES
• **Mobile gear**					
Bottom trawls	Vessels drag a weighted net over the seafloor aiming to catch bottom living or feeding species. E.g. shrimp, cod, flat fish etc.	> Km	Predominately soft and mixed ground (although can be used on flat, homogenous hard ground using a “rock hopper” (Roberts *et al.*, 2010))	• High levels of bycatch • Physical impact on seafloor: reduction of complexity, diversity, and productivity. • Physical impact on the seafloor depends on exposure of habitat to abiotic conditions (wave action, currents, etc) • Impacts over considerable distances and areas	• (Dulvy *et al.*, 2003) • (Watling and Norse, 1998) • (Watling and Norse, 1998) • (MacDonald *et al.*, 1996)
Dredges	Vessels drag a metal frame shaped like a scoop across the seabed. Target species include oysters and scallops. Dredges may have “teeth” under the bottom lip of the metal frame in order to uproot fauna within the substrate.	> Km	All ground types. Depth of “teeth” penetration varies with ground type (approximately 10 cm for soft ground, 2-3 cm for hard ground) (Kaiser *et al.*, 1996)	• Same impacts as bottom trawl. These are rated as having a comparable level of disturbance.	• (Kaiser *et al.*, 1996)
• **Static gear**					
Gill nets	Vessels deploy vertical panels of netting. There are several types of gill net that can either fish on the seafloor, at the surface or are left to drift. Target species are mainly pelagic finfish e.g. Salmon, Cod, Grouper	< Km	All habitats (when in contact with seafloor) although predominately on hard ground. In order to avoid conflict with other fishers, placement of static gear may be dictated by the use of mobile gears in the area (Fitzsimmons *et al.*, 2011)	• Some bycatch (although there are mixed conclusions on the level and importance of bycatch using gillnets) • Conclusions on physical impacts are mixed although cumulative impacts from repeat fishing may be significant	• (D'Agrosa *et al.*, 2000; Kelleher, 2005) • (Stephan *et al.*, 2000; Shester and Micheli, 2011)
Potting	Pots use bait to attract target species into a trap or creel. The trap design ensures that once the target species is inside the trap it is difficult to escape. Pots, traps or creels are deployed onto the seafloor in “fleets or strings” and held in place by weights or anchors at each end. Target species include finfish, lobster, prawns, whelks, cuttlefish and crabs.	< 300m	All habitats although predominately on hard ground. However, in order to avoid conflict with other fishers, placement of static gear may be dictated by the use of mobile gears in the area	• Very low bycatch • Physical impacts are expected to be low but little information is available.	• (Roberts *et al.*, 2010) • (Eno *et al.*, 2001; Coleman *et al.*, 2013)

1.4. Impacts of potting on habitat and benthic communities

Physical impacts of potting have received relatively little attention (Coleman *et al*., 2013) even though they are regularly mentioned in official guidance on habitat management for nature conservation as being a threat to fragile epibenthic species (Jones *et al.*, 2000; Defra, 2013; Eno *et al.*, 2013). Potting may directly impact habitat and benthic communities through physical interaction when fishing (Eno *et al.,* 2001) and through indirect impacts such as changes of food web structure (Siddon and Witman, 2004).

1.4.1. Nature of potting

In order to understand the physical impacts that potting may have on the seafloor, a more detailed description of the fishing process is needed. Potting methods vary between locations in the UK in terms of the materials used for the pot, the number of pots used per fleet, pot size and weight, size and weight of the anchor-weight and distance between pots (Armstrong N, 2012, pers. comm.). However, the fleet configuration and deployment method described below is considered standard fishing practice in the UK (Lovewell *et al.*, 1988; Bullimore *et al.*, 2001; Coleman *et al.*, 2013). Generally, 10-30 baited pots are attached to a ‘mainline’ by 2-3 m lengths of rope forming a ‘string’ or ‘fleet’ of gear (Fig 1.1). Traditionally, pots are evenly spaced along the mainline every 10 fathoms (~18 m) and anchor-weights are attached at each end in order to prevent dragging from wave action or strong currents. Marker buoys, which are used to facilitate retrieval, are attached to each end of the fleet with a rope length which is usually twice as long as the water depth of the fished site. Fleet deployment is initiated by dropping the first buoy-line and anchor-weight into the water (Fig 1.1, a). The weight of the anchor and the movement of the vessel in the opposite direction along the chosen fishing ground, allows the pots, followed by the second anchor-weight and buoy-line, to be pulled overboard (Fig 1.1, a). Once pots have been deployed they are left to ‘soak’ (fish) for 1–3 days (Fig 1.1, b). They are then ‘hauled’ (retrieved) which is the reverse process.

Due to water depth, the distance between pots will not be maintained (Fig 1.1, a). Anchors and buoys are very secure and are designed to remain static, but the slack in the mainline (Fig 1.1, b) allows pots the freedom to move (see section 4.2.2).

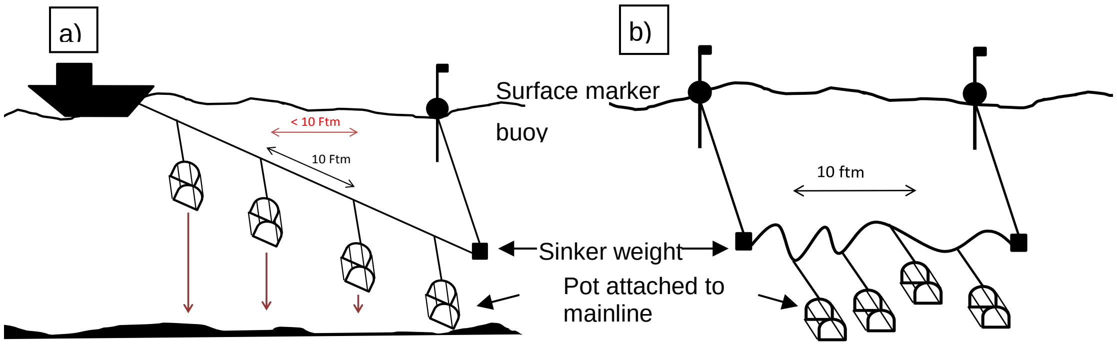


Fig 1.1. Lateral view diagram of parlour pot fleet deployment (a) and parlour pot fleet soaking (during fishing) (b). Note: distances used in this diagram are for illustration purposes only and do not necessarily represent real or accurate distances of pots underwater.

1.4.2. Direct impacts

Direct, physical impacts from the potting gear on the benthic environment can occur in three ways (Coleman *et al.*, 2013):

1) During deployment, potting gear (including parlour pots, end weights and ropes) may land on the seafloor and in the process crush or damage epibenthic organisms (Eno *et al.*, 2001).

2) During the fleet soak time, the gear may scrape, abrade and damage the benthos due to pot movement (Lewis *et al.*, 2009). Pot movement is most likely to occur due to strong tides and, or in combination with, high wave action during storms (Jones *et al.*, 2000) (Fig 1.1).

3) During retrieval, potting gear may drag along the seafloor and damage epibenthos as it is being lifted off the seafloor. Snagging of lines, weights and pots on rocks may further damage the benthos as stronger forces may be needed to free the gear (Eno *et al.*, 2001). Furthermore, if the gear is dragged laterally there will be a greater impact on the benthos, although this generally only occurs when high wind, strong tides or navigational hazards prevent a direct vertical lift (Eno *et al.*, 2001). However, fishers will try and avoid snagging of gear as this greatly increases the risk of wear and tear.

1.4.2.1. Impacts of pot deployment

Only two studies have examined impacts of pots landing on seafloor (Eno *et al.*, 2001; Shester and Michelli, 2011). These studies were undertaken in areas where potting is common. Evidence of physical impacts from pots landing on presumably fragile species (the sea pens *Penatula phosphorea*, *Virgularia mirabilis* and *Funiculina quadrangularis*) was investigated by Eno *et al.* (2001) in Badentarbet Bay on the West coast of Scotland. Results showed that the bow wave of the sinking pot was sufficient to bend sea pens away from the pots just before impact and that once the gear had been removed the sea pens righted themselves after 72-144 hours, without any noticeable damage.

Shester and Michelli (2011) examined the physical impacts of pots when landing on gorgonian corals (*Muricea californica, Eugorgia daniana, Eugorgia ampla, Leptogorgia diffusa*, and *Pacifigorgia* sp.) and the Southern Sea palm kelp (*Eisenia arborea*) on rocky reefs in Baja (California Sur, Mexico). *Eisenia* was able to withstand the force of dropped pots with no damage; therefore gorgonian corals were focused on as species representing the 'worst case scenario'. However, only one out of the 37 trials observed any damage (less than 1%) to colonies of *E. ampla*.

These two studies suggest that deployment of pots on to the seafloor have no, or negligible, impacts on epibenthos. However, both these studies only examine a small subset of local species perceived to be fragile and their conclusions may not be applicable to different epibenthic communities, habitats or locations.

1.4.2.2. Impacts of pot soak time

The impacts of pot movement on the seafloor during fleet soak time have been investigated briefly by Eno *et al.* (2001) and in more depth by Lewis *et al*. (2009).

Qualitative data from Eno *et al.* (2001) suggest that pots were normally static on the seabed. On occasion, pots dragged along the seafloor when wind and tidal streams were strong. In addition, strong tides and large swell were observed to cause the lead pot to bounce up and down on the seabed when insufficient line was deployed. Although there was pot movement during the soak time, no impacts were detected on epifaunal species at any of the sites. However, this study did not quantify pot movement and only provided observational data from SCUBA divers (Eno *et al.* 2001).

Lewis *et al.* (2009) assessed impacts on coral communities due to pot movement on the seafloor in the Florida Keys (United States). This three-year study showed that winter storms (defined as having winds over 27.8 $km.h^{-1}$ for more than 2 days) moved pots a mean (± SE) distance of 3.63 ± 0.62m, 3.21 ± 0.36m, and 0.73 ± 0.15m at 4m, 8m and 12m depths respectively. In addition, pots impacted a mean area of 4.66 ± $0.76m^2$, 2.88 ± $0.29m^2$, and 1.06 ± $0.17m^2$, of seabed (4m, 8m and 12m depths respectively). All sessile fauna were identified in this study. Results showed that stony coral, octocoral, and sponges were damaged or removed by pot movement.

Overall, sessile fauna cover within the impacted areas was reduced by 14%, 10% and 6% at 4m, 8m, and 12m depths respectively.

Lewis *et al*. (2009) conclusively showed that pot movement occurs due to strong winds and big swell, and that pots can damage sessile fauna during soak times. Depth plays a key role in the magnitude of potting impacts. At deeper sites, which are more sheltered from the effects of wind and waves, pots move less and therefore have a lower impact on sessile fauna than shallower sites. However, conclusions from this work are of limited applicability to shellfisheries in the UK since it focused on coral reefs, in shallow water (4-12m), in tropical storms and using single buoyed pots without anchor-weights. In contrast, potting in the UK is undertaken in larger fleets (10-30 pots), that are held in place using anchor-weights and are often fished at greater depths (often deeper than 20m), although potting also occurs in shallower water (primarily in good weather during the summer months) and is dependent on target species and other fishing gear use in the area (Turner, 2010)).

1.4.2.3. Impacts of hauling pots

The impacts of pots dragging across the seafloor during hauling have been studied by Eno *et al*. (2001) and Shester and Michelli (2011). Both studies replicated the speed and angle of pots being dragged by SCUBA divers. Gorgonian corals showed damage (< 5% of the skeleton was impacted) in 40% of trials but the Southern Sea palm kelp was not impacted. Neither gorgonian corals nor Southern Sea palm kelp were detached from the seafloor in any trials (Shester and Michelli, 2011). Eno *et al*. (2001) also observed very few immediate impacts on epifaunal species from rocky substrates in southern England. In Scotland, observations of supposedly sensitive species such as sea pens and sea fans showed no signs of impact. In addition, sea pens and sea fans which were experimentally detached re-established themselves when in contact with muddy substrate.

1.4.2.4. Evidence of direct impacts from experimental fishing

The impact of potting on sessile epifauna in rocky habitats has also been studied in its entirety rather than its individual components (deployment, soak and hauling) via experimental fishing studies. Eno *et al*. (2001) did not detect any impacts using a BACI (before, after, control, impact) method for experimental potting. However, there were problems with this study that may have affected the capacity to detect potting impacts. Firstly, this was a short-term study with only ten fleet deployments and retrievals over a one month period. Secondly, small sample numbers led to issues of power in the analysis. Finally, erroneous results may have confounded the analysis (Coleman *et al*., 2013) since certain species appeared to significantly increase in abundance in response to the potting treatment although it was concluded that this was not possible due to the life-histories of the species concerned (i.e. species that were not detected in the initial survey were detected after the one-month experiment. However, these species could not have grown within this time period and their presence was attributed to sedimentation prior to the experiment.

This study was not able to recommend maximum potting levels that sessile epifauna are able to withstand at a site. Indeed, the cumulative impacts from repeat fishing are not well understood and further site-specific studies are required to determine optimum fishing levels that satisfy both fishery and conservation interests (Eno *et al.*, 2001).

An in-depth study by Coleman *et al*. (2013) examined direct potting impacts and recovery of sessile epifauna in locally fished areas and a protected no-take zone

(NTZ) over four years in south west England. Sessile epifaunal assemblages in circalittoral reef habitats were expected to change more over time in non-fished areas as assemblages are only affected by natural variation compared to fished areas which are affected by fishing pressures and natural variation. Comparison of assemblages in sites both in fished areas and the NTZ showed there had been no change in sessile epifauna composition over four years between these sites. In addition, there was no evidence that temporal variation in the abundance of individual taxa differed between fished and non-fished areas. However, it could be argued that over a long period of time, pot fishing had led to a changed, but stable ecosystem state (Eno *et al.*, 2001). This ecosystem state would no longer be impacted by pot fishing and may not show any signs of recovery (Hughes *et al.*, 2005). In addition, control sites for this study differed in depths (5 – 10 m), exposures (controls were more wave exposed than experimental areas) and substrate differences (boulder reefs compared to bedrock) compared to experimental sites (Coleman *et al.,* 2012), which may have resulted in small scale impacts being overlooked.

Experimental potting in fished and non-fished locations also showed that epifaunal assemblages were not significantly impacted during the four year study. However, recovery of habitats exposed to sustained potting impacts could take longer than four years (Coleman *et al*., 2013). Indirect impacts of potting on the ecosystem as a whole were out of the scope of these studies, however, a meta-analysis of indirect fishing effects, showed that it took a mean of 13.1 ± 2.0 years for impacts to be detected on non-targeted species (Babcock *et al.*, 2010).

1.4.3. Indirect potting impacts

There is increasing evidence that fisheries are likely to lead to long-term changes in marine ecosystems (MacDonald *et al.*, 1996; Kaiser *et al.*, 2006; Sewell *et al.*, 2007). Depletion of top predators, such as lobsters, from a local area may have a destabilizing effect on the ecosystem through changes in food web dynamics (Eno *et al.*, 2001). The indirect impacts of fishing on taxa that occur through cascading trophic interactions have been relatively little studied due to the long time-scale needed for these studies (Babcock *et al.*, 2010). However, indirect potting impacts, from extraction of crab and lobster, can change benthic community structure (Siddon and Witman, 2004) highlighting the importance of understanding these trophic interactions.

Indirect impacts of fishing have been mostly studied in the context of NTZs, as protected areas offer good comparisons between impacted (fished) and recovering areas (NTZ). Extensive research into the effectiveness of NTZs suggests that highly-protected marine areas can trigger rapid increases in the abundance, diversity, and productivity of marine organisms (Halpern and Warner, 2002; Lubchenco *et al.*, 2003). In addition, ‘spill over’ into adjacent areas to the NTZ can occur via adult migration and larval dispersal, causing increased catches in these areas (Roberts and Polunin, 1993).

Indirect changes in marine reserves only occur if there are direct effects of fishing. Therefore if fishing is impacting species abundance and diversity in an area, protection from fishing produces an absolute increase in abundance, mean size of individuals, or biomass of targeted species. This is not simply a relative change compared to fished areas but rather a restoration to a former level (Babcock *et al*., 2010). There is strong evidence that lobster populations undergo rapid, large increase in the abundance and sizes within NTZs and that spill over of sublegal sized lobster occurs in areas adjacent to protected areas (Pande *et al.*, 2008; Barrett *et al.*, 2009; Hoskin *et al.*, 2011). In addition, crab and lobster species interact strongly with other non-target species. Thus, extraction of any crab or lobster species through commercial fishing is likely to change benthic community structure (Siddon *et al*., 2004).

Increases in decapod populations, due to protection offered by NTZs in temperate reef ecosystems in New Zealand, resulted in a decline of sea urchin populations, and of grazing, and the recovery of kelp forests (Babcock *et al.*, 1999). Similar changes of natural habitats have also occurred in the Gulf of Maine where harvesting of urchins and lobsters increased abundances of macroalgae through changes in prey and predator behaviour such as prey switching or multiple predator effects (Siddon *et al*., 2004).

Indirect impacts of potting may play a significant role in abundance and diversity of non-target sessile epifauna. However, the long time scale required to detect these impacts means that few studies have been conducted. In addition, no studies have investigated indirect impacts as they occur but rather have looked at recovery of non-target species and inferred from these situations how fishing may indirectly impact the environment.

In summary, fishing impacts vary in function of the sensitivity of organisms and fishing type. Mobile gears, which fish on a much larger scale than static gears, have been well studied but little information exists on impacts of static fishing gear. Potting impacts (direct and indirect) may affect marine assemblages but further work is required. The methods used to assess changes in benthic communities, including from potential potting impacts, will influence whether impacts are detected.

1.5. Hierarchical classification systems

Potting impacts on benthic communities are sparsely documented, although the few studies have focused on impacts on a small group of locally important species perceived as fragile (Eno *et al*., 2001; Coleman *et al*., 2013). Results from small-scale experimental fishing impact studies are difficult to extrapolate to an ecosystem-wide scale which are ideally required for EBFM (Hiddink *et al.*, 2006; Hinz *et al.*, 2009). No studies have looked at the direct impact of potting on communities and habitats as a whole. Ecological monitoring is often undertaken at this broader EBFM scale, resulting in a disconnect between conclusions from peer-reviewed literature with those from monitoring and surveillance reports. This section will introduce and critically appraise the most widely used habitat monitoring method: hierarchical classification systems in order to explore the usefulness of these monitoring methods for fishing impact studies.

Most hierarchical classification systems use the term “biotope” to describe a combination of the physical environment (habitat) and its distinctive assemblage of species (community assemblage) (Olenin and Ducrotoy, 2006). Habitat is defined according to geographical location, physiographic features and the physical and chemical environment (including salinity, wave exposure, strength of tidal streams, etc.) (Connor *et al.*, 2004). Community is described as ‘‘a group of organisms occurring in a particular environment, presumably interacting with each other and with the environment, and identifiable by means of ecological survey from other groups” (Hiscock and Tyler-Walters, 2006).

The term biotope is commonly found in recent national and international environmental documents due to the increasing reliability of classification systems (Connor, 1995; European Environment Agency, 2005). The EU CORINE biotope classification was developed in the 1980s but it was very broad and alternatives have since been proposed in order to address this classification's shortcomings. These include national classification systems such as the marine biotope classification in the UK (first published by the JNCC in 1997 and revised in 2004) (Connor *et al*., 2004), the Zones Nationales d'Intérêt Scientifique, Faunistique et Floristique (ZNIEFF) classification in France, (Dauvin *et al.*, 1996) and a regional classification of coastal biotopes and their complexes for the Baltic Sea (Nordheim and Boedeker, 1998). These classification systems are compatible with, and contribute to, a European habitat classification system (EUNIS) (Olenin and Ducrotoy, 2006; Roberts *et al.*, 2010).

1.5.1. The marine biotope classification of the United Kingdom

The UK marine biotope classification was developed through the analysis of empirical data sets, the review of other classifications and scientific literature, and in collaboration with a wide range of marine scientists and conservation managers (Connor *et al*., 2004).

Classification systems may describe the marine environment at different spatial scales. The UK marine biotope classification is split into six levels (that correspond directly with their EUNIS counterparts) ranging from very broad scales (environment, Table 1.2) to very small scales (sub-biotopes, Table 1.2). These different scales mean that classification of benthic communities is meaningful both for detailed scientific application and to the much broader requirements needed for management of the marine environment (Connor *et al*., 2004).

Table 1.2. Structure of the UK marine biotope classification (modified from Connor et al., 2004).

Classification level	Name	Description
1	Environment	Marine, in order to separate the marine environment from the freshwater and the terrestrial.
2	Broad Habitats	Large scale habitats such as reefs, mudflats etc.
3	Main Habitats	Habitats of large scale biological definition equivalent to areas of intertidal sites designated as Sites of Special Scientific Interest (SSSI).
4	Biotope Complexes	Groups of biotopes with similar physical and biological characteristics. These biotopes usually occur together and provide better mapping and management tools due to their broader scale and ease of identification through courser survey methods.
5	Biotopes	Areas of a minimum of $25m^2$ distinguished by their significant dominant species or collection of conspicuous species.
6	Sub-Biotopes	These are biotopes defined by less conspicuous species and geographical variation or a disturbed/ polluted variation of the natural biotope.

The aim of the classification is to provide a tool to aid the management and conservation of marine habitats (JNCC, 2013). Information on marine habitats and their associated benthic communities is needed in order to undertake the ecosystem based approach to management of the marine environment widely advocated at national and international levels (Defra, 2002).

The UK marine biotope classification is mainly used for surveillance and monitoring purposes (Defra, 2005a) as common standards ensure a consistent, integrated, UK-wide approach (JNCC, 2013). This standardised format allows for comparison between locations and studies. For example, journal papers often simply refer to fishing impacts on ‘sandy’ habitats. Within the UK biotope classification scheme, this could refer to any of the 17 biotopes occurring on sandy sediments. These habitats each have distinctive biological assemblages and environmental conditions making comparisons between studies error-prone (Roberts *et al*., 2010).

The appropriate scale used for ecological work will depend on the aim of the work being undertaken and the sampling methods used. For example, broad-scale maps for management of SACs on a national scale may only need “main habitats” (Table 1.2, Level 3), whereas more detailed maps of a smaller area, such as a single SAC, may need to use “sub-biotopes” (Table 2, Level 6). Higher levels of the classification system require a higher level of biological information which can sometimes be problematic. This is especially true in EMSs as these areas can only be sampled with non-destructive techniques in order to preserve the quality of their marine habitats and species which are deemed of international importance (Davies *et al.*, 2001). Davies *et al*. (2001) recommend using drop-down video, remotely operated underwater vehicle (ROV), diver-operated video or towed video to monitor biotic composition of subtidal reefs (Marine Annex I features, Habitats Directive). However, these visual monitoring methods are inherently problematic in temperate inshore waters due to seasonally poor visibility and may not be suitably sensitive (i.e. accurate and precise) for the identification of higher classification levels (i.e. sub-biotopes) (Saunders *et al.*, 2011) (see section 1.5.3 for further detail).

The UK marine biotope classification system (Table 1.2) remains the most prevalent for monitoring and surveillance due to the easy use of biotopes to describe and map the extent and geographical distribution of habitats and biological communities. It also allows for changes in habitat distribution and extent over time. Furthermore, the relative importance of habitats (or rarity) can be examined on national and regional levels, specifically for protected areas (i.e. SACs, SPAs, etc.), which allows prioritisation for conservation efforts (Sanderson, 1996; Connor *et al.*, 2004).

1.5.2. Habitat mapping

One of the primary uses for the biotope classification system (in addition to providing a consistent description of habitat types), is mapping geographical distribution of biotopes to aid spatial management of the marine ecosystem (Connor et al., 2004). This is achieved by combining biological information (biotope data) with recently developed acoustic survey techniques such as acoustic ground discrimination systems (AGDS), sidescan sonar systems (SSS) and multi-beam echo sounders (MBES) (Lucieer, 2008), to produce thematic seafloor maps that can be used for management purposes (MESH, 2008). Salomidi *et al*. (2012) recommends mapping biologically and ecologically important areas (using acoustic and in-situ techniques) with their associated human uses and political and legal arrangements. These holistic maps are seen as an essential first step towards effective marine spatial management (Crowder and Norse, 2008). However, few areas have been mapped in Europe due to the high cost and considerable technical knowledge needed for adequate acoustic mapping (but see MESH (2008) for European broadscale habitat mapping) (Galparsoro *et al.*, 2012). Spatial representations of biotopes are particularly useful as they may show changes otherwise not detected using biotope frequency alone, for example by showing changes in extent over time. This is particularly important with regards to investigating changes induced by spatially and temporally variable anthropogenic impacts such as fishing (Diesing *et al.*, 2009).

1.5.3. Critical appraisal of marine biotope classification systems

Although hierarchical classifications are a useful tool for comparable and standardised environmental monitoring, there are limitations. In temperate waters, only some habitats have conspicuous dominant species (e.g. kelp forests, mussel beds, maerl beds). Many habitats support a mosaic of species, none of which is particularly dominant (Connor, 1995). This patchiness can significantly vary over time with little information available on natural successions of assemblages. There are often no distinct boundaries between different biotopes. The gradual transition means that attempts to discriminate between biotopes may have variable outcomes (Connor *et al*., 2004). Additionally, some communities may be temporary or transitional and represent a stage between two or more ‘stable’ biotopes. This could be due to periodic disturbance from abiotic conditions (i.e. winter storms), or biotic conditions (i.e. increased grazing during summer). In addition, certain habitats may be so variable that the position of a biotope along a gradient cannot be accurately defined (Connor, 1995).

Natural variation and patchiness of benthic communities pose problems when using the biotope classification for environmental impact assessment. For example, most offshore wind farm developments are proposed for areas of mixed, coarse or mobile sediments. Benthic community structure and species population dynamics are variable in such environments due to natural environmental fluctuations (Connor *et al*., 2004). At present, there is no adequate way of distinguishing between natural variation and an impact except by mapping biotopes and monitoring changes in their distribution over time (Ducrotoy, 2010). Extensive monitoring over several years would be required to distinguish between these.

Assessments of fishing impacts on biotopes have received little attention in the academic literature. In addition, there is little information on ecological quality and sensitivity of biotopes, both of which are needed in order to assess environmental impacts. Ecological quality of biotopes is not described in any hierarchical classifications due to spatial variability of species composition of biotopes; the classification was designed so that biotopes would be broad enough to incorporate some differences in species composition due to differences in local biotic and abiotic conditions. However, there have been attempts, using indicator species, to provide a biotic index (BENTIX), which describes the sensitivity or tolerance to disturbance for use in monitoring (Simboura and Zenetos, 2012). This is specifically aimed at achieving Environmental Quality Standards outlined by the EU Water Framework Directive. Although the use of biotopes in hierarchical classifications and Environmental Quality Standards is aimed at monitoring marine ecosystems, these two monitoring systems are very different and are not compatible or easily comparable.

Information on the sensitivity of UK biotopes to various forms of anthropogenic disturbance is essential for protection of biodiversity in the UK (Hiscock and Tyler-Walters, 2006). This has been investigated by the Marine Life Information Network (MarLIN). Scientific literature was used to assess species intolerance and recoverability from change in human activities or natural events, for key structural or functional species within the biotope, species which if lost would change the biotope, and/or species important to the function of the community within the biotope. However, where information was lacking, recoverability and sensitivity assessments were undertaken using qualitative judgments. The intolerance and recoverability were then combined to provide a meaningful assessment of the overall sensitivity of different biotopes to environmental change (Hiscock and Tyler-Walters, 2006).

Assessment of intolerance requires a specified level of environmental perturbation. This was addressed by the development, through expert consultation and review of the literature, of a set of ‘benchmark’ levels of environmental change against which sensitivity could be assessed (Hiscock and Tyler-Walters, 2006). The benchmarks allow intolerance and sensitivity to be compared against predicted effects of planned projects or proposals.

Although this sensitivity assessment is useful for planning and management of rare or at risk biotopes, there are only 75 subtidal biotopes out of a total of 155 that have sensitivity information. In addition, biotope importance varies by location; a biotope may be deemed important in one location due its rarity but may be common in another. Further, biotope sensitivity has only been used to inform management at a broad scale and no peer-reviewed literature has yet investigated the usefulness of this sensitivity scale for more detailed impact assessments.

In summary, the UK marine biotope classification provides a practical way of classifying assemblages by habitat type and is mainly used for surveillance and monitoring purposes because common standards ensure a consistent, integrated, UK wide approach. These classifications can be combined with acoustic data to create thematic habitat maps useful for spatial management. Little work has been done on assessing fishing impacts on biotopes in the academic literature. Some work has been undertaken assessing sensitivity of biotopes to specific impacts but these are not exhaustive and may differ regionally. The level at which the classification is used for impact studies is important as low classification levels may be too broad and high levels may result in patchiness potentially masking change. In turn, the level of classification will often be determined by the sampling methods used.

1.6. Methodologies for the assessment of benthic marine environment

Methods for ecological sampling of the benthic marine environment include dredges, grabs and corers for sedimentary habitats and diver transects and sampling, suction sampling, epibenthic trawls and videography techniques for rocky or biogenic reefs (Davies *et al.*, 2001; Saunders *et al.*, 2011).

Diver sampling (removal of species for further analysis), suction sampling and epibenthic trawls are destructive methods (some more than others) and are therefore rarely used for the monitoring in EMSs. Recommended methods for monitoring of protected areas are drop-down video, ROV, diver-operated video or towed video (Davies *et al*., 2001) due to the non-destructive nature of these sampling techniques (Tkachenko, 2005). Although these methods are favoured for monitoring of protected subtidal benthic habitats, they have limitations (as with any ecological sampling method) that must be understood in order to effectively use these sampling methods (Porter and Meier, 1992). The following section will only describe videography methods and then critically appraise these for the use in subtidal reef monitoring.

1.6.1. Videography as a tool for ecological sampling

Videography is commonly used for sampling of marine epibenthic organisms (Collie *et al.*, 2000a; Houk and Van Woesik, 2006). There are several ways that cameras can be used to sample the epibenthos. These include: mounted cameras on ROVs (Barry and Baxter, 1992), hand-held cameras operated by SCUBA divers (Harvey *et* *al.*, 2002), towed video (usually on submersible sleds), drop-down video (Saunders *et al.*, 2011), and point counts, predominately using baited remote underwater video (BRUV) or baited underwater video (BUV) cameras (Willis and Babcock, 2000). Sampling designs include: transect counts, measurements of size and biomass (which can be done using scaled fields of view or stereo video) (Shortis *et al.*, 2009; Langlois *et al.*, 2010). Commonly used sampling designs aim to detect spatial and/or temporal changes and patterns in community structure by determining abundance and distribution of epibenthic organisms (Maliao *et al.*, 2009). However, other epibenthic studies determine the abundance of specific organisms or species through visual censuses encountered during a transect (Willis and Babcock 2000, Eno et al., 2001). Videography sampling methods can also be used to identify biotopes (hierarchical classification) and allow frequency of biotopes to be analysed (Saunders et al., 2011).

Videography techniques are not limited to sampling the epibenthos. Physical habitat data can also be analysed; for example, particle size discrimination of the substratum can help determine habitat type (Bullimore *et al.*, 2013). This may eliminate the need for grab samples or dredges for this purpose.

1.6.2. Critical appraisal of videography methods

Videography is particularly useful for ecological monitoring. Biological communities can be surveyed for change over time by repeatedly sampling a location and analysing the changes in epibenthos between sampling events (Shortis *et al.*, 2009; Saunders *et al.*, 2011). This type of repeat sampling has been successfully used to show anthropogenic pressures such as coral bleaching (Riegl *et al.*, 2001) and fishing impacts on benthic ecosystems (Collie et al., 2000). In addition, the non-destructive nature makes this sampling technique ideal for long-term monitoring as the study site is not physically altered. Other sampling methods such as trawls, dredges, grabs and suction sampling may physically impact or degrade the benthos and therefore may introduce a certain amount of sampling bias for future studies (Collie *et al.*, 2000a)

Videography techniques enable data to be rapidly acquired (McDonald *et al.*, 2006). This decreases the time required for surveys, allowing larger areas to be sampled in the same amount of time but also making this a cheap way of sampling the epibenthos. In addition, studies have found a greater likelihood of detecting rare species, as the video footage covers a continuous, large area of the sea floor (Lam *et al.*, 2006).

The attachment of cameras to underwater equipment such as sleds, ROVs and protective metal frames means that underwater surveys are not limited by bottom time as visual census surveys using SCUBA divers are. Attaching cameras to underwater equipment has also allowed sampling at greater depths (Bullimore et al., 2013) or locations that are difficult or dangerous to dive at (McDonald et al. 2006).

Camera systems are cost effective in the long-term; the start-up costs can be high, but the running costs are usually low (Langlois *et al.*, 2010) and are kept down by the fact that cameras need not be deployed by workers with experience in epibenthic identification (Saunders et al., 2011). In contrast, underwater visual census (UVC) requires trained divers and trained taxonomists for in-situ identification of organisms. In addition, a significant advantage of videography over other sampling techniques is that a permanent record of the sample can be kept (Lam et al, 2006). This allows processing of the data at a later time or reanalysis of footage. The permanent record of the visual data is particularly useful for long-term monitoring studies as the images can be compared and re-compared if there are changes in the analysis methodology. The ease of duplicating and sharing of the data also facilitates collaborative research (Lam *et al.*, 2006).

Videography techniques also have limitations. Video footage is 2-dimensional meaning that cryptic species that may shelter under physical structures or benthos will not be detected (Saunders et al., 2011) and diversity estimates may be affected. Video footage image resolution can be a limiting factor in the identification of small and cryptic species (Davies et al., 2001) and in-situ identification by a taxonomist is, in the majority of cases, more accurate (Lam et al., 2006). However, digital image resolution has vastly improved over the past few years and is expected to be further increased with improvement of the technology (Lam et al. 2006).

Bad weather (due to large swell, currents and high wind speeds) and limited underwater visibility (due to high turbidity or low lighting) can reduce the quality of the footage, especially in shallow inshore environments where waves and currents more easily affect benthic substrate. This may result in sampling not being suitably sensitive (i.e. accurate and precise) for the identification of less faithful or smaller epibenthic organisms (Saunders *et al.*, 2011). However, the importance of this limitation when deciding upon ecological sampling technique depends on the aim of the work. If the survey only requires coarse data then a high level of detail may not be needed. For example, results from drop-down video surveys for an environmental impact assessment (EIA) baseline study for offshore renewables off Orkney and the Pentland Firth (Moore, 2009; Moore, 2010) showed that drop-down imaging systems were effective even in wave exposed or strong tidal locations; the data collected was sufficient for the identification of characterising species and biotope. Thus, should a modification or large change in species composition occur, the drop-down camera data provided enough detail to detect potential impacts (Saunders *et al.,* 2011).

In summary, the benthic marine environment can be sampled using different methods and equipment. Videography is the most commonly used non-destructive sampling tool. Videography techniques enable quick collection of large amounts of data and do not physically impact or degrade the benthos therefore reducing sampling bias for future studies. Videographic data also have limitations; samples provide lower estimates of species diversity than some other methods and there is a reduction in the quality of images when collected during bad weather. Investigation of fishing impacts at an ecosystem-wide scale requires an understanding of the benthic ecology of the marine environment (hierarchical classifications and sampling techniques) but also the distribution of fishing activity and the likely pressures this can exert on the ecosystem.

1.7. Fishing effort distribution

Reliable information on human uses of the marine environment is needed in order to understand the interactions between industries and the environment (Eastwood *et al.*, 2007; Daw, 2008; Breen *et al.*, 2014). This includes having recent and reliable habitat and marine usage spatial data at the appropriate scales (Crowder and Norse, 2008). Fishing effort distribution is not uniformly distributed across regions (Jennings *et al.*, 1999; Jennings *et al.*, 2012) and the footprint (area of habitat interacting with the fishing gear), and as such potential impacts, will vary depending on fishing methods (Jennings and Lee, 2012; Vanstaen and Breen, 2014). Information on effort distribution and habitat selection/use specific to different fisheries is needed in order to investigate fishing impacts and focus on areas which may have the highest fishing intensities or are occurring on rare or sensitive habitats.

Various methods have been used to describe and map fishing effort. Logbook schemes, vessel monitoring schemes (VMS), consultative approaches and aerial, land and boat based surveillance are the most widely used to date (Witt and Godley, 2007; Woolmer, 2009; Breen *et al.*, 2014). This section will discuss current methods for fishing effort mapping as well as the current limitations of these data.

1.7.1. Logbook schemes

Logbook schemes are the simplest method technologically for recording fishing activity and effort (Woolmer, 2009). Fishers will record information such as location, time/date, fishing gear/method used and catch description. Logbook schemes vary regionally and with vessel size. In the European Union, all vessels >12m are required to keep up-to-date logbooks of their operations (recorded every 24-hours) including detailed information on catch per species, effort (fishing hours), and location (EC Council Regulation No 1224/2009). However, the large spatial scale of the recorded information (ICES rectangles, approximately 30 x 60 nautical miles) has meant that fishing activity maps derived from these spatial data are often over-aggregated and will only be suitable for the most broad-scale marine planning (Woolmer, 2009).

In addition, many UK shellfisheries vessels fish inshore and are < 10m long. This is particularly the case in Northumberland where, between 2003 – 2014, there was a mean 70 (± 9)% < 10m fishing vessels in the NIFCA district (see chapter 2) and at least 90% of fishing effort is estimated to occur within the 6Nm limit (pers com NIFCA). Thus, a large proportion of vessels within the NIFCA district will not have to comply with the EU requirement of recording logs every 24-hours. However, Inshore Fisheries and Conservation Authorities (IFCAs) operate a variety of permit schemes which vary regionally. In Northumberland, fishers are required to submit monthly shellfish return forms with information regarding the numbers (and if available the weight in kilograms) of lobsters, crabs, velvet crabs, prawns and whelks, types and number of fishing gear employed, the area fished (closest port) (NIFCA, 2014). Monthly returns alone do not provide enough information to accurately map fishing activity, however these provide added effort information that can be incorporated with other fishing activity data allowing effort density distribution to be mapped (Turner, 2010; Turner *et al.*, 2015) (see chapter 2 for further details).

1.7.2. Vessel monitoring system

A vessel monitoring system (VMS) consists of a global positioning satellite (GPS) receiver, a satellite transmitter and a power backup (lasting approximately 72-hours) that will automatically record the location of the fishing vessel at sea every 2 hours (Defra, 2005a; Witt and Godley, 2007). VMS units are required to report 99% of all vessel GPS location with an accuracy of at least 500m as well as data on heading and speed (Defra, 2005a; Defra, 2005b). Although no information on fishing activity is provided by VMS this can be inferred by the movement of the vessel, i.e. a reduced cruising speed can indicate trawling (Lambert et al., 2011) or no movement can indicate static gear retrieval or deployment (Woolmer, 2009). Due to the large spatial scale of fishing data provided by VMS, several studies have accurately mapped fishing activity (Nilsson and Ziegler, 2007; Witt and Godley, 2007; Lambert *et al.*, 2011; Jennings and Lee, 2012). These studies indicated that there is minimal fishing activity within 12Nm of the coast, however in reality, this is because VMS is not collected for smaller vessels that fish inshore (Breen *et al*., 2014). Static gears such as shellfish pots tend to be fished from smaller vessels (< 10m) located close to shore and tend to have patchy distributions (Nielsen *et al*., 2013). Unless VMS were installed on < 10m vessels it would not be an adequate method for mapping fishing activity of inshore fisheries. Given the influence of the UK inshore fleet socially and economically (shellfish fisheries in the UK contributed 35% of the total 2011 UK landings with crab and lobster worth a combined value of £70.2 million (MMO, 2011)), it is of paramount importance to understand patterns of use for conservation and successful marine spatial planning (Eastwood *et al.,* 2007). Successful alternative methods for mapping inshore fisheries have used surveillance information (sightings at sea).

1.7.3. Surveillance Information – Aerial or at sea observations

At-sea observations of fishing vessels are made by fisheries management agencies during routine patrols. IFCAs and Marine Management Organisation (MMO) record data in the first 6Nm from the coast and the Fisheries Patrol Squadron of the Royal Navy outside 6Nm. Ariel surveys are also available from the MMO (Breen *et al*., 2014). Recorded information includes vessel name, registration, home port, geographic position (GPS location) and observed activity. Observed fishing effort has been mapped by Sea Fisheries Committees (Clark, 2008) with further refinements in mapping methods allowing fishing effort to be determined (Turner, 2010; Breen *et al*., 2014; Turner *et al.,* 2015). The accuracy of fishing vessel locations and observed activity means that data can produce maps with very high resolution (1 x 1km, Turner *et al*., 2015). Sightings were compared to consultation based methodology maps. No significant differences were found between methods (Turner, 2010). Both methods were deemed to accurately represent fishing activity in the Northumberland IFCA district (Turner *et al.,* 2015).

Successful consultative approaches to mapping fishing activities have also been used (Des Clers *et al.*, 2008; Woolmer, 2009; Shepperson *et al.*, 2014; Turner *et al.*, 2015). These have provided spatial local ecological knowledge (LEK) which is increasingly being integrated into management plans. However, consultative approaches to mapping fishing activities collect data on a broader scale (areas of several km^2 used for fishing) whilst sightings data have the potential to shed light on smaller scale differences in fishing activities ($1km^2$).

1.8. Thesis outline and study site

In summary, there are still few studies that investigate whether shellfish potting has environment impacts (Coleman *et al*., 2013). The need for conclusive evidence on potting impacts is important due to the sustained growth of the industry and its importance nationally (MMO, 2011). Long-term impacts of potting were shown not to significantly impact epifauna in a study experimentally potting in fished and non-fished locations during a four year study (Coleman *et al.,* 2013). However, recovery of habitats exposed to sustained potting impacts could take longer than four years (Babcock *et* al., 2010; Coleman *et al*., 2013). The use of hierarchical classifications could provide a standardised and novel method for investigating potting impacts, however, the level of classification used will be important in determining the scale of impact. Videographic and photographic methods provide cheap, rapid and repeatable ways of ecological monitoring, although adverse weather conditions can negatively affect data collection and the 2-dimensional nature of the image may reduce diversity estimates. This thesis further explores understudied static gear potting impacts as part of an evidence based approach to management for protected sites in the UK.

1.8.1. European Marine Sites

There are currently 81 EMSs in the UK (Natural England, 2013). EMS collectively describes Special Areas of Conservation (SACs) and Special Protection Areas (SPAs) that are covered by tidal waters (Natural England, 2013). SACs and SPAs contain animals, plants and habitats that are considered rare, special or threatened within Europe. As such SAC features in EMSs are subject to regular condition monitoring required under Article 17 of the Habitats Directive, which aims to ensure that EMSs continue to be areas of international importance for the quality of their marine habitats and species (AONB Partnership and EMS Management Group, 2009). In order to meet the conservation objectives for EMSs, species and habitat monitoring for each SAC is undertaken every six years continually adding evidence to databases. Since before designation, some EMSs have been subject to long-term monitoring (see ‘Regional Marine Nature Conservation Review (MNCR) Series’ by the Joint Nature Conservation Committee (JNCC)). The primary aim of EMSs is to maintain the quality of marine habitats and species; however some activities are also allowed, including commercial fishing.

1.8.2. The Berwickshire & North Northumberland Coast European Marine Site

The Berwickshire & North Northumberland Coast European Marine Site (BNNC EMS) (Fig 1.2) has been inhabited and exploited for food and trade for centuries. Stretching 115km from Alnmouth in North-east England to Fast Castle Head in South-east Scotland, it incorporates 635km^2 of shoreline and coastal waters. The area includes Lindisfarne, St Abbs, the Farne Islands and the Eyemouth voluntary marine reserve (AONB Partnership and EMS Management Group, 2009), hosting a diverse range of marine ecosystems and habitats including rocky shore line, intertidal mudflats, rocky reefs, sea caves and sand beaches (Brazier *et al.*, 1998).

The BNNC EMS (Fig 1.2) was designated in 2000, in part because of the importance of ‘reef’ biotopes (Hartnoll, 1998), and incorporates Special Areas of Conservation (SACs) and Special Protection Areas (SPAs) with marine areas that are covered by continuously or intermittently tidal waters (Natural England, 2013). BNNC EMS habitat monitoring is undertaken every six years (Mercer *et al.*, 2003; Mercer, 2012), although even prior to the designation long-term habitat mapping and monitoring was undertaken (see (Edwards, 1983; Foster-Smith and Foster-Smith, 1987; Connor, 1989; Holt, 1994; Brazier *et al.*, 1998). Local management aims to ensure that the EMS continues to be an area of international importance for the quality of its marine habitats and species although fishing is still currently permitted in the BNNC EMS (AONB Partnership and EMS Management Group, 2009).

Seventy eight vessels habitually fish within the Berwickshire and North Northumberland Coast European Marine Site (BNNC EMS). Potting, which includes using traditional creels and parlour pots, mainly targets European lobster (*Homarus gammarus*), brown crab (*Cancer pagurus*), velvet crab (*Necora puber*), and is the most used fishing technique (93% of vessels) with some vessels using drift nets targeting salmon (<6%) and a very small number of vessels using towed gears (<0.2%)(Garside *et al.*, 2003). Indeed, the economic importance of shellfish fisheries in the UK has increased (Molfese *et al.*, 2014), particularly over the past 10 years (MMO, 2011) due to declines in demersal and pelagic fish landings (Turner *et al.*, 2009; Molfese *et al.*, 2014). Shellfish fisheries in the UK contributed 35% of the total 2011 UK landings with crab and lobster worth a combined value of £70.2 million (MMO, 2011). Despite this growing importance, they remain relatively lightly regulated with shellfishery byelaws conceived and enforced by local Inshore Fisheries and Conservation Authorities (IFCAs).

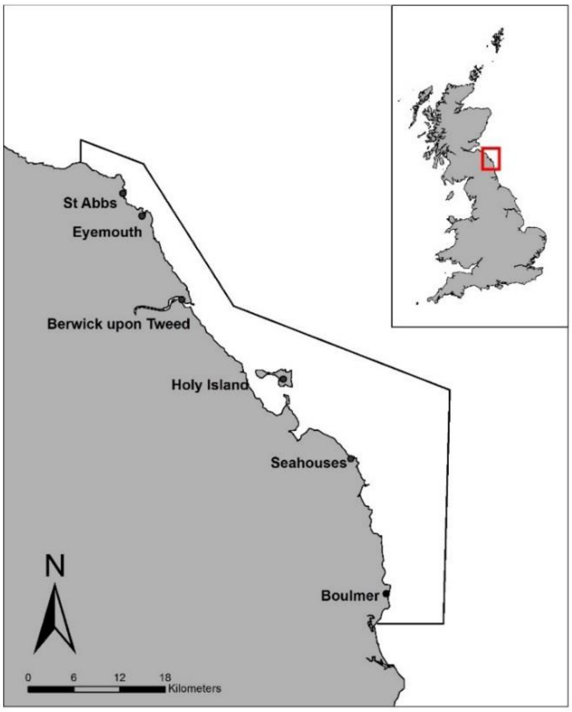


Fig 1.2. Berwickshire and North Northumberland Coast European Marine Site. Towns (red dots) and European Marine Site boundaries (black line) are shown.

1.8.3. Defra's revised approach to management of fisheries in European Marine Sites

On August 15th 2012, the Department for Environment, Food and Rural Affairs (Defra) announced a revised approach to the management of commercial fishing activities within European Marine Sites (EMSs) in England (Defra, 2013). Habitats Regulations Assessment (HRA) of each current fishing activity within EMSs with respect to its potential impact on conservation features of interest is required. Should a likely significant impact occur or be thought to be possible (alone or in combination with other plans or projects) an appropriate assessment must be carried out, which will also inform on mitigation (management) options that will ensure site integrity is maintained.

Conservation features of interest within EMSs identified as potentially impacted by fishing include subtidal reefs, in particular, kelp forest communities and sublittoral faunal turf communities (Natural England *et al.*, 2012). Subtidal hard substrata support the highest diversity of marine organisms in temperate regions (Hartnoll, 1998), as many require attachment to a hard surface. Shallow infralittoral zone communities (extending down 10-20 m depth) are algal dominated whilst deeper circalittoral communities are largely animal dominated (Connor, 1995), due to the attenuation of light in the water column. Fishers in the BNNC EMS target these rocky habitats due to the high abundances of economically valuable lobsters present (NIFCA, 2013, pers. comm.; Galparsoro *et al.,* 2009). These marine communities can be affected by fishing activities due to their high species diversity and high abundances of long-lived, erect and presumably fragile species (Roberts *et al.*, 2010).

The priority and focus for the revised approach to fisheries management in EMSs was initially on trawling and other mobile gears on subtidal reef features. Investigation into impacts of bottom towed gears within EMSs was being undertaken at the time of writing with some previous literature already existing (Sewell *et al.*, 2007). However, further work is also required to understand lower-level impacts on benthic habitats, including subtidal rocky reef, from possibly less destructive static gears such as pots and creels as there is little scientific evidence for environmental impacts that these may have (Eno *et al.*, 2001; Coleman *et al.*, 2013).

Potting activity is generally assumed to have little physical impact on these hard substrate habitats and epibenthos. The few potting impact studies undertaken to date have focused on assessing impacts using indicator species perceived to be sensitive to potting due to their life history traits, e.g. erect, fragile and sessile species (Eno *et al.*, 2001; Shester and Micheli, 2011; Coleman *et al.*, 2013). However, potting may alter trophic dynamics through the removal of keystone species incurring ecosystem changes (Siddon and Witman, 2004). There is currently a lack of peer-reviewed research into long-term in-direct impacts of potting as well as direct potting impacts through abrasion and crushing of epibenthos. These are crucial to our understanding of pot fisheries interactions with the environment (Siddon and Whitman, 2004). In addition, a lack of habitat information and fishing pressure on these is also lacking – although often recommended for an ecosystem based approach to management (Caveen *et al.*, 2014).

1.8.4. Aims and objectives

The aim of this thesis was to examine whether potting activity in the BNNC EMS and the wider NIFCA region is likely to impact epibenthos of reef habitats. Firstly, analysis of spatial and temporal potting effort and fisher habitat preferences allowed potting pressure on various habitats to be investigated. This provided an evidence base for the investigation of long-term ecological change in the BNNC EMS (through analysis of historical monitoring data between 2002 – 2011), and whether any changes found could be attributed to the intensity of shellfish potting activity. Finally, quantification of direct impacts through *in-situ* experimental fishing was investigated.

The objectives were to:

1. Investigate decadal spatial and temporal trends in potting effort in the NIFCA district (chapter 2) and fisher habitat use (chapter 3).
2. Investigate the adequacy of using biotope analysis (chapter 4), taxonomic composition of species (chapter 5), species richness (chapter 5) and indicator species (chapter 5) for detecting ecological change in the BNNC EMS using existing data.
3. Discuss the usefulness of a) frequently used monitoring methods such as hierarchical classification systems (Chapter 4) b) videography data (chapter 5) c) long-term monitoring data (Chapter 4 and 5) d) photoquadrat data (Chapter 6) for use in ecological studies.

4. Investigate direct impacts of potting through *in-situ* observation of experimental potting in two commonly found rocky habitats in Northumberland (Chapter 6).

Chapter 2 . Decadal Inshore Fishing Activities in Northumberland: Spatial, Temporal Changes

2.1. Introduction

In the UK, static gear fisheries are an important component of the fishing industry, landing 86,600 tonnes worth £173.6 million per annum and landing 41% of the shellfish in 2014 (MMO, 2015). These include high value species such as lobster and crab which are primarily caught inshore (defined as coastal waters out to 6Nm) using pots or traps (MMO, 2015). As a result, potting has an important socio-economic function particularly in inshore fisheries (Turner *et al.*, 2009; Kaiser, 2014).

The effects of mobile fishing gears on the marine environment have been well documented (Kaiser *et al.*, 1996; MacDonald *et al.*, 1996; Collie *et al.*, 1997; Collie *et al.*, 2000a; Collie *et al.*, 2000b; Kaiser *et al.*, 2006; Lambert *et al.*, 2011), whereas static fishing gears have been assumed to be relatively benign (Eno *et al.*, 2001; Lewis *et al.*, 2009; Shester and Micheli, 2011; Coleman *et al.*, 2013). There is limited evidence to support the latter (Shester and Micheli, 2011). The need to address this information deficit has become increasingly relevant due to the requirements to assess the effects of fishing activities in UK designated conservation areas: European Marine Sites (EMSs) and Marine Conservation Zones (MCZs). Many of these conservation areas are located inshore and are currently subject to static-gear fishing (MMO, 2016). Effects on the marine environment of static fishing gears requires an understanding of the distribution, frequency and intensity of these fisheries (Kaiser, 2014).

Geographical Information System (GIS) based decision-support tools have been successfully used in Australia and the USA to enable managers to model different scenarios and Marine Protected Area (MPA) designs that satisfy both conservation and socio-economic objectives (Possingham *et al.*, 2000; Ball *et al.*, 2009). Similar methods are increasingly being used for designation of protected areas in the UK and require accurate fishing activity and effort information (Smith *et al.*, 2009), but to date the resolution of these data is inadequate for this purpose. Fishing has the largest spatial footprint of human activities in the marine environment (Eastwood *et al.*, 2007; Diesing *et al.*, 2013), making it crucial that fishing interactions with marine benthos for prediction or mitigation of potential impacts are better understood (Eno *et al.*, 2013).

To date, research has predominately focused on mapping mobile gear fishing activity using VMS (Nilsson and Ziegler, 2007; Stelzenmüller *et al.*, 2008). However, a primary limitation of VMS data is that it is only recorded for large vessels (> 12m) and as such, inshore fishing fleets which are largely composed of small vessels (< 12m) are not well represented by VMS (Breen *et al.*, 2014). This is particularly the case in Northumberland where, between 2003 – 2014, a mean 70 (± 9)% < 10m fishing vessels in the NIFCA district (see section 2.3.1) and at least 90% of fishing effort occurred within the 6Nm limit (NIFCA pers. comm.).

Alternative approaches to mapping inshore fishing activity, such as surveillance methods, have successfully described distribution and intensity of various fishing gears, however, only over a single period of time (Breen *et al.*, 2014; Turner *et al.*, 2015). Fishing spatial patterns can vary over time (Kaiser et al., 2002; Nilsson and Ziegler, 2007), the availability of target species, gear and territoriality being important drivers (Acheson, 1975; Rijnsdorp et al., 2001; Turner et al., 2012). These inter-annual variations in fishing activity may make short-term studies inadequate (Lynch, 2014), yet the spatio-temporal variability must be understood to inform appropriate management. This research, building on previous inshore mapping methods, investigated spatial-temporal changes of the static gear pot fishery in Northumberland. The use of static gears may increase in inshore UK waters (Turner *et al.*, 2012) due to declines in the demersal and pelagic fish landings (Turner *et al.*, 2009; Molfese *et al.*, 2014). In light of the potentially increasing use of static gear, this research sought to test the hypothesis that pot fishing effort had increased in Northumberland coastal waters (2004 – 2014) and that changes varied in extent and magnitude between years. The use of these data for fisheries monitoring and management are discussed.

2.2. Methods

2.2.1. Observed fishing activity

Sightings of fishing vessel activity were recorded by NIFCA fishery officers on routine patrols and were combined with landings data to estimate and map fishing activity between 2004 - 2013. The methodology of combining sightings and landings data was developed by Turner (2015)**,** built on previous work (Clark *et al.*, 2008; Des Clers *et al.*, 2008; Turner *et al.*, 2009; Turner, 2010; Spencer, 2013) and was adapted for this research (Fig 2.1).

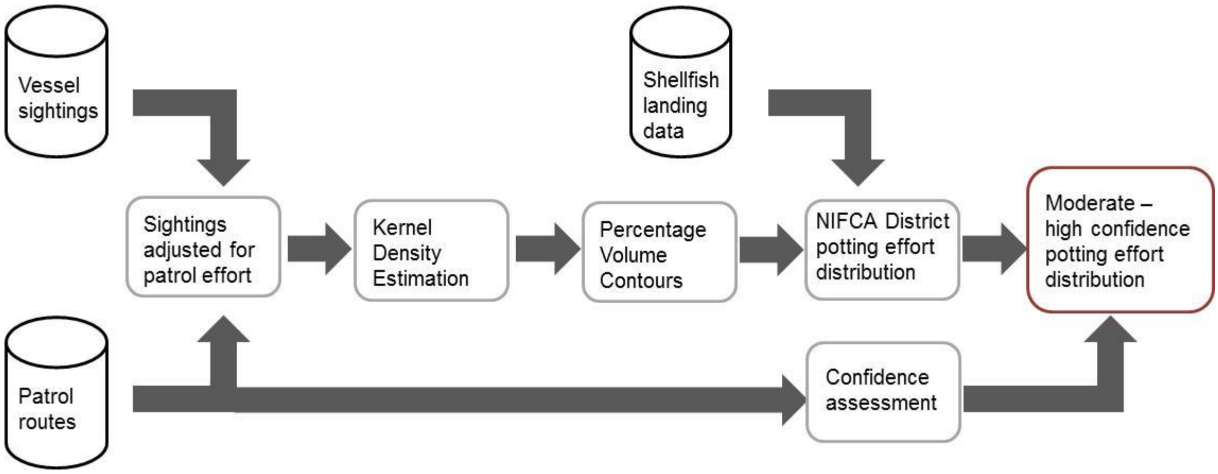


Fig 2.1. Diagram of GIS processes undertaken to spatially map distribution of potting effort densities in Northumberland IFCA district. Raw data (black cylinders), GIS mapping procedures (grey boxes) and final potting effort density distribution map (red box) are shown.

2.2.2. Vessel sightings

Fishing vessel sightings were recorded during routine NIFCA patrols between 2004 and 2013 (Table 2.1). Vessel name, registration, home port, geographic position and observed activity (i.e. hauling or deploying pots, steaming) were recorded. Sightings in 2004 – 2013 of potting vessels targeting crab and lobster, and recorded as deploying or hauling pots, were mapped as point data using ArcView GIS version 10.2 (ESRI, 2014) (Fig 2.2). All sightings outside of the NIFCA district boundaries were excluded from analysis. Sightings data provide strong confidence of association with actual fishing activity because of the direct recording by the NIFCA officers of activity such as shooting, hauling or attendance of gear by a fishing vessel.

Table 2.1. Shellfish vessel sightings and NIFCA patrol route data. Not available (N/A).

Year	Shellfish vessel Sightings	Unique vessels observed	NIFCA patrols	Patrol routes available (%)*
2004	1159	91	104	90 (87)
2005	771	81	99	47 (47)
2006	749	83	86	4 (4.7)
2007	515	N/A	86	49 (58)
2008	433	N/A	85	56 (75)
2009	529	N/A	71	45 (63)
2010	546	98	72	59 (81)
2011	539	72	101	79 (78.2)
2012	496	69	85	84 (98.8)
2013	479	69	83	71 (85.5)
2014	490	65	86	76 (88.4)

* The number of patrol routes (GIS shapefile) which were provided by the NIFCA. Not all patrol routes were available.

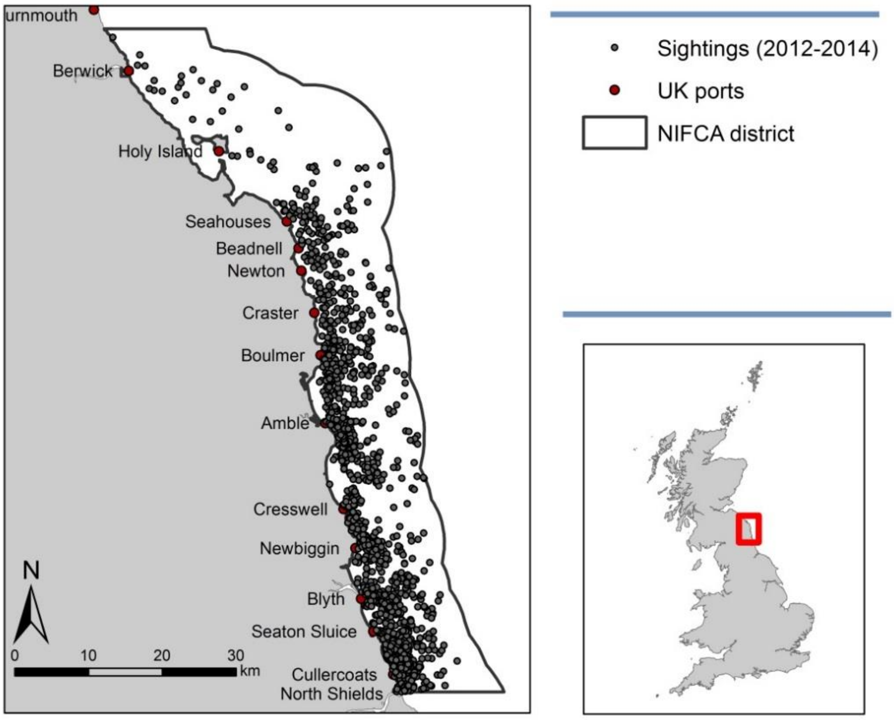


Fig 2.2. Example of potting vessel sightings (grey dots) in the NIFCA district (black line) during 2012 - 2013.

Patrol effort, i.e. the number of times the NIFCA enforcement vessel went to sea, varied between years (Table 2.1). In order to standardise sightings across years

(Table 2.2), sightings data were randomly reduced to 71 days for each year (the minimum patrol number was 71 in 2009 (Table 2.1)). Examination of the mean number of active vessels at each port in the NIFCA district (2004 – 2013) revealed that these were largely similar ensuring that sightings were not biased by port size within the district during the study period (Fig 2.3).

Table 2.2. Standardised annual Northumberland potting fleet sightings (2004-2014).

Year	Standardised Sightings
2004	905
2005	578
2006	672
2007	478
2008	378
2009	509
2010	532
2011	451
2012	452
2013	431
2014	443

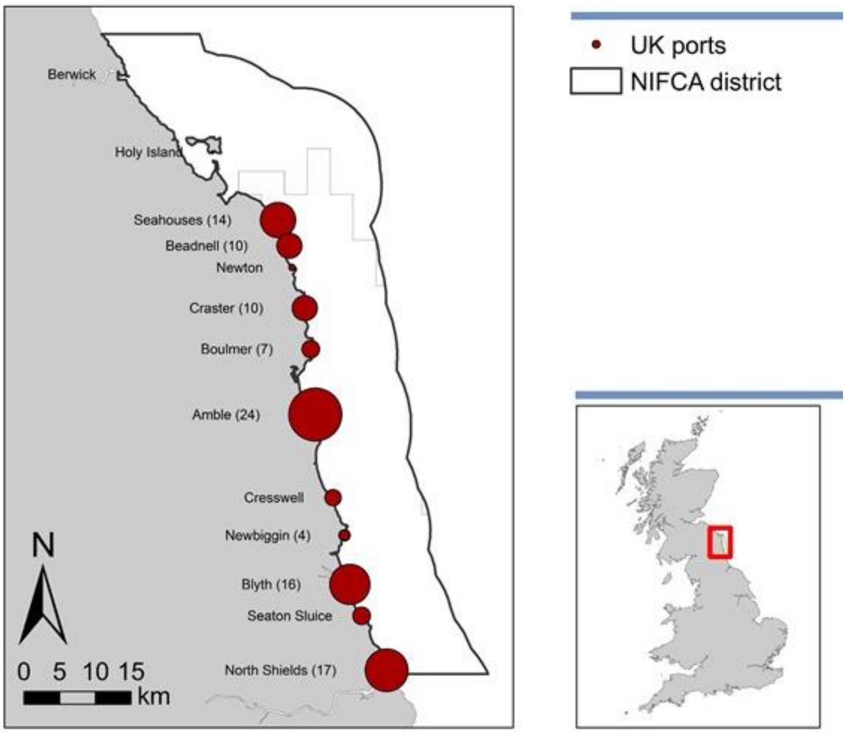


Fig 2.3. Mean number of active vessels per port (2004 – 2013) represented by size of the red circles. Values noted next to port name.

2.2.3. Vessel tracks

There was a bias in the vessel sightings data due to the patrol vessel being moored in the river Tyne resulting in a high number of patrol tracks in the south compared to the north of the district (Fig 2.4); there was a higher chance of observing vessels in the south compared with the North of the district. The frequency and location of fishing vessel sightings will be influenced by the timing and route of the NIFCA patrols (Breen *et al.*, 2014; Turner *et al.*, 2015). Thus, sightings were spatially adjusted for patrol effort. NIFCA vessel GPS locations were recorded at regular intervals during patrols (2004 – 2014) and tracks were georeferenced and displayed in ArcGIS as polylines (Fig 2.4). A mean of 73% of patrol routes were available: 699 out of a total 958 patrols undertaken by the NIFCA between 2004 and 2014 (Table 2.1).

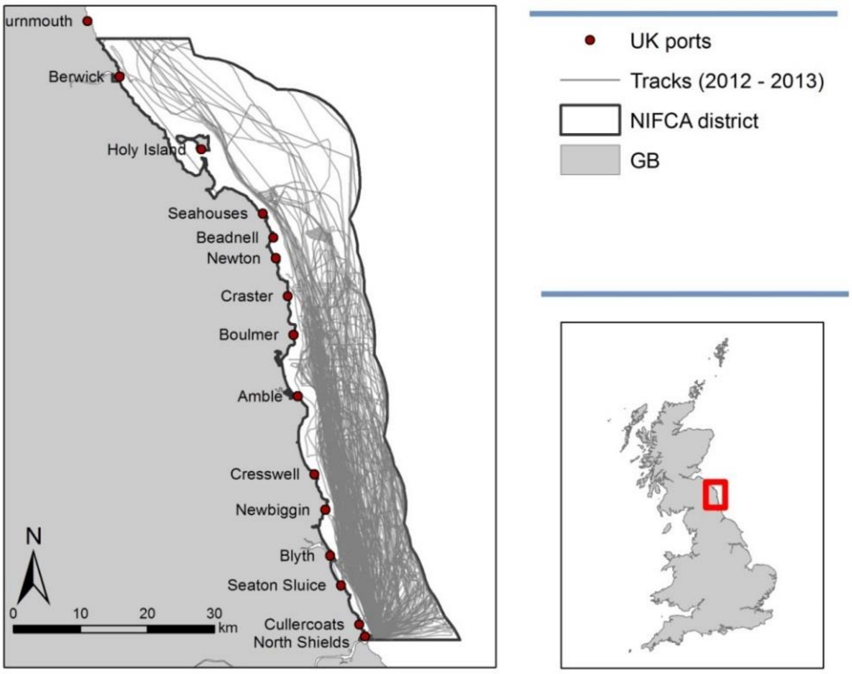


Fig 2.4. Example of patrol vessel track data for the NIFCA district (2012-2013).

A 3Nm^2 grid was created in ArcGIS and superimposed over the NIFCA district (Clark *et al.*, 2008; Turner, 2010) (Fig 2.5): if a fishing vessel was within a single 3Nm^2 grid cell at the same time as the patrol vessel, and visibility was adequate, it is assumed that the fishing vessel would be observed and recorded (Turner, 2010)

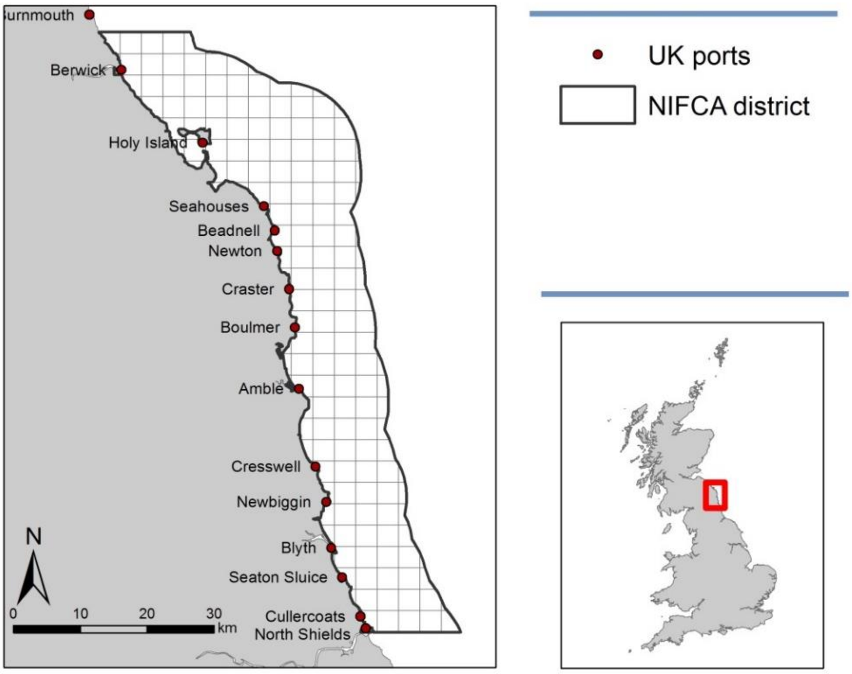


Fig 2.5. 3Nm^2 grids in the NIFCA district.

The number of patrol routes falling within each 3Nm^2 were counted and a measure of patrol effort (PE$_1$) for each cell (the proportion of patrols passing through each cell) was calculated (Eq 1) (Turner, 2010):

$$\textbf{Eq 1:} \qquad PE_1 = \frac{\boldsymbol{n}}{\boldsymbol{N}}$$

n is the number of patrols passing through the cell and N the total number of patrols. However, some sightings occurred in grids where no patrol vessel tracks were recorded. This could either mean that the patrol data were missing or that the observation distance of the patrol vessel was greater than assumed (Turner, 2010). In order to account for this, a second measure of patrol effort was calculated for each grid which assumes that patrol effort decreases linearly with distance from patrol routes (PE$_2$, Eq 2):

$$\textbf{Eq 2:} \qquad PE_2 = \frac{\boldsymbol{D}_{\mathbf{max}} - \boldsymbol{D}_{\boldsymbol{g}}}{\boldsymbol{D}_{\mathbf{max}} - \boldsymbol{D}_{\boldsymbol{min}}}$$

Where the inverse Euclidean distance from the centre of each grid cell to the centre of the closest grid cell which contains a patrol route (D$_g$) is normalised as a proportion of the minimum distance from the patrol route to the centre of the grid cell containing the patrol route (D$_{min}$) and the maximum distance from the centre of the grid cell containing the patrol route to the centre of the furthest grid cell from it (D_{max}) (Turner, 2010; Spencer, 2013) (Fig 2.6).

Eq 1 and 2 were combined and used to calculate the overall patrol effort (PE, Eq 3) for each grid square. The resulting values positively weighted sightings data in cells with low patrol effort and negatively weighted sightings in cells with high patrol effort.

Eq 3: $$PE = (1 - PE_1) + (1 - PE_2)$$

Calculation of PE (Eq 3) is illustrated in Fig 2.6, where PE_1 is illustrated by four patrol routes (blue lines, $N = 4$) with example grids showing the number of patrol routes passing through them ($n = 1 - 3$) (Fig 2.6). PE_2 illustrated for the red grid cell, uses distances (red lines) to the closest grid cell containing a patrol route (D_g), the minimum distance from the patrol route to the centre of the grid cell containing the patrol route (D_{min}) and the maximum distance from the centre of the grid cell containing the patrol route to the centre of the farthest grid cell from it (D_{max}) (Fig 2.6).

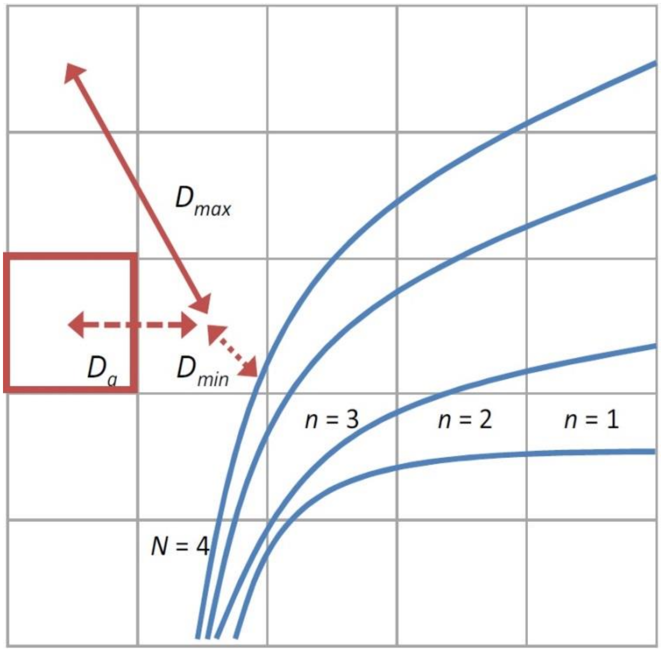


Fig 2.6. Example of parameters used in calculation of patrol effort for 3Nm^2 grid cell outlined in red. Blue lines represent patrol routes passing through 3Nm^2 grids (N =4; n =1-3). Red lines represent distances between grid cells and gird cells containing patrol routes (from Turner, 2010).

2.2.4. *Mapping observed fishing activity*

Sightings data were pooled in 2-year groupings (2006 - 2007; 2008 - 2009; 2010 - 2011 and 2012 – 2013), due to the limited number of vessel sightings per year, which was in part due to the small number of fishing vessels operating from some ports, and the limited number of routine patrols undertaken by NIFCA (Turner et al., 2009). Numbers of sightings standardised by patrol effort were similar between two-year groupings (Table 2.3). Grouping data into two year intervals may mask annual change (Turner *et al*., 2015), but it allowed longer-term trends to be explored with greater confidence.

Table 2.3. Standardised annual Northumberland potting fleet sightings by group.

Year	Sightings
2004 – 2005	1483
2006 – 2007	1150
2008 – 2009	887
2010 - 2011	983
2012 - 2013	926

Adjusted vessel sightings point data were transformed to continuous surface data in order to provide information on relative intensity of fishing activity in different areas, including areas where samples were not available (De Freitas and Tagliani, 2009). A non-parametric kernel density estimation (KDE) method was chosen over other raw point data or interpolation methods because it does not assume that fishing activity is normally distributed or a continuous spatial coverage (Alessa *et al.*, 2008).

A KDE raster with cell size 100 x 100m using a quadratic kernel function was produced in ArcMap (10.2) (Esri GIS) (Silverman, 1986). A smoothing parameter, termed bandwidth, determines the circular area (or kernel) around a given location within which data points contribute to the probability estimate and is a critical assumption when using KDE (Wand and Jones, 1995; Van Der Veen and Logtmeijer, 2005). A small bandwidth will result in patchy density distributions resulting in a “ring around points effect” (Kie *et al.*, 2010). However, a large bandwidth will result in a greater smoothing effect (or a more generalized surface) which risks removing meaningful spikes from the original data distribution (Carlos *et al.*, 2010). In this case, the data distribution was examined in order to inform choice of bandwidth (Wand and Jones, 1995; Kie *et al.*, 2010). The distribution of the density data was unimodal, fairly symmetric and did not have large tails. Thus the *normal distribution* *approximation* bandwidth estimation method (Silverman, 1986) most closely resembled the reference distribution of normally distributed data. However, this method, when used with movement data that exhibits clumping in location (as was the case here), could result in a bandwidth that is too large, over-smoothing the data and creating a utilization area that is inaccurate (Kie *et al.*, 2010). Thus bandwidth was reduced to a fixed proportion of 0.8 (Bertrand *et al.*, 1996; Kie and Boroski, 1996; Kie *et al.*, 2002; Kie *et al.*, 2010) ensuring that detail in the density distribution was not lost whilst reducing the "ring around points" phenomenon (Silverman, 1986).

2.2.5. Mapping distribution of potting density

Percentage volume contours (PVCs), which delineate the smallest area accounting for a specific proportion of the probability density distribution (St. Martin and Hall-Arber, 2008), were created from the potting activity KDEs using the 'Isopleth' tool in GME (Beyer, 2012). A 70% PVC delineates the area within which there was a 70% probability of observing vessels fishing within the district. PVC Polygons of 50, 60, 70, 80, 90 and 95% were created, uploaded into ArcGIS (v 10.2), clipped to the extent of the NIFCA district and area calculated (Turner, 2010; Spencer, 2013). 95% was chosen over 100% in order to minimise the influence of possible positional errors or mis-identification of vessels, providing a statically robust vessel sightings estimate (Turner *et al*., 2015).

Shellfish permit holders in Northumberland are legally mandated to provide monthly data on vessel home port, landing port, number of pots worked per month, number of landings and weight of landings. Monthly shellfish landings data for 2001 – 2014 were obtained for the NIFCA district. Data were divided into bi-annual groupings (2004 - 2005; 2006 - 2007; 2008 - 2009; 2010 - 2011; 2012 - 2013) and number of pots worked for each vessel per month over the whole NIFCA district was used to calculate fishing effort (f, number of pots year^{-1}):

Eq 4: $$\textit{Mean Annual f} = \frac{\boldsymbol{\Sigma}\ \textbf{pots set per month}}{\boldsymbol{n}}$$

Where *n* is the number of years. For years 2006 – 2009 fishing effort data were missing information for vessels > 10m. These data were collected by the Marine Management Organisation (MMO) and have been requested, although at the time of writing were not available. The total number of pots deployed between 2006 and

2009 were estimated by averaging the proportion (%) of pots fished per year by vessels > 10m for years 2010 – 2014. The mean proportion of fished pots by vessels > 10m was added to the available data for years 2006 – 2009. Average proportion of pots fished by vessels > 10m was 14.3 ± 5.8%.

In order to calculate annual potting density, 50, 60, 70, 80, 90 and 95% of the mean annual fishing effort (Eq 4) was calculated and proportioned to the corresponding PVC polygon (Turner, 2010). For example, 50% of annual fishing effort was allocated to the 50% PVC with pot density at sea calculated as number of pots km $^{-2}$ year $^{-1}$.

2.2.6. Confidence assessment

Some areas were infrequently or never patrolled (Fig 2.4), resulting in these areas having uncertain or unknown fishing intensity and distribution (Breen *et al.*, 2014). A confidence assessment similar to that used by Breen *et al.* (2014) was undertaken to provide a simple measure of uncertainty. The quality of the sightings data was considered good with fishing activity recorded using GPS based methods (Breen *et al.,* 2014). The frequency of patrol tracks contained within each 3Nm^2 grid cell (section 2.1.2) was used to produce a confidence classification (Table 2.4).

Table 2.4. Confidence classification used for the confidence assessment (modified from Breen *et al*., 2014).

Patrol effort over 1 year	**Confidence class**
More than once in 2 weeks	High
Less than once in 2 weeks but more than once in 2 months	Moderate
Less than once in 2 months	Low

The resulting confidence class for 2004 – 2013 patrols within each 3Nm^2 grid cell was mapped; 63.3% of the NIFCA district had moderate – high confidence (Fig 2.7). Potting effort distribution in areas with low confidence were excluded from the analysis.

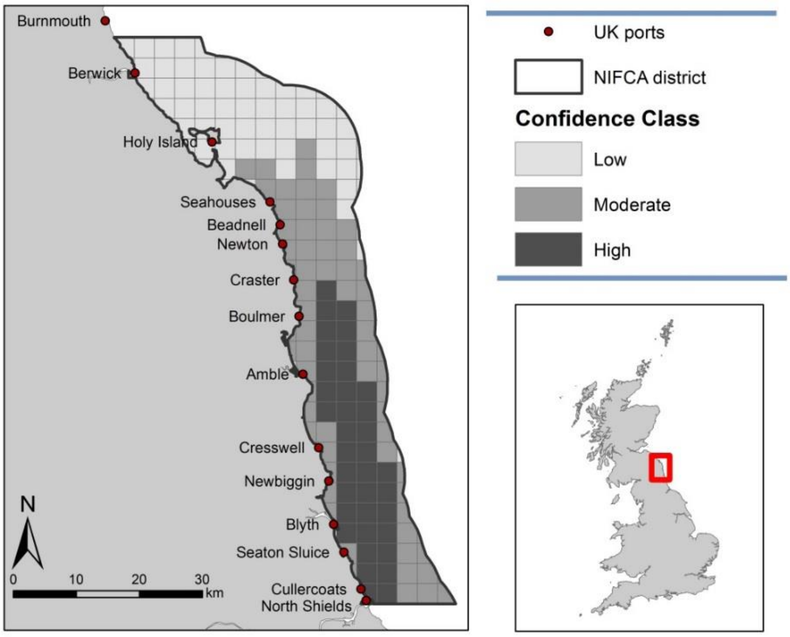


Fig 2.7. Confidence data layer for the NIFCA district annual potting density map (2004 - 2013).

2.2.7. Data analysis

Spatial and temporal trends in fishing effort distribution were investigated in three ways. Firstly, temporal trends in fishing effort data for the district (number of pots deployed per month) were investigated. Secondly, temporal changes in fishing vessel distribution were examined. Finally, these two analyses were combined to investigate temporal changes in distribution of fishing effort. By including these three analyses it was possible to determine whether changes in fishing effort distribution were driven by changes in fishing effort, changes in fishers' spatial use of the NIFCA district, or a combination of the two.

2.2.7.1. Temporal changes in fishing effort

Temporal trends in fishing effort were analysed using NIFCA landings data (2001 – 2014). This included aggregating data on number of permits issued per year, active vessel number, total pots worked in the district, median pots deployed per month and total pots worked per year. Changes in fishing effort (number of pots fished per month) through time were analysed using a linear regression model with number of active vessels as a covariate. In order to account for the inherent seasonality a harmonic function was included but these covariates were non-significant and it was deemed that monthly active vessel number encapsulated seasonality. As discussed in section 2.2.5, total number of pots deployed between 2006 and 2009 were estimated by adding the average proportion (%) of pots fished per year by vessels >

10m for years 2010 – 2014 (14.3 ± 5.8%) to the available < 10m vessel data. In order to ensure that these estimates did not affect results, the linear model was also run using the lower and higher estimates of number of pots fished by > 10m vessels (8.5 and 20.1% respectively). Results using the lower and higher estimates did not change the overall trend with all variables significant. The Rstudio packages used, model code and results using lower and upper estimates of pot number for > 10m vessels for 2006 – 2009 are presented in appendix 1.

2.2.7.2. Spatial-temporal changes in fishing distribution

In order to highlight differences in space use between years, biannual KDE maps were compared using absolute thematic change maps showing pixels with differing values (Remmel, 2009). This was undertaken by merging KDEs of vessel sightings in ArcGIS for the years concerned (E.g. 2004 – 2005 and 2006 – 2007) and subtracting the number of vessel sightings (Number of vessel sightings km^{-2}) from each year – this highlighted areas where fishing vessel sightings increased or decreased and by how much (Fig 2.8, C).

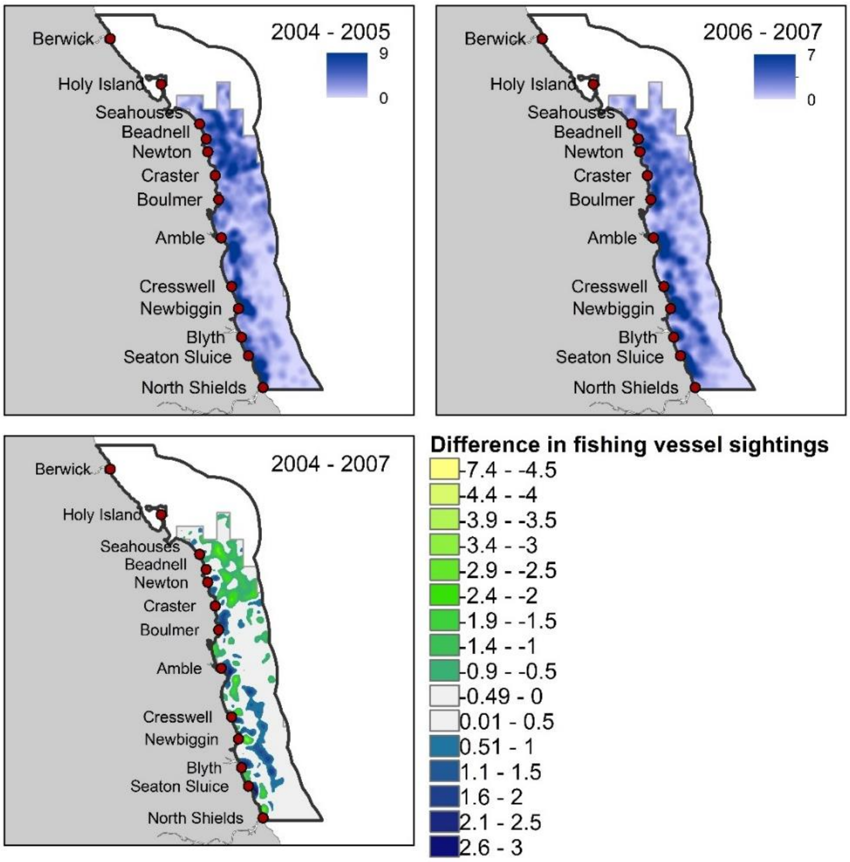


Fig 2.8. KDE of vessel sightings for the NIFCA district for 2004 – 2005 (fishing vessels km $^{-2}$ year $^{-1}$) (A); and 2006 – 2007 (fishing vessels km $^{-2}$ year $^{-1}$) (B); Difference in vessel sightings between 2004 – 2005 and 2006 – 2007 (fishing vessels km $^{-2}$ year $^{-1}$) (C).

2.2.7.3. Spatial-temporal changes in fishing effort

Spatial-temporal changes in Northumberland shellfishing effort were investigated through the comparison of fishing effort distribution maps created in ArcGIS (2004 - 2005; 2006 - 2007; 2008 – 2009; 2010 - 2011; 2012 - 2013). There are few established methods for statistically comparing the distribution of variables with numerical values, such as the intensity of fishing activity (Hagen-Zanker 2006). Until recently, spatial comparison of maps has been undertaken using ‘cell by cell’ methods (Hagen-Zanker, 2006) or through quantification of map similarity whilst accounting for spatial structure (Hargrove *et al.*, 2006), including taking into account spatial autocorrelation (Hagen-Zanker, 2009). However, these map comparison methods, many of which were primarily developed for accuracy assessment of simulated maps, give an indication of similarity but not significance of change. In order to highlight long-term trends and significant differences, maps were analysed by randomly sampling locations in two maps (i.e. 2004 – 2005 vs 2006 – 2007, 2004 – 2005 vs 2008 – 2009, etc) and comparing fishing effort values of these locations between years. Random samples were permuted using Monte Carlo simulation, a novel application of this analysis method. Firstly, fishing effort maps were converted to raster format in ArcGIS (pixel sixe 100 x 100m) and imported to RStudio (RStudio, 2012). Each raster contained 88,604 unique pixels, of which 5000 were randomly sampled in each map. The fishing effort value of sampled pixels were compared between years using a two-tailed paired t-test. This t-test was permuted (50,000 times) and mean p-values recorded and t-values graphed in a histogram. The number of pixels sampled was decided by running the simulation with different sample numbers i.e. 100, 500, 1000, 5000 and 10,000. The range of t-values differed between simulations when using lower sample number (100 - 1000) but not with higher sample numbers. In order to restrict excessively long running times of the model 5000 was deemed an appropriate sample number. Jackson and Somers (1989) recommended that a minimum of 10,000 randomizations are undertaken with up to 100,000 in critical cases for biological studies. In this case, 50,000 randomisations was chosen as this was the highest number which did not greatly affect the processing time of the simulation. All R code is provided in the appendix.

If 47,500 t-tests (95% of all permutations) were significant, then it was deemed that the compared maps differed. The number of statistically significant t-tests, mean t-values and p-values from the permuted t-tests was calculated. All fishing effort maps were tested against each other using this method.

In years where significant differences were found, the two fishing effort maps were subtracted providing an absolute thematic change map of pixels with differing values using the same methodology described in section 2.2.7.2 (Remmel, 2009).

2.3. Results

2.3.1. Temporal changes in fishing effort

The number of permits issued, active vessel number, total pots worked in the district, median pots deployed per month and total pots worked varied among years (Table 2.5).

Table 2.5. Annual Northumberland potting fleet statistics (2001 – 2014). Not available (NA). Estimates of total pots worked year $^{-1}$ used in the linear model for 2006 – 2009 are shown with the original data or the < 10m vessels for these years shown in brackets.

Year	**Permits issued**	**Active Vessels (% active)**	**Active Vessels < 10m (%)**	**Median pots month $^{-1}$**	**Max pots worked month $^{-1}$ ***	**Total pots worked year $^{-1}$**
2001	155	108 (70%)	52	250	32,624	257,450
2002	151	111 (74%)	54	250	33,087	250,030
2003	153	117 (76%)	57	250	31,121	242,391
2004	136	97 (71%)	60	270	28,620	233,642
2005	130	97 (75%)	60	300	31,433	246,085
2006	120	61 (51%)	NA	300	17,770	179,365 (156,925)
2007	NA	55	NA	300	24,140	179,538 (156,802)
2008	NA	61	NA	335	26,806	194,651 (170,299)
2009	NA	60	NA	360	29,326	221,687 (193,952)
2010	121	52 (43%)	82	400	24,341	186,740
2011	107	87 (81%)	86	430	43,252	345,086
2012	114	81 (71%)	85	450	42,666	332,471
2013	118	89 (84%)	84	400	39,934	354,193
2014	119	92 (77%)	80	500	41,044	388,575

*** Total pots worked per month: sum of the maximum number of pots deployed per month by each vessel**

Permit numbers (i.e. the number of fishing vessels which have a licence to fish in the NIFCA district) decreased steadily from 2001 – 2011 (155 to 107 permits) followed by a small increase after which they remained constant (2012 – 2014, 114 -119 permits)

(Table 2.5). However, not all fishing vessels with permits fished during the year. Active vessels largely follow the decrease and increase in permit number, with the proportion of active fishing vessels in the NIFCA district remaining relatively constant between ca. 70 – 80%, except for a particularly low number of active vessels in 2010 (only 43% of vessels with permits were active) (Table 2.5). The proportion of the active <10m fishing vessels consistently increased, peaking in 2011 (86%) followed by a slight decrease from 2012 to 2014 (Table 2.5).

The median number of pots deployed per vessel doubled from 2001 to 2014 (Table 2.5). The total number of pots fished per year in the district generally increased over time from 2001 to 2014 (Table 2.5 and Fig 2.9). Total number of pots fished per year between 2001 and 2006 declined, with a small increase between 2007 and 2010 followed by a much larger increase from 2010 to 2014 (Table 2.5).

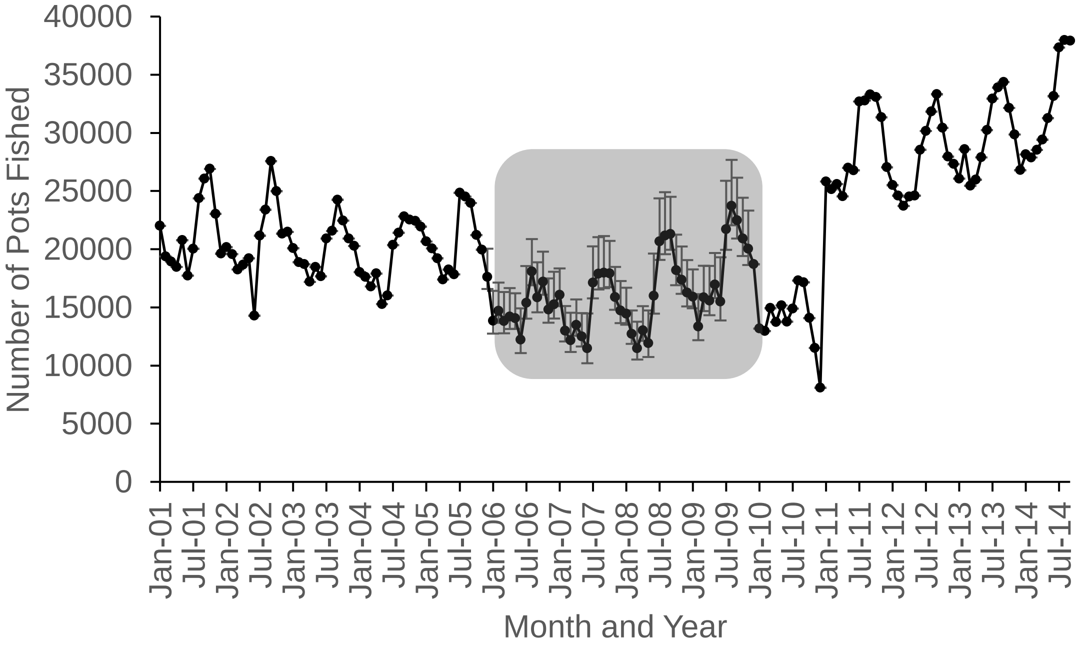


Fig 2.9. Monthly number of pots fished in the NIFCA district (2001 – 2014). Upper and lower estimates of the number of pots fished year $^{-1}$ for vessels >10m (2006 – 2009) are shown in the grey box.

Fishing effort per month declined between 2001 and 2007, increased between 2008 and 2010, and increased substantially from 2010 to 2014 (Fig 2.9). A negative binomial model provided the best fit to the data as there was evidence of overdispersion in number of pots fished per month (Fig 2.10). Since total numbers of pots deployed between 2006 and 2009 were estimated by adding the average proportion of pots fished per year of 10-12m vessels for years 2004 – 2006 and 2010

– 2014 (14.3 ± 5.8%) to the available < 10m vessel data, the linear model was run using lower and higher estimates of number of pots fished by > 10m vessels (8.5 and 20.1% respectively). Results using the lower and higher estimates did not change the overall trend with all variables significant (results available in appendix 1).The number of active fishing vessels accounted for a large proportion of the observed variance in the number of pots fished (including seasonality) (active vessels fishing per month, z-value = 18.3, $p < 0.0001$). The number of active vessels fishing accounted for a monthly increase of 1.1 pots fished per additional active vessel. Time (month) was also highly significant (months, z-value = 17.4, $p < 0.0001$) with a predicted increase of 1.01 pots fished per month. Cumulative increases and decreases are shown by the line of best fit modelled from the regression coefficients obtained from the model in (Table 2.6).

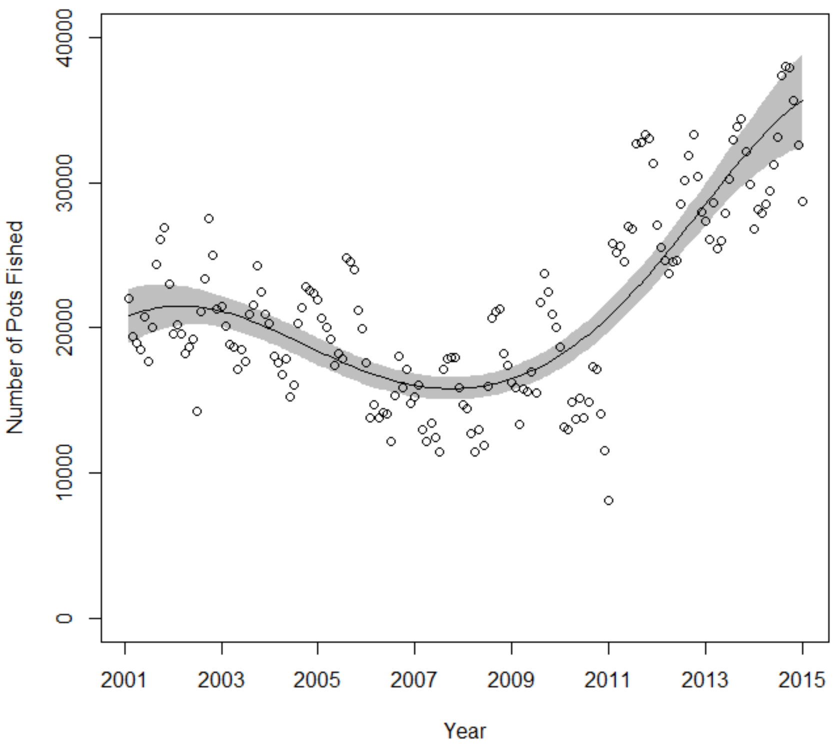


Fig 2.10. Monthly number of pots fished in the NIFCA district (2001 – 2014) with a line of best fit (black line and 95% confidence interval (grey polygon) modelled from the regression coefficients obtained in the negative binomial regression analysis.

Table 2.6. Regression coefficients, standard errors, t - value, and p-values significance for each variable using a negative binomial regression model.

	Estimate	**Std Error**	**z - value**	**p- value**
Intercept	8.6	0.07	126.6	< 0.0001
Months	0.005	0.0005	17.4	< 0.0001
Active vessels	0.01	0.002	18.3	< 0.0001

Crab and lobster landings data were available as part of the NIFCA fishery data. However, CPUE was not calculated here due to the uncertainties associated with this metric in the Northumberland pot fishery; the size of pots (i.e. volumes of pot) and the number of fleet hauls were not known and therefore would have resulted in uncertain and potentially inaccurate estimates of effort required for the calculation of CPUE.

2.3.2. *Spatial-temporal changes in fishing distribution*

Kernel density estimations of fishing vessel sightings provided a smooth continuous surface which represent the predicted number of vessels likely to occur per km^2 per year (Fig 2.11).

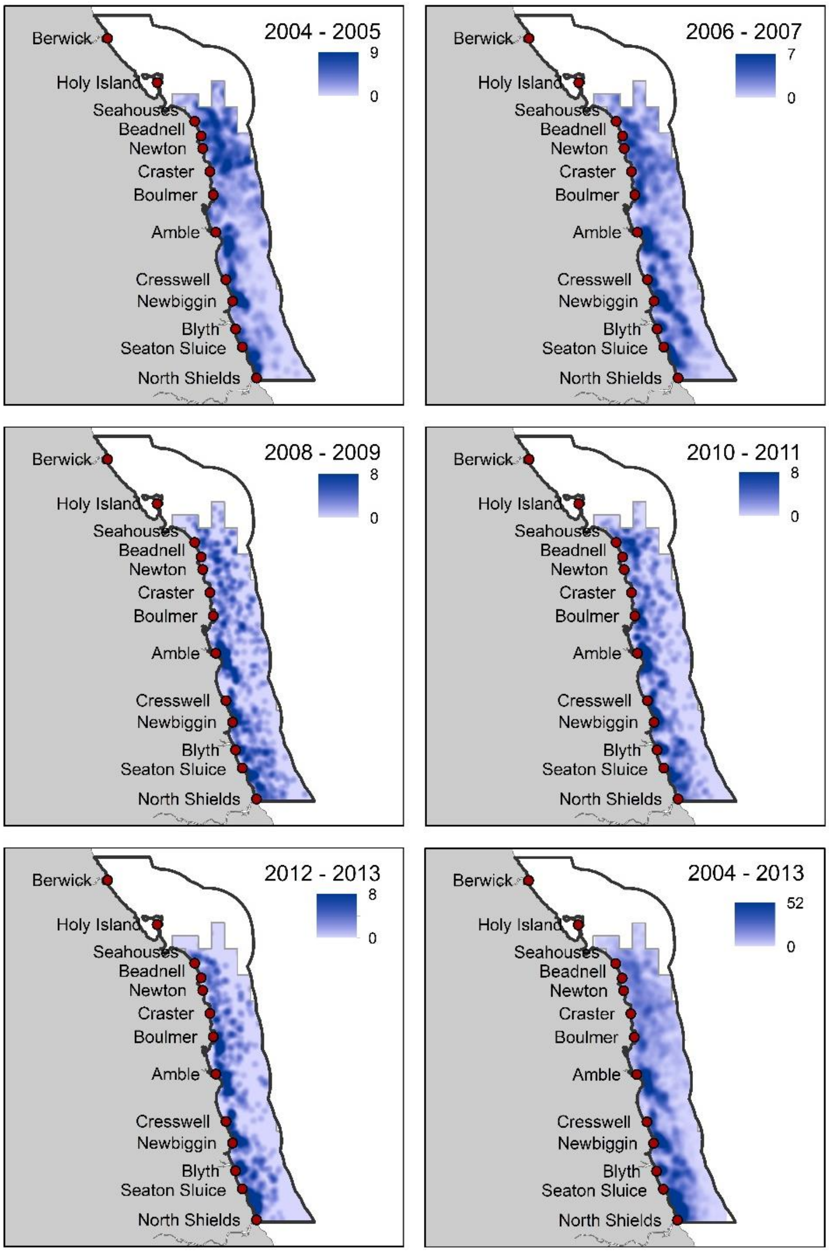


Fig 2.11. KDE of vessel sightings (vessel sightings km $^{-2}$) in areas with moderate - high confidence for years: 2004 – 2005; 2006 – 2007; 2008 – 2009; 2010 – 2011; 2012 – 2013 and 2004 - 2013.

Vessel sightings (number of vessel sightings km $^{-2}$ year $^{-1}$) had similar ranges between years (min 0, max 7 – 9) (Fig 2.11). These were largely concentrated close to shore for all years with variable sightings further off shore (Fig 2.11).

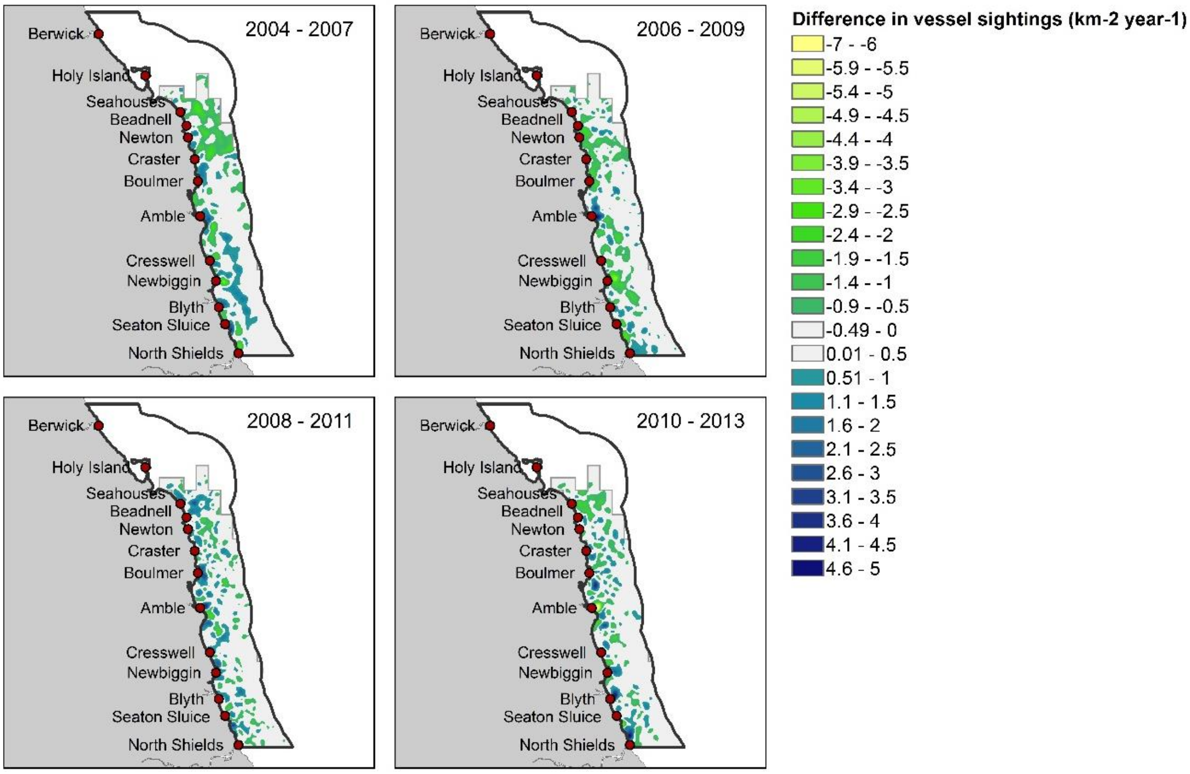


Fig 2.12. Changes in fish vessel sightings (km $^{-2}$ year $^{-1}$) in areas with moderate - high confidence between years. A) 2004 – 2005; 2006 – 2007 B) 2006 - 2007; 2008 – 2009 C) 2008 – 2009; 2010 – 2011 D) 2010 – 2011; 2012 – 2013.

The majority of differences in vessel sightings between years were small (approximately 95% of vessel sightings ranged from -1 to 1 sightings km $^{-2}$ year $^{-1}$) (Fig 2.12). Overall there was decline in sightings in the north of the NIFCA district, especially in the 2004 – 2009 period. The most variable changes both positive and negative were observed close to shore (Fig 2.12). For larger areas, predominantly further off shore, differences were negligible: vessel sightings were similar between years, varying between -0.5 and 0.5 sightings km $^{-2}$ year $^{-1}$ (shown in grey, Fig 2.12).

Cumulative changes in fishing vessel sightings (2004 – 2013) showed a clear trend (Fig 2.13). Over the study period, vessel sightings changed little over the vast majority of areas (716km^2, 80.5% of the area ranged from -0.25 to 0.25 sightings km $^{-2}$ year $^{-1}$ shown in grey, Fig 2.13). Inshore vessel sightings increased in the Southern (Cullercoats – Blyth) and middle sections of the NIFCA district (Amble – Craster). There were also notable decreases in vessel sightings, close to shore between Newbiggin and Amble and a large area in the north east of the district (from Craster to Seahouses). Although decreases (< -0.25km $^{-2}$ sightings year $^{-1}$) in fishing vessel sightings occurred over a larger area compared to increases (> 0.25 sightings km $^{-2}$

year $^{-1}$) (134km^2 compared to 39km^2), these were smaller (decreases ranged from - 0.25 to - 0.87 whereas increases ranged from 0.25 – 1.8 sightings km $^{-2}$ year $^{-1}$) (Fig 2.13).

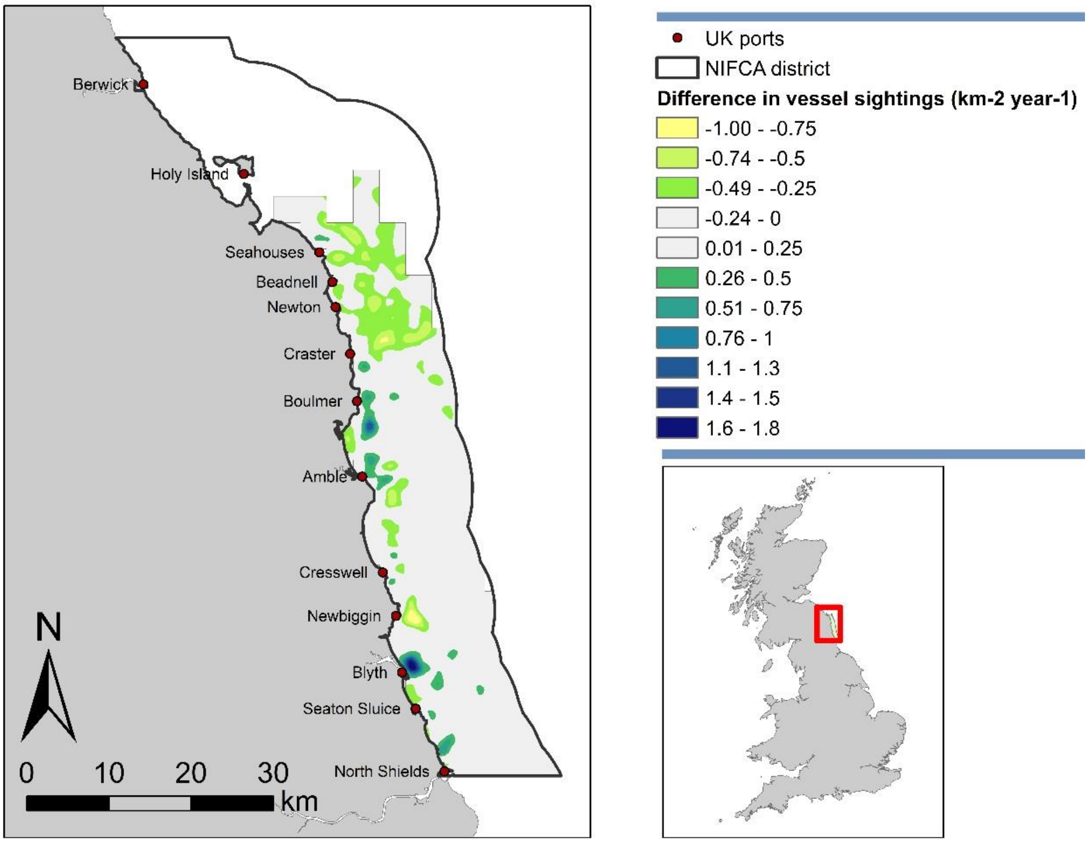


Fig 2.13. Changes in KDE of fishing vessel sightings (km $^{-2}$ year $^{-1}$) in areas with moderate - high confidence between 2004 – 2013.

2.3.3. *Spatial-temporal changes in fishing effort*

Fishing effort distribution significantly differed among all two-year periods (Table 2.7). Fishing effort decreased between 2004 - 2005 and 2006 - 2007 over large areas (Fig 2.14, positive t-value in Table 2.7). Fishing effort increased in many areas between 2006 – 2013 (Fig 2.14, negative t-value Table 2.7), particularly close to the shore. Differences in distribution between fishing effort maps also reflect these results (Fig 2.15, Fig 2.16) but highlight the variability between years with areas increasing in fishing effort in some years and decreasing in others (Fig 2.15).

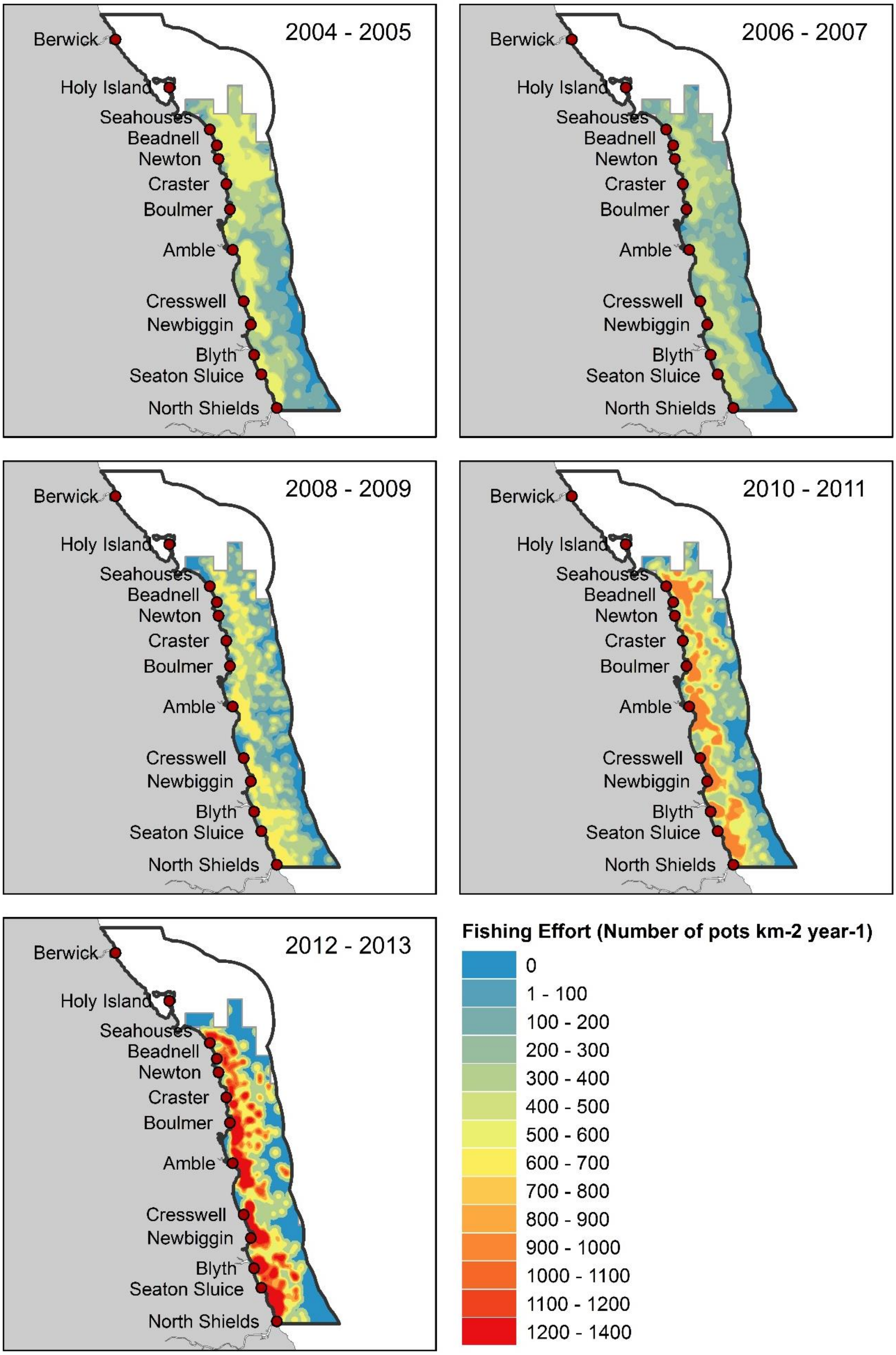


Fig 2.14. Distribution of fishing effort (number of pots km $^{-2}$ year $^{-1}$) in areas with moderate - high confidence for years: 2004 – 2005; 2006 – 2007; 2008 – 2009; 2010 – 2011; 2012 – 2013.

Table 2.7. Mean t-value and p-value of Monte Carlo simulated bi-annual comparison of fishing effort distribution maps (2004 – 2005; 2006 – 2007; 2008 – 2009; 2010 – 2011; 2012 – 2013).

Years compared	Number of significant t-tests	Mean t-value	Mean p - value
2004 - 2005 vs 2006 -2007	50,000	36.1	< 0.0001
2006 - 2007 vs 2008 -2009	50,000	-35.2	< 0.0001
2008 - 2009 vs 2010 -2011	50,000	-39.4	< 0.0001
2010 - 2011 vs 2012 -2013	50,000	-19.2	< 0.0001

Once again, a clearer trend emerged in absolute changes of thematic maps when examining changes occurring over the decade. Cumulative yearly changes (2004 – 2013) showed areas further from shore had either stable or decreasing fishing effort with a large majority of inshore areas increasing in fishing effort (Fig 2.16). The maximum increase in fishing effort is much larger than the maximum decrease in fishing effort: increases of up to a 1150 pots km $^{-2}$ year $^{-1}$ compared to decreases of 450 pots km $^{-2}$ year $^{-1}$ (Fig 2.16).

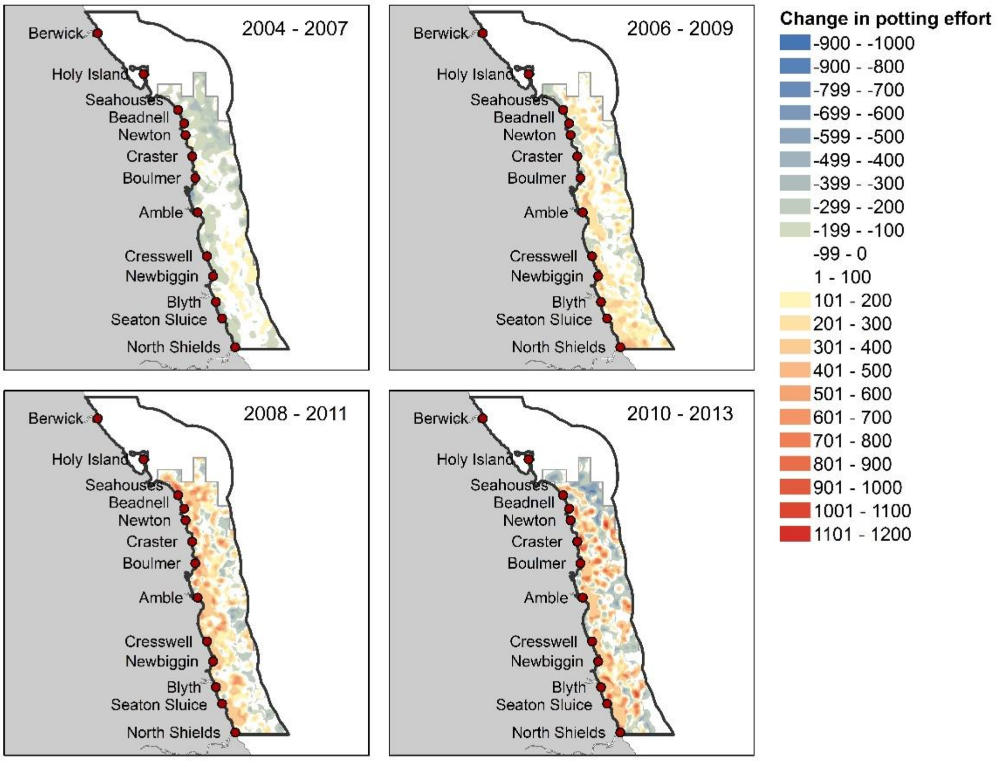


Fig 2.15. Changes in distribution of fishing pressure (number of pots km^{-2} year^{-1}) in areas with moderate - high confidence between years. A) 2004 – 2005; 2006 – 2007 B) 2006 - 2007; 2008 – 2009 C) 2008 – 2009; 2010 – 2011 D) 2010 – 2011; 2012 – 2013.

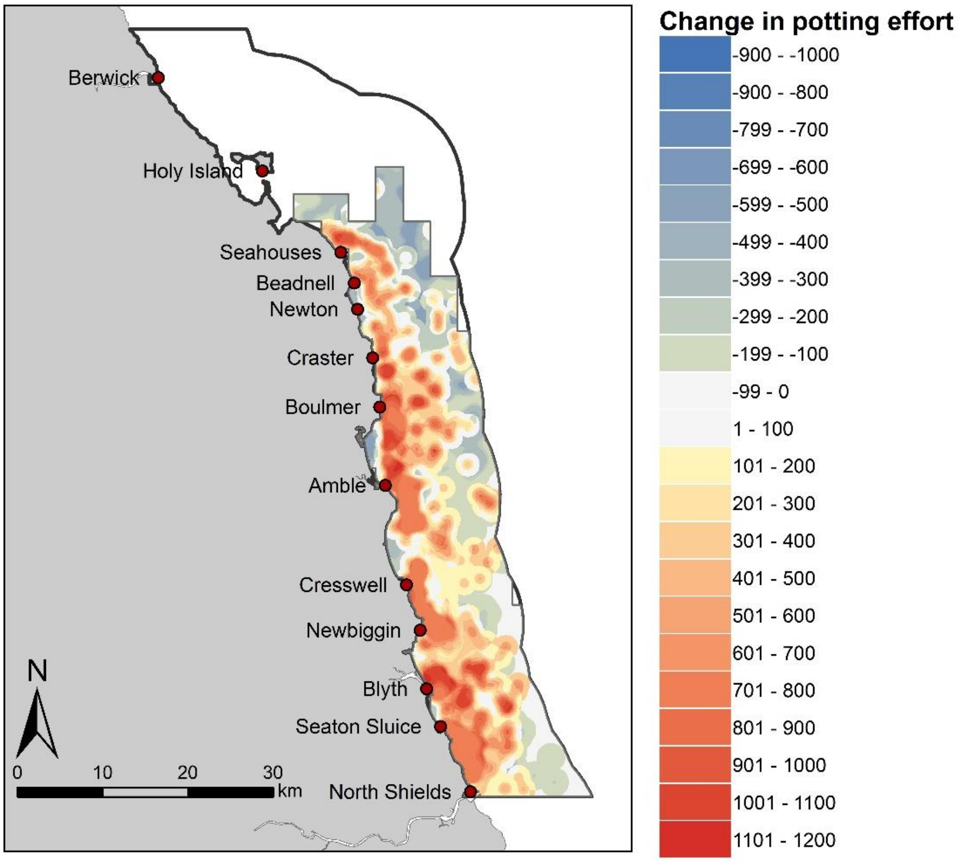


Fig 2.16. Changes in distribution of fishing effort (number of pots km $^{-2}$ year $^{-1}$) in areas with moderate - high confidence between 2004 – 2013.

2.4. Discussion

Similarly to previous research (Breen et al., 2014, Turner et al., 2015), our results show that existing datasets collected through routine fisheries patrols can be used to accurately map shellfishing effort, highlighting viable alternatives to VMS monitoring for smaller vessel inshore fisheries. Mapping of the pot fishery and other static gear fisheries, are likely to be accurate because the recorded sightings are for locations where deploying or hauling of gear occurs. This analysis built on previous work but in addition now evaluates temporal and spatial changes in a local fishery at very high resolution for the first time. These temporal and spatial changes in fishing effort in the Northumberland pot fishery between 2004 and 2013 are discussed here in reference to marine spatial management.

2.4.1. Temporal changes in fishing effort

Fishing effort (number of pots at sea per month) increased in Northumberland between 2001 and 2014 (Fig 2.10). Changes in fleet composition or fishers' behaviour may explain increases in effort, although available information is largely anecdotal and further work is recommended. The proportion of vessels < 10m in the NIFCA district has steadily increased over time (Table 2.5). These smaller vessels are not subject to as much legislation as vessels > 10m and have cheaper fishing vessel licenses. Furthermore, between 2001 – 2014 there was increased uptake by local fishers of improved fishing technology, including GPS and echosounder, better vessel layouts and more efficient and powerful engines, allowing them to fish a greater number of pots per month and target more specific areas or habitats (NIFCA, pers. comm.). This increase in efficiency may explain the doubling in median pots deployed per vessel between 2001 and 2014 (Table 2.5) which in turn is contributing to the large yearly increases in pots fished over the same period (Fig 2.10).

Another possible explanation for this increase in effort is that in the face of low abundance of target species, fishers increased effort in order to maintain levels of catch or moved focal area. However, landings per unit effort for both lobster (Telsnig, 2013) and crab also increased between 2001 and 2014 (NIFCA, pers. comm.). It seems that the abundances of target species were sufficient to allow a sustained increase in effort over the 10 year period although it is unclear whether overall stock abundances increased or decreased in Northumberland.

A clear increase in fishing effort (number of pots at sea per month) has occurred in the district between 2001 and 2014. This is partially explained by changes in fleet composition and behaviour, although drivers of change remain largely anecdotal. Further stakeholder engagement is recommended to further investigate these drivers.

2.4.2. Temporal changes in potting vessel distribution

Variability in locations of fishing vessel sightings between bi-annual maps were reduced when grouped across all years, revealing that distribution of vessels changed relatively little across the study period (Fig 2.13). Variability in fishing vessel locations may be due to true differences in fishers' distribution between years or an incomplete representation of vessel sightings (see section 2.4.5 for further details on assumptions and potential sources of error of mapping techniques). However, over a decade, potting distribution in the NIFCA district remained relatively constant with areas close to shore used consistently through time but with more variable use in areas further offshore. In addition, there was a slight increase in concentration over time of fishing vessel sightings in areas close to shore (Fig 2.13). The increased sightings close to shore may tentatively be related to the increase in small boats within the fleet and increasing fuel prices although further evidence is required to state this with confidence. In a Greek fishery, small < 9m vessels' average travel time to fishing locations was much lower than that of fishing vessels > 15m (Tzanatos *et al.*, 2006); smaller vessels were more likely to fish closer to shore than larger vessels (FAO, 2005). In addition, volatile fuel prices affected fishing behaviour in the UK with fishing occurring closer to port and reduced exploratory fishing (Abernethy *et al.*, 2010).

2.4.3. Temporal changes in potting effort distribution

Changes in fishing effort distribution between years (section 2.3.3) were found across large sections of the study area, highlighting the high inter-annual variability of fishing effort over time at a regional scale (Lynch, 2014). This further highlights the usefulness of monitoring fisheries over long-time periods as 'snap shots' of the fishery may lead to under or over estimates of fishing effort (Lynch, 2014). Increases were concentrated close to the shore, particularly around the larger ports of the district. Decreases in areas, or areas that showed little or no change were in the offshore sections of the district. Increases in fishing effort inshore are likely to be heavily determined by the steady increase in overall pots fished from 2008 – 2014 (section 2.3.1) in combination with the slight concentration of fishing vessels sightings inshore (section 2.3.2). The trend of increasing fishing effort close to shore is consistent with the change in fleet composition (increases in <10m fishing vessels) and fishing behaviour (increased in median pots fished per vessel) discussed in previous sections.

The interannual variability of the Northumberland shellfishery highlights the importance of long-term monitoring. The increase in fishing pressure across the district but particularly inshore over the course of a decade means that further monitoring will inform conservation and fisheries management. This research reinforces the usefulness of combining effort data with fishing spatial distribution. Cumulative changes in fishing effort distribution provided greater detail on temporal trends than either analysis of fishing effort data (section 2.3.1) or vessel distributions independently (section 2.3.2), for example demonstrating that fishing effort did not increase uniformly across the district but has become highly concentrated inshore, especially from 2010 – 2013.

2.4.4. Temporal fishing effort distribution for management of the marine environment

Commercial fishing activities are often reported at very large scales (e.g. ICES rectangles, approximately 30 x 60Nm) (Brehme *et al*., 2015). These data rarely accurately reflect the heterogeneity of ocean activities (Brehme *et al.*, 2015) and only allow the broadest of fishery-habitat interactions to be examined. Finer resolution of fishing activity has been repeatedly highlighted as priority for future research (Breen *et al.*, 2014; Campbell *et al.*, 2014; Brehme *et al.*, 2015) as a prerequisite for the assessment and management of fisheries impacts on the seabed and of their interactions with other industries or MPAs (Crowder and Norse, 2008; Stewart *et al.*, 2010), and to help reduce conflict between competing marine sectors (Katsanevakis *et al.*, 2011). However, the peripatetic nature of fisheries has been a significant obstacle to mapping their usage of the marine environment (Stewart *et al.*, 2010), although some of these difficulties are being addressed through the successful use of VMS for > 12m fishing vessels and through modelling of activities using sightings at sea for smaller inshore fisheries (Breen *et al.*, 2014; Vanstaen and Breen, 2014; Turner *et al.*, 2015). The work presented here provided predictions of potting activities at a resolution of 1km^2, a much finer resolution than for other similar fishing activity mapping research (6km^2 – 50km^2; Breen *et al.*, 2014; Brehme *et al.,* 2015). Without maps of fine-scale spatial fishing effort, important localised changes and habitat sensitivities may be overlooked. The present study demonstrated that fishing effort did not increase uniformly across the district over time but became highly concentrated inshore, especially from 2010 – 2013. From a socio-economic perspective, potting may therefore be more vulnerable to changes in legislation (e.g. area closures) in the busy inshore marine environment where competing demands exist between larger numbers of users and where conflicts between different stakeholder groups occur (Dalton *et al.*, 2010). The more accurately activities can be mapped in these areas, the greater the ability of policy makers to reduce conflict and develop successful marine spatial plans (Dalton et al., 2010).

Although fishing distribution maps are increasingly proposed for marine spatial management, temporal distribution of fishing effort has often been neglected (Brehme *et al.*, 2015). This study highlights the high inter-annual variability of fishing effort over time at a regional scale and usefulness of monitoring fisheries over long time-periods (Lynch, 2014). The lobster fishery in Maine changed over time depending on fishers' responses to market forces (Steneck *et al.*, 2011), informal rules amongst fishers (Acheson and Brewer, 2003; Brewer, 2010), lobster population responses to changes in oceanographic conditions (Steneck and Wilson, 2001; Incze *et al.*, 2006; Holland, 2011; Zhang *et al.*, 2011) and to harvesting practices within the fishery (Acheson, 1988; Acheson and Brewer, 2003; Brewer, 2010). Fishers in Northumberland are likely to respond to similar drivers and changes in fishing behaviour (Turner *et al.,* 2015) and therefore for accurate management decisions, temporal patterns must also be understood.

The Northumberland shellfishery has the highest potting vessel sightings per unit effort of the UK (Vanstaen and Breen, 2014). The available data would make it well suited as a case study on the effects of area closures and other types of management. However, in order to fully understand the implications of fisheries spatial management, fisheries – habitat interactions must be understood (chapter 3).

2.4.5. Critical appraisal of data and models used

The accuracy and validity of the data underpinning the model will determine the accuracy of the outputs. Sources of error may be introduced by assumptions made, as discussed in the following section. These can be visualised in the flow chart of the modelled process in Fig 2.17. Red arrows indicate where potential errors in data or assumptions may negatively affect model outputs (Fig 2.17).

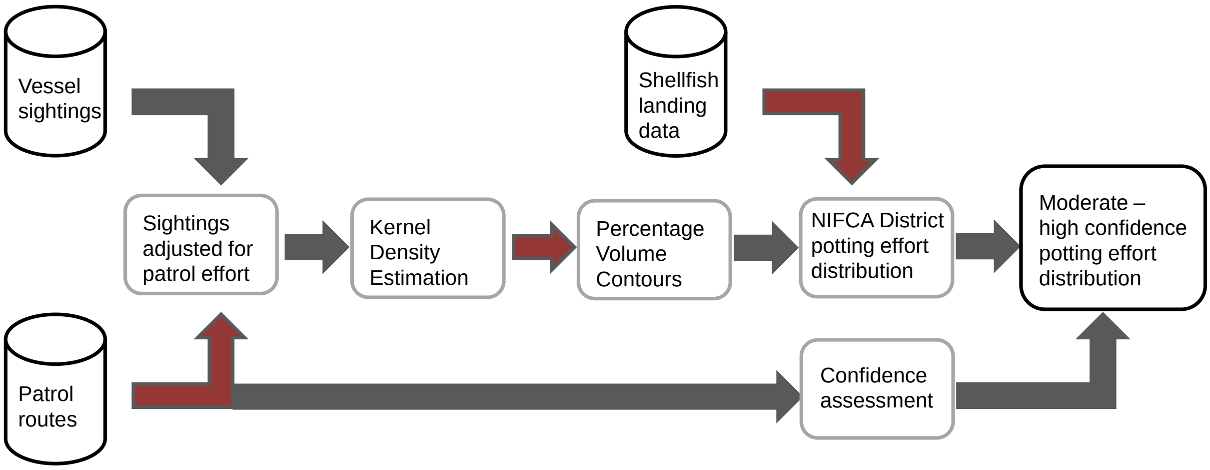


Fig 2.17. Diagram of GIS processes undertaken to spatially map distribution of potting effort densities in Northumberland IFCA district. Raw data (black cylinders), GIS mapping procedures (grey boxes) and final potting effort density distribution map (black box) are shown. Red arrows show indicating sections where errors in data or assumptions may negatively affect model outputs.

Elements of the raw data used in this research were collected opportunistically. For example, vessel sightings at sea were collected during routine enforcement patrols. Similarly, the number of patrols conducted each year were driven by NIFCA requirements. Caveats in this data included missing GPS coordinate data for some patrol routes and incomplete sightings for all active fishing vessels in the district. A proportion of patrol routes were lost for all years (Table 2.1). Reasons for this are unknown.

In addition, not all active fishing vessels were sighted during routine patrols - the number of active fishing vessels in the district (Table 2.5) is always greater than the number of different fishing vessels sighted at sea (Table 2.1). This may be due to the timing of the NIFCA patrols which generally occurred between 8 am and 4 pm (dependant on tides), whereas potting routinely occurs in the early hours of the morning (leaving port at 2am returning for 12am). Thus, a certain amount of fishing activity will not be recorded during routine patrols. However, other studies have used similar sightings from enforcement patrols to successfully map fishing activity (Woolmer, 2009; Breen *et al.*, 2014; Vanstaen and Breen, 2014; Turner *et al.*, 2015).

In addition, the use of weighting and standardisation of vessel sightings ensured the model was robust to differences in patrol effort between years; patrol effort data were standardised between years (section 2.2.2), weighting of individual vessel sightings dependant on likelihood of observation (section 2.2.3) and a confidence layer of the district based on patrol routes was created (section 2.2.3).

Monthly shellfish landings data for 2001 – 2014 included vessel home port, landing port, number of pots worked per month, number of landings and weight of landings for all permit holders even if no fishing was undertaken during the month. However, as with other data collected from stakeholder participatory studies, the accuracy of this data relies on the truthfulness or accuracy of the fishers' submitting these permit return forms. In this case the author has no reason to doubt the accuracy of the collected data and there is no obvious motivation for fishers to lie (Brehme *et al.*, 2015).

Predicting the number of vessels that are likely to be sighted in a given area per year through interpolation between actual vessel sightings (creation of KDEs, Fig 2.8) will always result in uncertainties. However, the bi-annual grouping of sightings was felt to provide an adequate density of vessel sightings for accurate predictions. In addition, methods used for bandwidth selection of the KDEs, a critical assumption for the prediction of values between actual vessel sightings, followed recommendations for the best available methodology, proving to be robust in several ecological studies (Bertrand *et al.*, 1996; Kie and Boroski, 1996; Kie *et al.*, 2002; Kie *et al.*, 2010).

Many methods for map comparison exist and the choice of appropriate methodology will be determined by the nature of the maps (i.e. the type of data used to create these) and the aims of the research (Foody, 2007). As the approach used will determine which differences or similarities are observed and what may be missed by the analysis, awareness of the pros and cons of map comparison methods is important (Foody, 2007). Qualitative comparison of spatial maps - presenting maps side by side and stating that these look similar or dissimilar - is common because quantification of similarity between two spatial patterns can be difficult (Rose et al., 2009). However, pixel based map comparisons have recently grown in application with methods such as the Kappa statistic, which compare the partial distribution of variables in terms of their presence or absence, or other categorical classifications (Hagen-Zanker 2006; Ban et al., 2009). However, there are fewer established methods for statistically comparing the distribution of variables with numerical values, such as the intensity of fishing activity (Hagen-Zanker 2006). Until recently, spatial comparison of maps has been undertaken using ‘cell by cell’ methods (Hagen-Zanker, 2006) or through quantification of map similarity whilst accounting for spatial structure (Hargrove *et al.*, 2006), including taking into account spatial autocorrelation (Hagen-Zanker, 2009). However, these map comparison methods, many of which were primarily developed for accuracy assessment of simulated maps, give an indication of similarity but not significance of change. Furthermore, for optimal results, the maps used in the analyses must have the same classes, in terms of number and meaning (Haack and Rafter, 2006). Northumberland inshore fishing effort is highly variable, spatially and temporally; inshore fishing practices have to adapt quickly to many socio-economic factors, for example, fluctuation in fuel costs, catch levels and weather (Abernethy *et al.*, 2007; Daw, 2008). This resulted in different map classes between years. Even when standardised across maps using a fuzzy approach (a potential source of error (Fritz and See, 2005)), high dissimilarity was found between years, providing little information that could not already be described by visually examining fishing effort maps. In this research issues of non-normal distribution of mapped data, spatial autocorrelation and statistical power related to many map comparison analysis (Hagen-Zanker, 2006; Hagen-Zanker, 2009) were overcome through the use of random point selection and large number of permutations in the Monte Carlo analysis for the detection of change between fishing effort distribution maps.

2.5. Concluding remarks

Fishing effort has increased significantly across the majority of areas in Northumberland with particularly large increases inshore, attributable tentatively to changes in local vessels, but evidence remains largely anecdotal and further work to investigate social drivers behind changes in effort is recommended. Results from this work are applicable for management locally, and can be replicated using routinely collected fisheries enforcement data for other parts of the UK. Distribution of fishing effort is integral for management of fisheries by providing information on areas that may be especially susceptible to fishing impacts, as well as informing selection of areas best suited for protection, by accommodating both conservation and socio-economic goals.

Chapter 3 . Decadal Inshore Fishing Activities in Northumberland: Spatial, Temporal Changes of Habitat Usage

3.1. Introduction

Seabed habitats and their associated communities are predominately determined by their ambient abiotic conditions (Connor *et al.*, 2004) (i.e. substratum, depth, exposure to wave action/tidal currents, salinity, topography, geology, oxygenation and scour/turbidity). In addition to natural environmental conditions, the physical interaction of fishing gears on the seafloor may exert further pressure on benthic communities (Nielsen *et al.*, 2013). Fishing is not uniformly distributed across regions (Jennings *et al.*, 1999; Jennings *et al.*, 2012) and the footprint of gear on the seafloor will vary depending on the fishing method (Vanstaen and Silva, 2010; Jennings *et al.*, 2012). Sensitivity of habitats to fishing impacts will be determined by the intensity, frequency and extent of natural disturbance to which the habitat and its associated species are subject to (Kaiser, 2014). For example, in high-energy environments fishing disturbances may have minimal effects compared to natural disturbance (Kulbicki *et al.*, 2007; Sciberras *et al.*, 2013). However, there are many examples where fishing disturbances have caused long-lasting changes in the marine environment (see Kaiser *et al.* (2006) for meta-analysis of fishing impacts).

Mobile-gear fishing activities can impact biodiversity or alter habitat either directly (through physical disturbance) or indirectly (through changes in food web dynamics) (Kaiser *et al.*, 2006; Armstrong and Falk-Petersen, 2008; Babcock *et al.*, 2010), but little is known about spatial distribution and intensity of static-gear fishing activities affecting the seafloor (Breen *et al.*, 2014). Assessments of full impacts of a fishery require basic data on the conservation status (i.e. health quality and status) of individual habitats and species, as well as data for fine-scale distributions of ongoing fishing activities (Pedersen *et al.*, 2009; Eno *et al.*, 2013).

To date, research world-wide has predominately focused on mapping mobile gear fishing activity using VMS and overlaying this information on existing broadscale habitat datasets (Nilsson and Ziegler, 2007; Stelzenmüller *et al.*, 2008). However, a primary limitation of VMS data is that it is only recorded for large vessels (> 12m) and as such, inshore fishing fleets which are largely composed of small vessels (< 12m) are not well represented by VMS (Breen *et al.*, 2014). Research has provided a broadscale snapshot (5 x 5km) of mobile gear fishing habitat use for a single year (Nilsson and Ziegler, 2007; Lambert *et al.*, 2011) thus not accounting for change over time in fisheries uses of habitats. Fishing spatial patterns have been shown to vary over time (chapter 2) (Kaiser *et al.*, 2002; Nilsson and Ziegler, 2007), the availability of target species, gear and fishers' territoriality being important drivers (Acheson, 1975; Rijnsdorp *et al.*, 2001; Turner *et al.*, 2012). The inter-annual variability of fishing means short-term studies are inadequate to fully inform appropriate management (Lynch, 2014), yet the spatio-temporal variability of fishing effort must be understood. Some temporal trends are known for large vessels (> 18m) over broadscale marine landscapes (Connor *et al.*, 2006) at 2 x 2Nm resolution (Stelzenmüller *et al.*, 2008), but at a finer scale, significant gaps remain in habitat use by smaller vessels (Caveen *et al.*, 2014). The need to address this information deficit has become increasingly relevant due to implementation of new management measures in European Marine Sites (EMSs) and Marine Conservation Zones (MCZs) in England, with many of these areas located inshore where a large portion of the fishery uses smaller vessels (< 12m). However, it is debatable whether these management measures are currently being underpinned by robust science while lacking detailed information on fisheries-habitat interactions (Caveen *et al.*, 2014).

The Northumberland fishery (Fig 2.7) is a mixed fishery (Garside *et al.*, 2003), with the majority of Northumberland fishers targeting crustacea: European lobster (*Homarus gammarus*), velvet crab (*Necora puber*) and edible crab (*Cancer pagurus*) using baited pots (or traps). These shellfish species use habitats differently; their distributions, movements and abundances are influenced by habitat type, quality and location (Galparsoro *et al.*, 2009; Geraldi *et al.*, 2009; Skerritt *et al.*, 2015). Lobster (*H. gammarus*) and velvet crab (*N. puber*) are found predominantly on shallow rocky ground although lobster can also be found at 60m or deeper (Wilson, 2008; Galparsoro *et al.*, 2009). The edible crab (*C. pagurus*) is found in all habitat types but probably prefers coarse sediment and offshore muddy sand (Neal and Wilson, 2008). Thus fishermen are likely to be selective for habitat when targeting different shellfish species. Fishing with mobile gear varies inter-annually in extent and in habitat selection (Jennings *et al.*, 2012; Diesing *et al.*, 2013), but no comparable studies of fishing with static potting gear are apparent. These may differ, for example, due to fishers' territorial behaviours (Turner *et al.*, 2012). Spatial clustering of fishers in specific habitats can indicate productive areas (i.e. high catches of target species) and over time can indicate the persistence of these favourable fishing conditions.

Information on the distribution of potentially sensitive habitats, or those most vulnerable to inshore fishing activities, are crucial so that these can be afforded an adequate degree of protection (Eno *et al.*, 2013; Kaiser, 2014). However, further information is required to better understand the ways Northumberland’s small scale (< 10m) inshore fishery uses habitat in order to support effective management. In particular, accurate fishing vessel positioning data, over adequate time scales is needed to quantify variability in fishing activity and fine-scale and accurate habitat maps are required. Building on chapter 2, where fishers’ spatial fishing pressure was investigated over a decade, this research explores temporal changes in fishers’ habitat use (2004 – 2014), providing evidence that would be of use for fisheries and conservation management. This research sought to test the hypothesis that fishers were selective when targeting habitats (hypothesis 1), specifically that fishers targeted rocky habitats over other habitats (hypothesis 2) in order to catch high value lobster (Wilson, 2008; Galparsoro *et al.*, 2009). This is achieved using routine patrol vessel sightings in combination with high resolution broadscale (EUNIS level 3) and fine scale (EUNIS level 5) habitat maps, providing a case study for the assessment of other inshore fisheries. The scale needed for management of fisheries – habitat interactions (broadscale and fine-scale habitat maps) was also explored.

3.2. Methods

3.2.1. Habitat distribution

Reliable and up-to date habitat maps were not available for the whole NIFCA district. However, the new Coquet to St Mary’s Marine Conservation Zone (hereafter CQSM MCZ) (Fig 3.1), was surveyed in 2014 using high resolution multibeam echosounder (MBES) (1m resolution) coupled with multi-gear groundtruthing to map habitats considered for conservation (see Fitzsimmons *et al.* (2015).

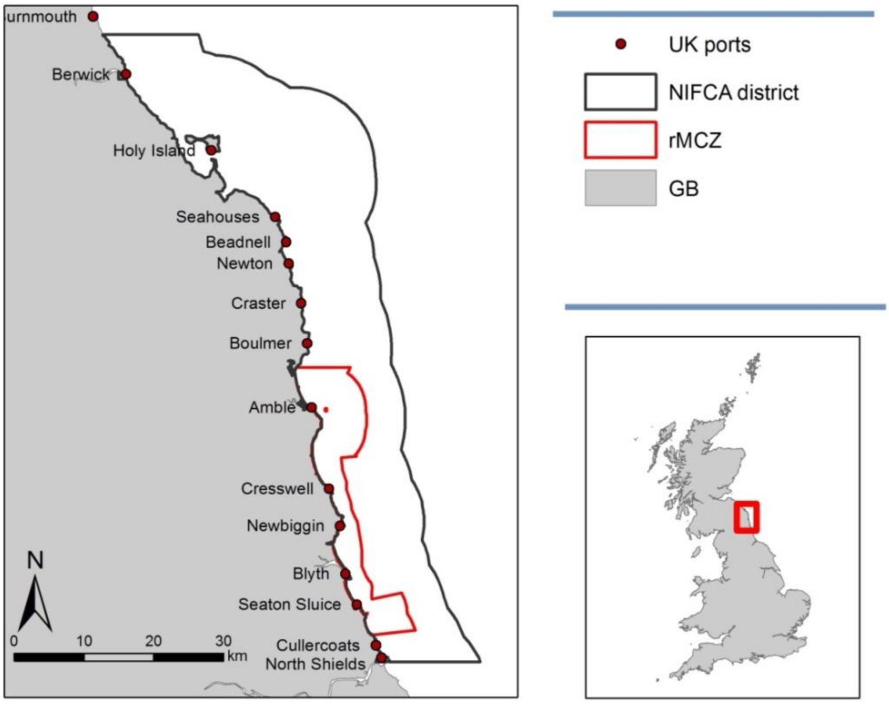


Fig 3.1. Location of the Coquet to St Mary's MCZ (red line) within the NIFCA district (black line). Ports are labelled (red dots).

EUNIS level 3 (broadscale habitats) were produced as part of the evidence base for the CQSM MCZ (Fitzsimmons *et al.,* 2015) (Fig 3.3). This level of classification had the advantage of reducing erroneous habitat classification and had a high MESH confidence score (84) (overall accuracy of 86% and a 'moderate' Kappa score of 0.41 as defined by Landis and Koch (1977)) (Lightfoot, unpublished). EUNIS level 3-6 maps were produced for areas classified as rocky substrate as part of on-going research at Newcastle University (overall accuracy of 51% and a Kappa score of 0.26 ('fair')(Lightfoot, unpublished)(Fig 3.4). Higher EUNIS levels of classification of sediment habitats were not attempted due to the lower-level of groundtruthed biological information available for these habitats and the lack of an adequate physical environmental dataset (only backscatter, bathymetry, slope and derivatives of these were available) (Galparsoro *et al.*, 2013). Very shallow inshore areas were not surveyed as the MCZ would only be designated for circalittoral features (Fig 3.3, Fig 3.4).

The UK biotope classification system describes the marine environment at different spatial scales and is split into six levels (see chapter 1 for full description) ranging from very broad spatial scales (environment, Fig 3.2) to very small spatial scales (sub-biotopes, Fig 3.2). Individual levels can be combined (in hierarchical order) to the level desired (Fig 3.2). For example, CR.MCR.EcCr.FaAlCr.Pom has 5 levels (marine biotope) and would be described as ‘Faunal and algal crusts with *Pomatoceros triqueter* and sparse *Alcyonium digitatum* on exposed to moderately wave-exposed circalittoral rock’. EUNIS classification used in the MCZ habitat maps was translated into the UK biotope classification system because these were used throughout this thesis and are directly comparable (Chapter 1).

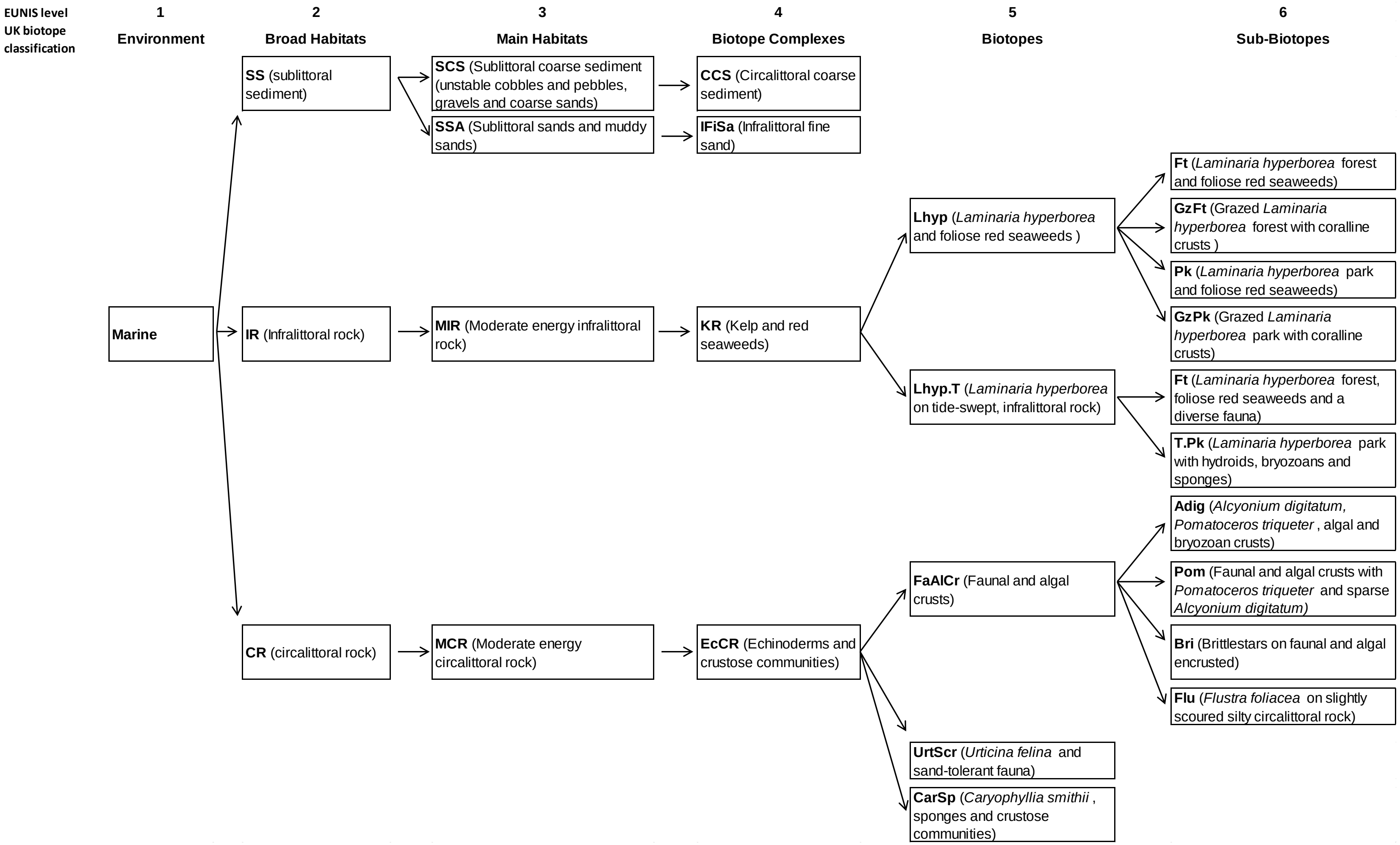


Fig 3.2. Hierarchical classification of habitats and biological assemblages commonly found in the Berwickshire North Northumberland Coast European Marine Site. Classification codes are shown in bold followed by a brief description.

Fitzsimmons *et al.* (2015) identified three broadscale habitats types (EUNIS level 3) within the CQSM MCZ (Fig 3.3). ‘Moderate energy circalittoral rock’ (CR.MCR) which occurs on exposed to moderately wave-exposed circalittoral (below the photic zone) bedrock and boulders, subject to moderately strong and weak tidal streams. This habitat type contains a broad range of biological subtypes, from echinoderms and crustose communities to *Sabellaria* reefs and mussel beds (European Environment Agency, 2005). ‘Sublittoral sand’ (SS.SSa) consists of clean medium to fine sands on open coasts, offshore or in marine inlets. These habitats are subject to some wave action and tidal currents which may restrict the silt and clay content to less than 15%. This habitat is characterised by a range of taxa including polychaetes, bivalve molluscs and amphipod crustaceans (European Environment Agency, 2005). ‘Sublittoral mud’ (SS.SMu) consists of mud and cohesive sandy mud and can be found from the extreme lower shore to offshore, circalittoral habitats. This habitat is predominantly found in stable deeper/offshore areas where the reduced influence of wave action and/or tidal streams allows fine sediments to settle. Such habitats are often dominated by polychaetes and echinoderms, in particular brittlestars such as *Amphiura* spp (European Environment Agency, 2005). Area (km^2) and proportion were calculated for each EUNIS level 3 habitat present in the CQSM MCZ in ArcGIS (Table 3.1).

Table 3.1. Area and proportion of EUNIS level 3 habitats in the CQSM MCZ.

	Area (km^2)	Proportion
CR.MCR	64.13	0.39
SS.SSa	51.76	0.32
SS.SMu	47	0.29

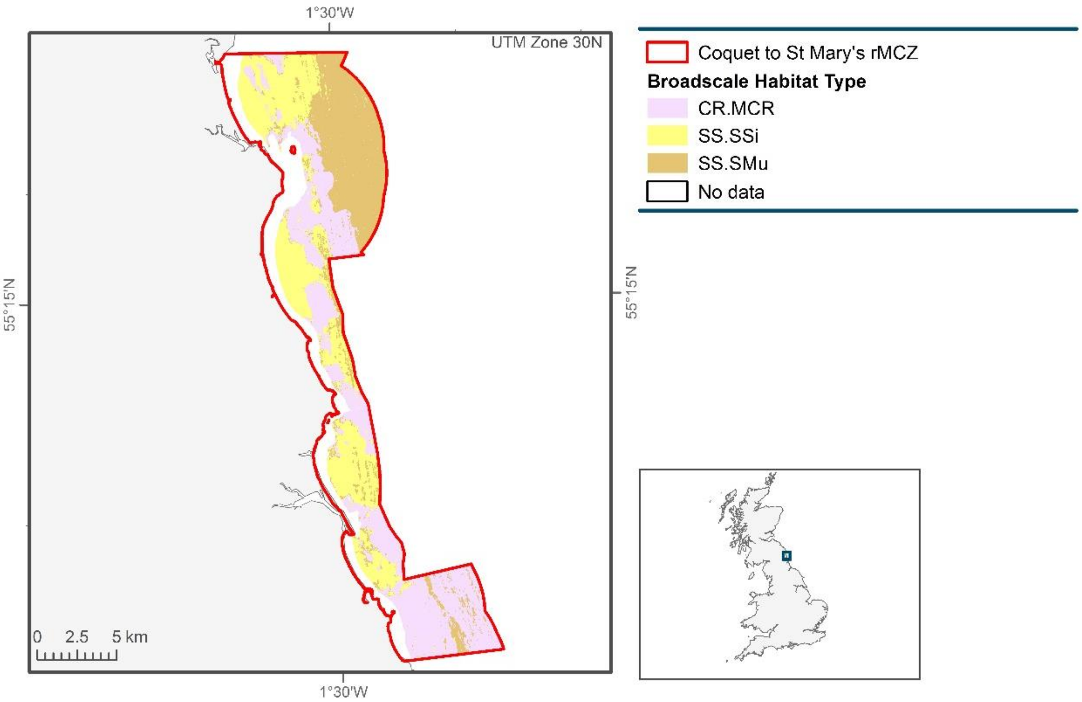


Fig 3.3. Broadscale habitat map (EUNIS Level 3) of the CQSM MCZ (modified from Fitzsimmons *et al.,* 2015).

Three broadscale habitats, 1 biotope and 3 sub-biotope classifications (EUNIS level 3 - 6) were identified within the higher classification mapping of the CQSM MCZ (Fig 3.4). Broadscale habitats were CR.MCR, SS.SSa and SS.SMu (described in previous section). More in-depth biological classifications (EUNIS level 5 - 6) included 'Faunal and algal crusts on expose to moderately wave-exposed circalittoral rock', (CR.MCR.EcCr.FaAlCr, abbreviated as FaAlCr), '*Flustra foliacea* on slightly scoured silty circalittoral rock' (CR.MCR.EcCr.FaAlCr.Flu abbreviated as FaAlCr.Flu), '*Alcyonium digitatum*, *Pomatoceros triqueter*, algal and bryozoan crusts on wave-exposed circalittoral rock' (CR.MCR.EcCr.FaAlCr.ADig, abbreviated as FaAlCr.ADig) and 'Faunal and algal crusts with *Pomatoceros triqueter* and sparse *Alcyonium digitatum* on exposed to moderately wave-exposed circalittoral rock' (CR.MCR.EcCr.FaAlCr.Pom, abbreviated as FaAlCr.Pom). Full descriptions of species assemblages are available from Connor *et al.* (2004). All rocky reef biotopes identified were sub-categories of CR.MCR and therefore share the same broad physical environmental characteristics (Fig 3.2). As would be expected, small scale heterogeneous distribution of biotopes and sub-biotopes (EUNIS level 5 - 6) occurred (Fig 3.4) (Connor *et al.*, 2004). Area (km^2) and proportion of habitats are shown in Table 3.2.

Table 3.2. Area and proportion of EUNIS level 3 - 6 habitats in the CQSM MCZ.

	Area (km^2)	Proportion
CR.MCR	9.25	0.06
SS.SSa	51.76	0.32
SS.SMu	47.00	0.29
CR.MCR.EcCr.FaAlCr	25.61	0.16
CR.MCR.EcCr.FaAlCr.ADig	17.55	0.11
CR.MCR.EcCr.FaAlCr.Bri	0.60	0.01
CR.MCR.EcCr.FaAlCr.Pom	4.12	0.03
CR.MCR.EcCr.FaAlCr.Flu	4.95	0.03

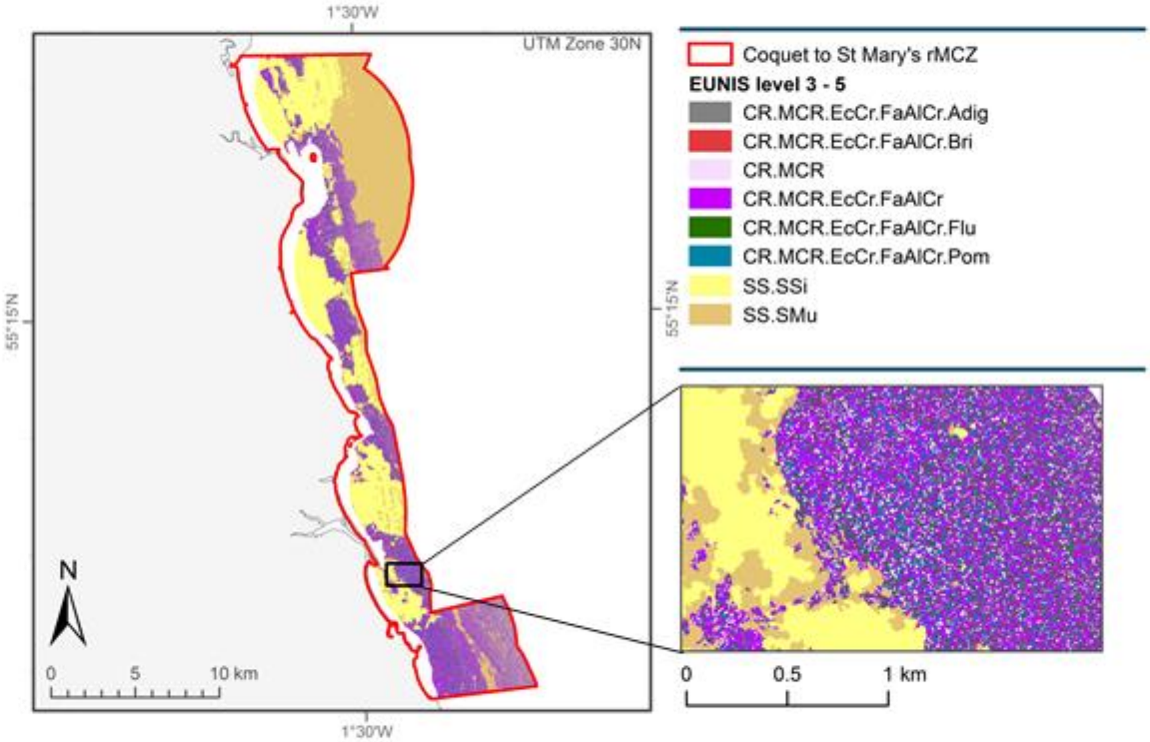
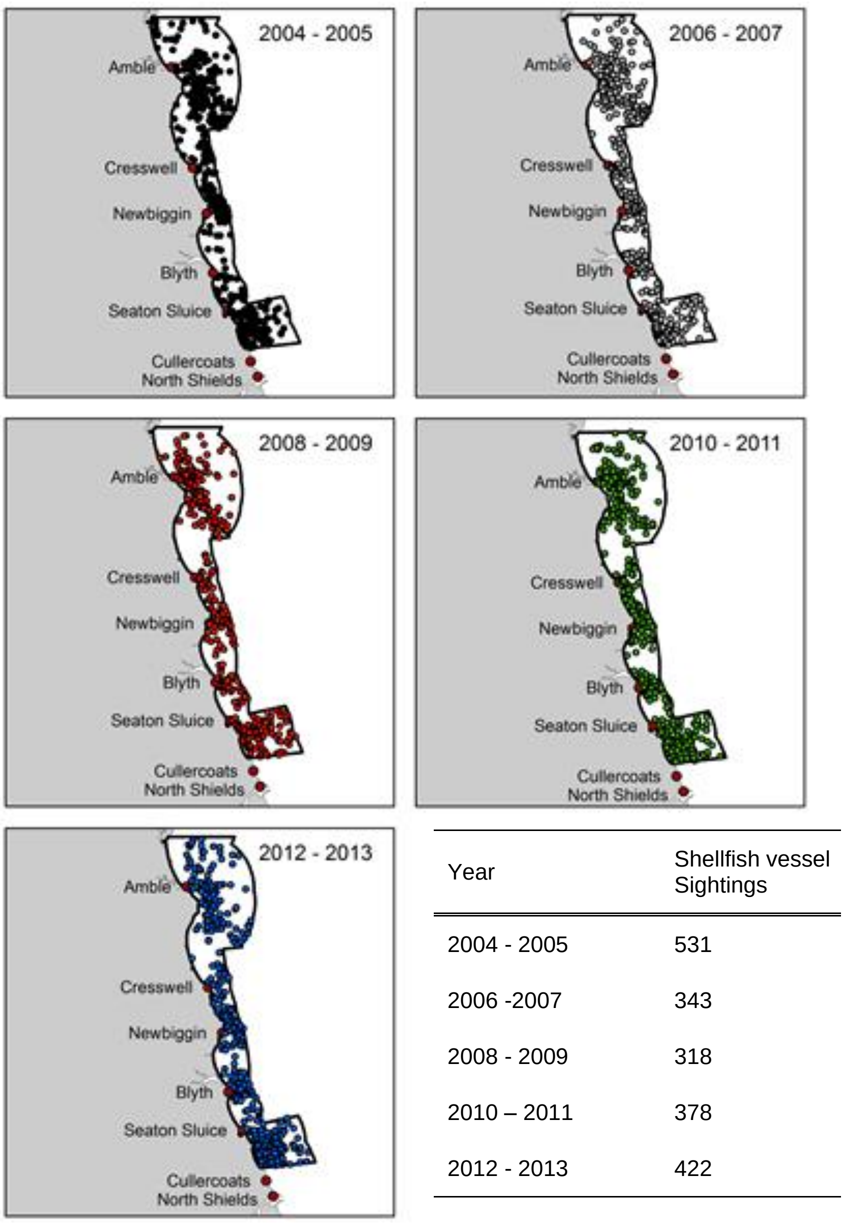


Fig 3.4. EUNIS level 3 - 5 habitat map of the CQSM MCZ (from Lightfoot, unpublished)

3.2.2. *Observed potting activity*

Potting vessel sightings (chapter 2) were extracted for the CQSM MCZ area and mapped as point data using ArcView GIS version 10.2 (ESRI, 2014) (Fig 3.5). Two year groupings were used for the fishing vessel distribution analysis as data from a single year were too sparse to accurately map. Temporal fishing effort distribution maps (chapter 2) were combined with habitat data (section 3.2.1) in ArcView GIS (v 10.2) and changes over time examined.



Year	Shellfish vessel Sightings
2004 - 2005	531
2006 -2007	343
2008 - 2009	318
2010 – 2011	378
2012 - 2013	422

Fig 3.5. Potting vessel sightings (2004 – 2005, black dots; 2006 -2007, grey dots; 2008 – 2009, red dots; 2010 – 2011, green dots; 2012 – 2013, blue dots) in the CQSM MCZ (black line).

3.2.3. Analysis of fishers spatial distribution

The presence of spatial clustering of fishing vessel sightings for each year group (2004 – 2005; 2006 – 2007; 2008 – 2009; 2010 – 2011; 2012 – 2013) within the CQSM MCZ was investigated using Ripley's K function (Ripley, 1977; Ripley, 1979) including an isotropic edge correction function (Ohser, 1983; Ripley, 1991) in the Spatstat package (Baddeley and Turner, 2005) in R (v 0.98.1103) (R Core Team, 2013). The K function $\widehat{K}(h)$ describes characteristics of the point processes at many distance scales (h) (Dixon, 2002). The numbers of fishing vessel sightings were tested for complete spatial randomness (i.e. testing whether fishing vessel sightings were distributed in a homogenous Poisson process) using the associated $\hat{L}(h)$ metric. This is easier to use in practice, as variance is approximately constant where spatial randomness is complete (Dixon, 2002). $\hat{L}(h)$ was plotted against h with values of zero suggesting spatial randomness and peaks in positive values suggesting clustering (Budge *et al.*, 2014). Formal assessment of significance of observed peaks was investigated through the calculation of upper and lower envelopes of $\hat{L}(h)$ under complete spatial randomness by 500 independent Monte Carlo simulations. $\hat{L}(h)$ of fishers sightings greater than these simulation envelopes indicated significant clustering (Bailey and Gatrell, 1995; Budge *et al.*, 2014). Maximum distance h was selected by $(A\ /\ 2)^{1/2}$ where A is the area (km^2) of the study region in order to avoid over inflated edge effects (Dixon, 2002) – in this case maximum h was 6 km. All R code used is available in appendix 2.

3.2.4. Analysis of fishers' space-time distribution

Fishing is a dynamic process where spatial distributions change over time due to factors such as catch levels, fuel costs, management measures and territorial behaviour (Abernethy *et al.*, 2007). It is important to investigate whether observed spatial clustering patterns change over time in order to provide information, and allow monitoring of, productive fishing grounds or preferred habitats (Bailey and Gatrell, 1995). *K* function space – time cluster analysis was undertaken to investigate if space-time interactions were present in fishing vessel sightings in the CQSM CMZ between 2004 – 2013, using the Splancs R package (Rowlingson and Diggle, 2013). The bivariate space-time *K* function $\widehat{K}(h,t)$ is defined as the expected number of fishing vessel sightings within distance h and time interval t, scaled by the number of fishing vessels sightings expected if complete spatial randomness is assumed, per unit area and time (Bailey and Gatrell, 1995). If space and time do not interact then $\widehat{K}(h,t)$ is just the product of the two *K* functions: $\widehat{K}(h)$ and $\widehat{K}(t)$ (Bailey and Gatrell, 1995). Thus a test for space-time interaction is based on the observed differences of:

$$\widehat{D}(h,t) = \widehat{K}(h,t) - \widehat{K}(h) * \widehat{K}(t)$$

$\widehat{D}(h,t)$ values were plotted against distance and time in a three dimensional plot with values above zero indicating space – time clustering (Bailey and Gatrell, 1995). To test for significance, 500 simulations were performed, where *n* fishing vessel sightings were randomly labelled with the observed *n* time markers. Where the actual $\widehat{D}(h,t)$ distribution was larger than 95% of the simulated values, it was taken as evidence of space-time interactions (Bailey and Gatrell, 1995).

To visualize these analyses, SaTScan software v 8.0 (Kulldorff *et al.*, 2009) was used to plot contours of space-time clusters in ArcGIS. Clusters of potting vessel sightings were detected using a Monte-Carlo space-time permutation scan statistic (Kulldorff *et al.*, 2005). This detects spatial and temporal groupings of events by centering a time-space cylinder on each event in the dataset (i.e. GPS location of potting vessel sighting) (Kulldorff, 2015). The diameter of the cylinder defines the geographical area of the cluster whilst its height defines time (in this case the number of years). This results in many overlapping cylinders which are assessed for significance by comparing the observed number of potting vessel sightings within the cylinder to the simulated expected number of observations (999 Monte Carlo permutations) (Kulldorff *et al.*, 2005; Webb *et al.*, 2008). Specifically, clusters which persisted through all years were examined further.

3.2.5. Analysis of fishers' habitat use

Compositional analysis (Aebischer *et al.*, 1993) of fishers' habitat use was undertaken by investigating the relationship between observed and expected potting use of each categorical habitat type in the CQSM MCZ (under the null assumption that habitat is used proportionally to availability) using 500 randomisation tests for annual groupings of potting vessel sightings using the adehabitatHS package (Calenge, 2006) in R. The significance of habitat selection was tested using a Wilks lambda and a ranking matrix was constructed. The ranking matrix indicated whether the habitat type was used significantly more or less than expected for each yearly fishing vessel sightings value, and ranking of habitat selection by fishing vessels and year in order of preference was displayed (Calenge, 2006). The extent of the CQSM MCZ polygon outline was selected as fishing vessel home range (available habitat, second order habitat selection (Johnson, 1980)).

In order to further explore differences in habitat use between years an eigenanalysis of selection ratios was undertaken (Calenge and Dufour, 2006). This method undertakes an additive linear partitioning of the White and Garrott statistic (White and Garrott, 1990), so that the difference between habitat use and availability is maximized on the first factorial axes (Calenge and Dufour, 2006).

3.3. Results

3.3.1. Fishers' distribution in space

There was significant aggregation of potting vessel sightings in the CQSM MCZ for all years (Fig 3.6). Measures of spatial aggregation (*L*) were all larger than the 95% confidence envelopes of the simulated spatially random points from 0 – 5500m. Clustering patterns for all bi-annual potting vessel sightings were similar. *L* for 2004 – 2005, 2010 – 2011 and 2012 – 2013 reached a peak at 2500 m compared to 2006 - 2007 and 2008 – 2009 which reached maximum clustering at closer distances (1500 and 2250m respectively) (Fig 3.6).

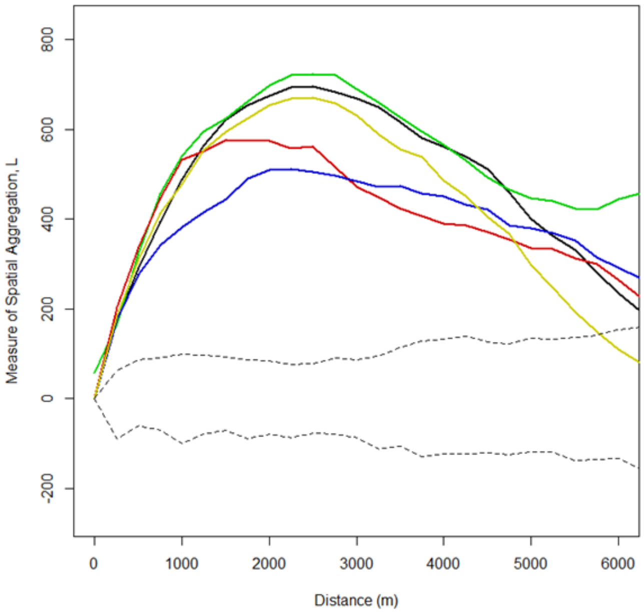


Fig 3.6. Estimated number of fishers' sightings within distance (m) for years 2004 – 2005 (grey), 2006 -2007 (red), 2008 – 2009 (blue), 2010 – 2011 (yellow) and 2012 – 2013(green). Broken lines indicate the simulation envelopes above which significant clustering occurs.

3.3.2. Fishers' distribution in space-time

Given spatial clustering of fishing vessel sightings up to a distance of 5500m, spatial clustering over time was also investigated (Fig 3.7, A). The space-time *K*-function showed evidence of significant clustering of fishing vessel sightings at distances of 0 – 6000m over 6 year periods (time 1 - 3 in Fig 3.7, A) – with the peak at 5700 m over a 4 year period (Fig 3.7, A). Smaller yet still significant clustering at distances of 2000 – 3200 m over the 10 year period was also detected (Fig 3.7, A). The Monte Carlo significance test statistic (15 x 10^7) was significantly larger than the randomly generated permutations (Fig 3.7, B).

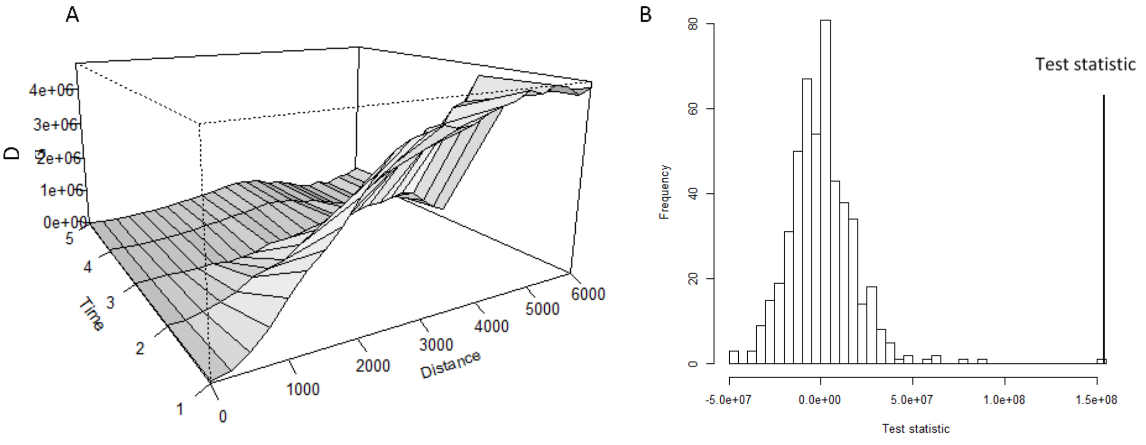


Fig 3.7. Space–time clustering of fishers' vessel sightings using K-function tests. Three-dimensional plot of the D(s,t) function (A). High values on the z axis indicate there is greater aggregation of fishing within the given spatial and temporal separation than would be expected if there were no clustering. Monte Carlo significance test of a space–time interaction (B). The bold line indicates the data statistic, which is larger than that of the 95% of the Monte Carlo samples, indicating significant space–time clustering.

SaTScan space-time permutation scan statistic showed evidence of potting vessel sightings aggregation with 19 significant clusters identified (all clusters highly significant, Table 3.3). These clusters varied in location, time, cluster size and duration (Table 3.3). The location, size and duration of fishing vessel sightings clusters in the CQSM MCZ which persisted through the majority of years were displayed in Fig 3.8. Seven clusters, located close to ports, showed evidence of consistent use by fishers over the course of a decade (Fig 3.8) and when compared visually with vessel sightings data match very closely (Fig 3.5). Other clusters were omitted from the map as these were either overlapping and/or occurred over fewer years.

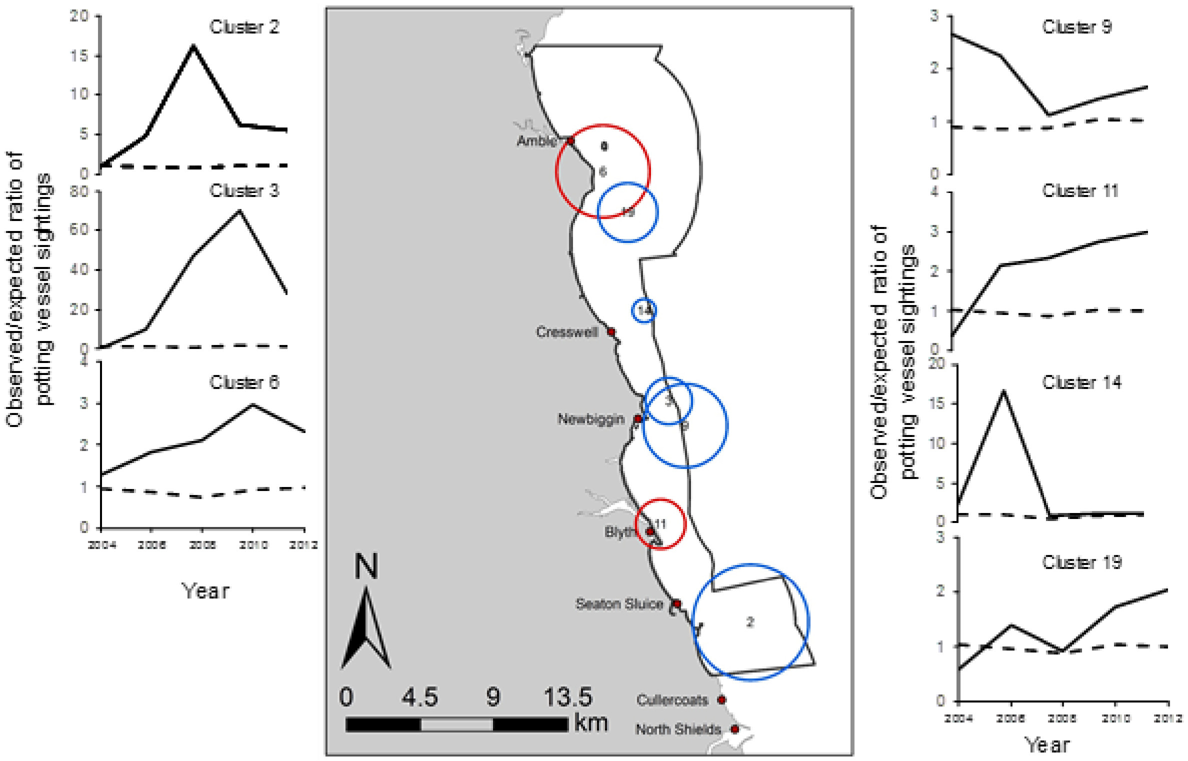


Fig 3.8. Significant SaTScan space-time clusters (open circles of approximate radius, p-value < 0.05) in the CQSM MCZ. Numbers refer to cluster rank, with 1 being the most likely. Clusters that persisted through time are presented: 2006 – 2013 (red circles) and 2004 – 2013 (blue circles). A list of all other space – time clustering is presented in Table 3.3. Inset graphs show the ratio of observed versus expected potting vessel sightings over time in numbered clusters. Black lines represent ratio within the cluster and dashed lines represent ratio outside the cluster.

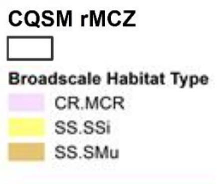
Table 3.3. Significant space–time clusters of potting vessels from the CQSM MCZ from 2004 - 2013 identified by SaTScan. Clusters are ordered by significance.

Cluster number	**Radius (km)**	**Start year**	**End year**	**p-value**	**Observed potting vessels**	**Expected potting vessels**	**Latitude of centroids**	**Longitude of centroids**
1	3.35	2004	2013	< 0.001	478	149.39	55.05541	-1.44754
2	3.53	2004	2013	< 0.001	597	306.13	55.073161	-1.400307
3	1.41	2004	2013	< 0.001	171	41.30	55.194841	-1.477033
4	3.99	2004	2013	< 0.001	516	263.85	55.091357	-1.447045
5	1.00	2006	2013	< 0.001	95	12.84	55.334386	-1.53826
6	2.82	2006	2013	< 0.001	332	147.11	55.320906	-1.538417
7	3.53	2004	2013	< 0.001	341	157.89	55.212848	-1.484656
8	1.41	2006	2013	< 0.001	104	26.24	55.127374	-1.462231
9	2.54	2004	2013	< 0.001	222	101.74	55.181293	-1.461506
10	2.82	2004	2007	< 0.001	130	53.01	55.181492	-1.508619
11	1.50	2006	2013	< 0.001	114	44.10	55.127475	-1.485756
12	2.06	2008	2011	< 0.001	127	58.90	55.347866	-1.538103
13	1.00	2004	2007	< 0.001	26	3.99	55.289131	-1.460047
14	0.71	2004	2013	< 0.001	37	8.50	55.244366	-1.499979
15	1.58	2004	2007	< 0.001	38	9.57	55.280144	-1.460168
16	3.99	2010	2013	0.006	117	61.20	55.046043	-1.369407
17	3.15	2008	2011	0.006	117	65.88	55.339022	-1.577625
18	2.55	2010	2013	0.007	80	42.14	55.140819	-1.454206
19	1.80	2004	2013	0.011	188	128.09	55.298349	-1.515052

3.3.3. Fishers’ habitat use over time of EUNIS level 3 habitats

EUNIS level 3 habitat within temporally persistent SaTScan space-time clusters showed fishers targeted circalittoral rock (51km^2 within significant clusters) over both subtidal sand and subtidal mud which had similarly lower values (7.59 and 7.91km^2 respectively) (Fig 3.9). Formal testing of habitat use by potting vessels within the CQSM MCZ using compositional analysis confirmed potting vessel selection of rock over other substrata (Table 3.4).

Cluster number	Circalittoral rock (km^2)	Subtidal sand (km^2)	Subtidal mud (km^2)
2	27.2	0.75	3
3	2.5	0.47	0.16
6	7.36	0.46	2.3
9	4.1	2.9	0.26
11	2.2	1.3	0.7
14	0.7	0.29	0.1
19	6.95	1.42	1.39
Total	51.01	7.59	7.91



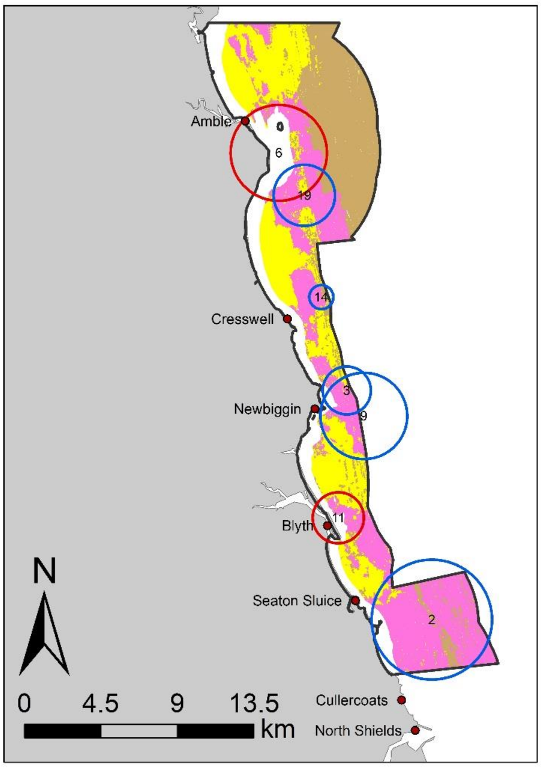


Fig 3.9. Significant temporally persistent SaTScan space-time clusters (2006 – 2013, red circles; 2004 – 2013, blue circles) overlaid on EUNIS level 3 habitat classifications of the CQSM MCZ. Inset table shows area (km^2) for habitat classifications in each space –time cluster.

Table 3.4. Proportion (%) of available and observed use of Eunis level 3 habitats for potting vessel sightings in the CQSM MCZ over 10 years.

Year	CR.MCR		SS.SSi		SS.SMu	
	Available (%)	Used (%)	Available (%)	Used (%)	Available (%)	Used (%)
2004	39	71.2	32	16.2	29	12.5
2005	39	64.3	32	11.8	29	23.7
2006	39	57.0	32	16.3	29	26.6
2007	39	68.2	32	14.5	29	17.2
2008	39	75.2	32	15.6	29	9.4
2009	39	70.1	32	15.8	29	14.0
2010	39	75.8	32	15.3	29	8.7
2011	39	69.6	32	15.1	29	15.1
2012	39	68.0	32	21.8	29	10.1
2013	39	74.0	32	14.4	29	11.5

For all years, potting vessel sightings differed among substrate types (Lambda = 0.0152; p = 0.004), with vessels showing a significant preference for moderate energy circalittoral rock habitat and avoidance of both sublittoral sand and sublittoral mud habitats (p-value < 0.05). Fishing vessels tended to target sublittoral sand over sublittoral mud habitats, although these differences were not significant. The factorial analysis showed all year groups following a similar pattern with strong selection for rock over other substrata (Fig 3.10).

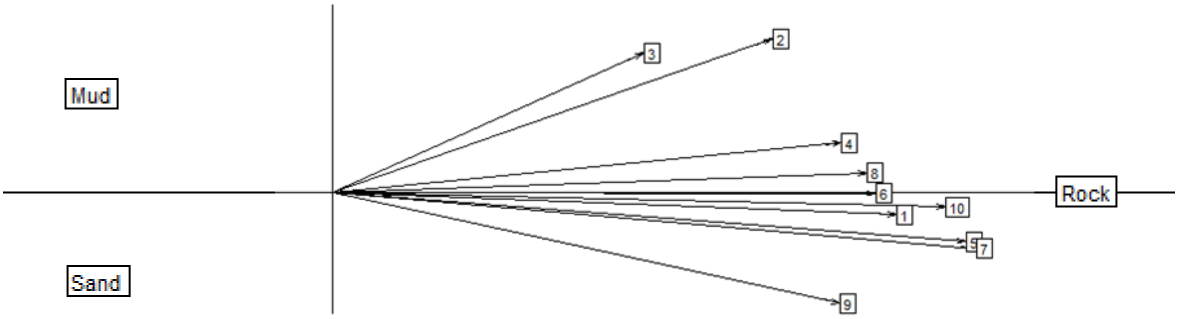


Fig 3.10. Habitat type loadings on the first two factorial axes and results of the eigenanalysis of selection ratios for habitat selection of potting vessels for ten years (2004 – 2013, labelled 1 - 10) on Eunis level 3 habitat variables.

3.3.4. Fishers' habitat use over time of EUNIS level 3-6 habitats

At a finer scale habitat level (EUNIS level 3 – 6), potting vessel sightings also differed among substrate types in all years (Lambda = 0.0157; p-value = 0.002). Fishers showed a significant preference for all rocky habitats over both sublittoral sand and mud habitats (Table 3.5, Fig 3.11). Fisher's preference of specific rocky habitats varied although all EUNIS level 5-6 rocky habitats were targeted over the broader classified CR.MCR habitat (Table 3.5). In addition, FaAlCr.Flu habitat was clearly selected over other rocky habitats (Table 3.5). FaAlCr was also preferred over other habitats (with the exception of FaAlCr.Flu and a non-significant preference over FaAlCr.ADig and FaAlCr.Pom) (Table 3.5).

Table 3.5. Fishers’ EUNIS level 5 habitat preference matrix. The order of listed habitats in the first column are ranked from most preferred at the top to least preferred at the bottom. + indicates preference, - indicates avoidance, grey cells are significant selectivity ($p < 0.05$).

	FaAlCr.Flu	FaAlCr	FaAlCr.ADig	FaAlCr.Pom	CR.MCR	SS.SSa	SS.SMu
FaAlCr.Flu	0	+	+	+	+	+	+
FaAlCr	-	0	+	+	+	+	+
FaAlCr.ADig	-	-	0	+	+	+	+
FaAlCr.Pom	-	-	-	0	+	+	+
CR.MCR	-	-	-	-	0	+	+
SS.SSa	-	-	-	-	-	0	+
SS.SMu	-	-	-	-	-	-	0

Eigenanalysis of selection ratios for habitat selection of potting vessels showed that in 2004 fishers targeted FaAlCr.ADig over other habitats (Fig 3.11). Fishers in all other years (2005 – 2013) targeted FaAlCr, FaAlCr.Pom and FaAlCr.Flu over other habitats (Fig 3.11). Overall, habitat preference between years were similar with tight grouping in the eigenanalysis of selection ratios, with the exception of 2004 habitat selections (Fig 3.11), and small variations between years in the proportion of observed habitat use (Table 3.6).

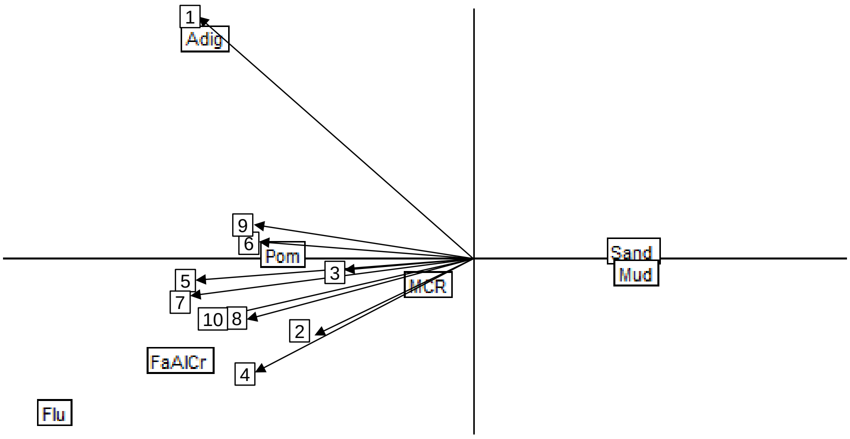


Fig 3.11. Habitat type loadings on the first two factorial axes and results of the eigenanalysis of selection ratios for habitat selection of potting vessels for ten years (2004 – 2013, labelled 1 - 10) on Eunis level 5 habitat variables

Table 3.6. Proportion (%) of available and observed use of EUNIS level 5 habitats for potting vessel sightings in the CQSM MCZ over 10 years.

Year	CR.MCR		SS.SSa		SS.SMu		FaAlCr		FaAlCr.ADig		FaAlCr.Pom		FaAlCr.Flu	
	Available (%)	Used (%)	Available (%)	Used (%)	Available (%)	Used (%)	Available (%)	Used (%)	Available (%)	Used (%)	Available (%)	Used (%)	Available (%)	Used (%)
2004	6	5.6	32	11.5	29	8.9	16	21.3	11	43.6	3	4.9	3	3.9
2005	6	10.1	32	12.3	29	24.6	16	30.4	11	11.6	3	4.3	3	5.8
2006	6	6.1	32	16.6	29	27.1	16	24.3	11	15.5	3	4.4	3	5.5
2007	6	7.4	32	14.9	29	17.6	16	37.8	11	11.5	3	2.7	3	8.1
2008	6	4.3	32	15.7	29	9.6	16	34.8	11	20.9	3	4.3	3	9.6
2009	6	13.4	32	15.9	29	14	16	25.6	11	19.5	3	5.5	3	6.1
2010	6	3.4	32	15.7	29	9	16	38.2	11	19.7	3	5.1	3	8.4
2011	6	6.4	32	15.4	29	15.4	16	34	11	16	3	4.5	3	7.7
2012	6	7.1	32	22	29	10.1	16	29.8	11	20.8	3	4.8	3	5.4
2013	6	7.9	32	14.9	29	11.9	16	32.7	11	15.8	3	7.4	3	8.9

3.3.5. Changes in habitat potting pressure

Potting pressure (pots fished km^{-2} year^{-1}) in the NIFCA district increased over the period 2004 – 2013 (chapter 2), especially in areas close to shore such as the CQSM MCZ (Fig 3.12). Unfished areas decreased in extent over this time.

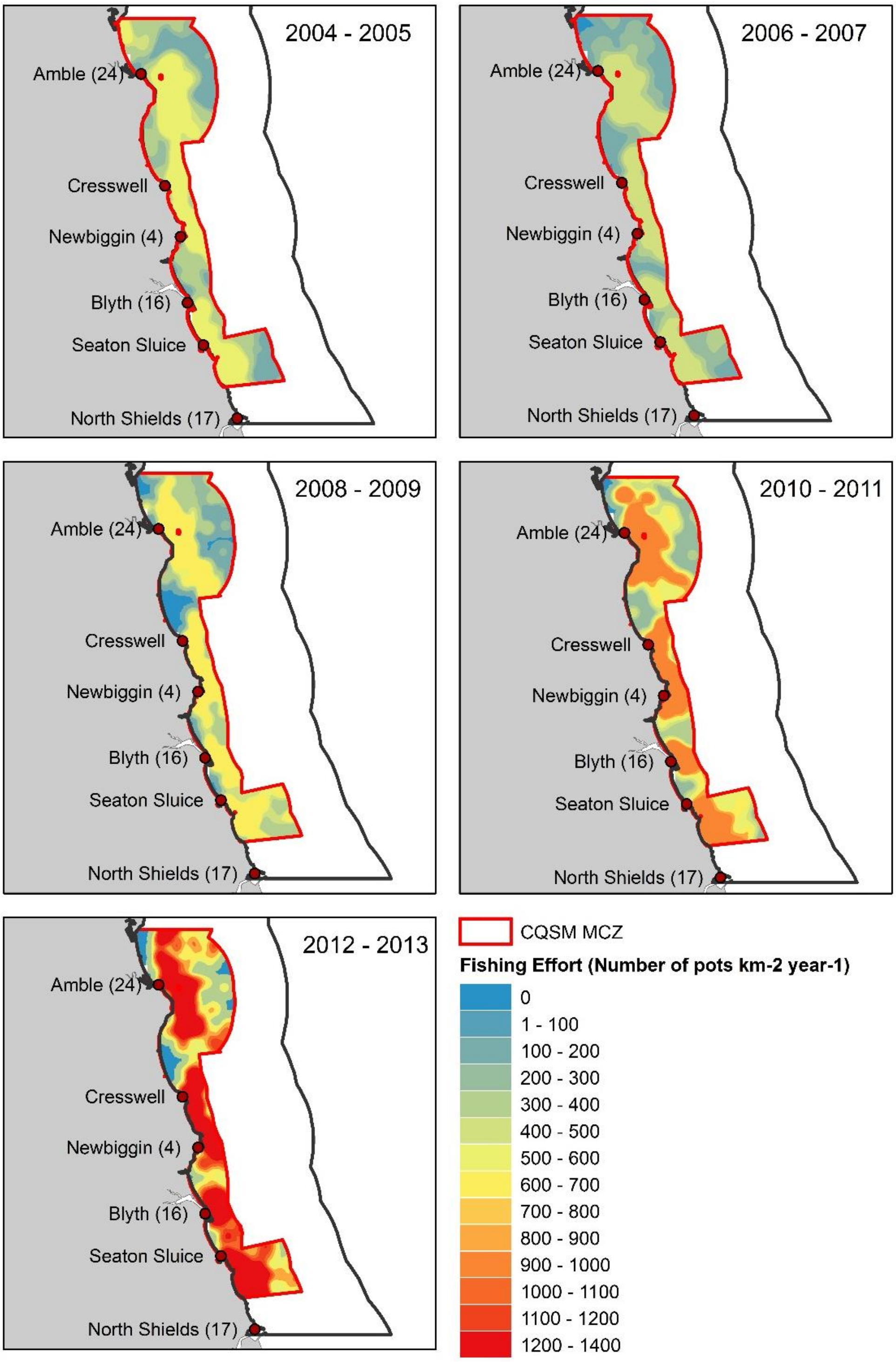


Fig 3.12. Distribution of fishing effort (number of pots km^{-2} year^{-1}) in the CQSM MCZ for years: 2004 – 2005; 2006 – 2007; 2008 – 2009; 2010 – 2011; 2012 – 2013.

Across all years the lowest fishing pressures were on sand and mud habitats (SS.SSa and SS.SMu, respectively) followed by CR.MCR (Table 3.7 and Table 3.8). FaAlCr.Pom and FaAlCr.Flu habitats had the highest fishing pressure across all years (Table 3.8). Potting pressure more than doubled on each habitat between 2004 and 2013 (EUNIS levels 3- 5, Table 3.7 and Table 3.8) but the proportion of total pots deployed on each habitat remained constant, with only slight increases or decreases between years (Table 3.7 and Table 3.8). Between 2004 - 2013 fishers consistently targeted the same habitats each year and with increasing potting effort.

Table 3.7. Mean potting pressure (number of pots km $^{-2}$ year $^{-1}$) per habitat (EUNIS level 3) and proportion of pots deployed in the CQSM MCZ.

	2004 – 2005		2006 – 2007		2008 – 2009		2010 – 2011		2012 – 2013	
	Potting pressure	Proportion of pots	Potting pressure	Proportion of pots	Potting pressure	Proportion of pots	Potting pressure	Proportion of pots	Potting pressure	Proportion of pots
CR.MCR	448.0	0.45	363.2	0.46	565.9	0.50	800.3	0.47	1117.4	0.48
SS.SSa	377.4	0.31	286.8	0.29	414.2	0.29	642.4	0.30	898.5	0.31
SS.SMu	319.5	0.24	275.7	0.25	327.9	0.21	521.8	0.22	649.8	0.21

Table 3.8. Mean potting pressure (number of pots km $^{-2}$ year $^{-1}$) per habitat (EUNIS level 3 - 5) and proportion of pots deployed in the CQSM MCZ.

	2004 – 2005		2006 – 2007		2008 – 2009		2010 – 2011		2012 – 2013	
	Potting pressure	Proportion of pots	Potting pressure	Proportion of pots	Potting pressure	Proportion of pots	Potting pressure	Proportion of pots	Potting pressure	Proportion of pots
CR.MCR	406.1	0.14	342.4	0.14	549.1	0.15	747.5	0.14	1034.6	0.14
SS.SSa	377.4	0.13	286.8	0.12	414.2	0.12	642.4	0.12	898.5	0.12
SS.SMu	319.5	0.11	275.7	0.11	327.9	0.09	521.8	0.10	649.8	0.09
FaAlCr	468.3	0.16	372.2	0.15	573.0	0.16	828.9	0.16	1160.8	0.16
FaAlCr.ADig	416.9	0.14	347.8	0.14	552.3	0.15	753.1	0.14	1043.6	0.14
FaAlCr.Pom	504.0	0.17	390.8	0.16	582.8	0.16	868.7	0.17	1226.1	0.17
FaAlCr.Flu	491.3	0.16	388.1	0.16	594.0	0.17	867.7	0.17	1228.5	0.17

3.4. Discussion

Fishers' space-time distribution analysis showed clustering occurred in Northumberland between 2004 and 2013 and targeted rocky over sediment habitats. In this section the implications of observed spatial clustering and habitat selection over a decade are discussed in relation to management, and recommendations made on the scale of mapped habitat hierarchical classification for future use.

3.4.1. Fishers' space and space – time clustering

Fishers clustered in space and time at various distances across all years. They were not randomly distributed in the CQSM MCZ but actively targeting specific areas of the seabed. Since fishers target unseen prey, decisions of fishing location are based on knowledge from previous catches, observation of competitor distributions, environmental hazards, technological limitations and perceptions of other constraints (FAO, 2005; Abernethy *et al.*, 2007). Catches per trap of target shellfish species vary depending on habitat type (Miller, 1989; Geraldi *et al.*, 2009; Skerritt, 2014) and as such, it is expected that fishers will target seabed habitats that provide the most profitable catches. The Ideal Free Distribution theory (IFD) predicts that the distribution of foraging organisms (in this case fishers) between sites will match the distribution of resources (Fretwell and Lucas, 1969; Kennedy and Gray, 1993; Gillis, 2003; Rijnsdorp *et al.*, 2011). It assumes that organisms are free to enter and use the area, ideally distribute themselves to maximise foraging efficiency and will have a perfect knowledge of the resource profitabilities (Kennedy and Gray, 1993). Although this ecological theory is intuitive, by application to fishers, experimental results have shown key assumptions of the ideal free distribution theory for site selection were rarely met (Abernethy *et al.*, 2007). This was attributed to imperfect fisher knowledge resulting in selection of areas without the greatest rewards (Abernethy *et al.*, 2007) and/or not all fishers seeking to maximise profit by increasing fishing pressure with increasing resources (Swain and Wade, 2003; Abernethy *et al.*, 2007). In addition, fishers were not free to distribute themselves among habitats as groups of fishers are territorial, and there are other physical constraints (i.e. boat size, distance from port, etc.) (Abernethy *et al.*, 2007). In Northumberland, fishers have local ecological knowledge that is underpinned by historic information as well as widespread use of acoustic and positioning technology which may allow targeting at a broad habitat level (i.e. rock, sand and mud). It is not known whether fishers have knowledge of community composition (EUNIS level 4 -6) although this seems unlikely. Thus fishermen do not have a perfect knowledge of the resource profitabilities. In addition, territoriality which has been widely documented in shellfisheries both internationally (Acheson, 1988) and locally (Turner *et al.*, 2012) means fishers' distribution is neither 'ideal' nor 'free' within the NIFCA district (Turner *et al.*, 2012).

Temporal clustering between 2000 – 3200m occurred across the 10 year-period (Fig 3.7) with yearly peaks in clustering also occurring at this range (Fig 3.6). Landing ports in the study area (an indication of territory size (Turner, 2010)) were a mean of 6250m apart – a larger distance than the observed clustering – and may therefore only in part explain distances between clusters. It is proposed here that fisher distribution in Northumberland is likely to be governed by a combination of fisher behaviour targeting areas with high catches and within their given territories. The exact influence of each of these factors in the final distribution of temporally persistent clustering is difficult to elucidate with ecological data alone. Social drivers would need to be investigated to fully understand fisher behaviour and distribution. In addition, the question also remains whether fishers are targeting specific habitats or are simply persistently targeting areas that have high catches, regardless of the underlying seabed habitat. This will be further discussed in the following section.

3.4.2. Fishers' habitat use

At a broad scale, fishers in the CQSM MCZ showed a preference for rocky habitats over sediment habitats (Table 3.4) and at a finer scale showed preference for *Flustra foliacea* on slightly scoured silty circalittoral rock (FaAlCr.Flu) and Faunal and algal crusts on exposed to moderately wave-exposed circalittoral rock (FaAlCr) habitats over other rocky habitats (Table 3.5). This assumes that the distribution and extent of habitats predicted from data collected in 2014 are representative of the seabed throughout 2004 - 2013, but in this moderate energy environment at circalittoral depths, major changes in the distribution of substrata and broad community types are unlikely. Whether fishing grounds are chosen depending on fishers' knowledge of habitat, 'following' catch of target species or a combination of both is unknown. In addition, there is little information on how fishers perceive subtidal habitats; whether they have estimates on the hardness (i.e. hard and soft ground), based on sonar or gear deployment, or whether they have knowledge about the biology in these habitats, through historical knowledge and observation of epibenthic species caught in the mesh of the pots.

It is likely that fishers know seafloor hardness from historical experience and/or sonar. Therefore, fishermen may be targeting habitat type at a broadscale level. At a fine scale, it is unlikely that fishers are targeting specific biological communities but rather targeting areas which have the highest catch. Potting in Northumberland is a mixed fishery and it is not clear which species were targeted. The inshore location of the MCZ and fishers' active preference for rocky substrate suggests they were targeting high value European lobster (Galparsoro *et al.*, 2009; Turner *et al.*, 2012), although brown and velvet crab species may still be caught in relatively large numbers as by-catch. Separating effort for each species requires further data with either more detailed landings forms, fishing vessel on-board observations of catch or social surveys of fishers.

Habitat preference at both classification levels varied very little between 2004 - 2013 (Fig 3.10 and Fig 3.11), albeit with increasing fishing pressure over time (Table 3.7, Table 3.8). Although fishing pressures were similar between rocky habitats, differences in biological communities provided by higher classification levels may help prediction or testing of, and subsequently management of various anthropogenic impacts. Potting impacts on various habitats and at different classification levels are explored in chapters 4, 5 and 6. However, for currently designated MCZs or EMSs evidence is needed for 'features of interest' at broad scales (EUNIS level 3).

Fishing pressure increased across the MCZ (Fig 3.12). Repeat potting in areas over the course of the decade may have negatively affected epibenthos, either through direct or indirect impacts (Eno *et al.*, 2001; Coleman *et al.*, 2013). As potting has persisted in locations over time, indirect habitat effects (i.e. changes to habitat through species removal, trophic cascades, or unstable food webs) may also occur (Babcock *et al.*, 1999; Siddon and Witman, 2004; Babcock *et al.*, 2010) (see chapter 4 and 5). Potting effort levels that impact the benthos require further information on interactions of the pot with benthos (see chapter 6). Further evidence is required, namely, larger sample size of spatial habitat data over time (chapter 4 and 5). These issues are of particular importance in the UK with the recent protection of rocky reef habitats as features of conservation interest in EMSs; this has particular implications for management, which is explored in later chapters.

Although this work focusses on Northumberland, generalisations can be made. The shallow inshore habitats observed are representative of many other inshore areas in the UK and Europe in terms of habitat composition and diversity. This is especially true for the East coast of Scotland and England (Brazier *et al.*, 1998). However, due to the inshore nature of the study area, deeper habitats further offshore are not represented - a large portion of the habitat types available for shellfish potting. Fishers targeting offshore areas may have fishing strategies which vary depending on the target species and fishing equipment and may result in very different clustering and habitat use. Anecdotal evidence suggests that fishers may show preference for targeting sediment habitat further offshore in order to target high crab landings (NIFCA pers. comm.). Investigation into fishers' habitat use for areas further offshore are recommended.

3.4.3. Management considerations

The high resolution habitat maps (1m) allowed distribution of fishers to be examined on a fine scale. Furthermore, the resolution allowed classification up to EUNIS level 6 for which a minimum scale of 5 x 5m is recommended (i.e. low resolution maps may have more than one biotope across each pixel, thus increasing uncertainty in mapped outputs). Results of temporal fisher-habitat interactions presented here had finer spatial and biological resolution than previous studies (Nilsson and Ziegler, 2007; Stelzenmüller *et al.*, 2008; Lambert *et al.*, 2011), as well as providing a quantified estimate of fishers' historical use of an MCZ - evidence which is often lacking during the stakeholder engagement process when consulting on designations (Caveen *et al.*, 2014). There were, however, some limitations with the data and resulting maps. Namely, four habitats which were infrequently observed in the ground truthed data (sublittoral coarse sediment and sublittoral mixed sediments at EUNIS level 3 and *Sabellaria spinulosa* encrusted circalittoral rock and Brittlestars on faunal and algal encrusted exposed to moderately wave-exposed circalittoral rock at EUNIS level 6) were excluded from OBIA models used to create the habitat maps as the low sample number resulted in a low confidence for distribution of these habitats (Lightfoot, unpublished). In addition, shallow waters (< 7m depth) were not surveyed in the MCZ CQSM as these were too shallow for the research vessel to safely sample in (Fitzsimmons *et al.*, 2015). Therefore, kelp dominated habitats were not included in maps – areas which are heavily targeted by fishers in the summer months (chapter 6) and where important habitat selection is likely to have taken place.

Different levels of EUNIS classification resulted in different (although consistent) results for fisher-habitat interactions. This has implications for the use of these data in spatial marine management. Carrying out analyses using broad habitat classification at EUNIS level 3 had the advantage of having higher accuracy than the EUNIS level 3 - 6 map, although the latter provided greater biological resolution which is particularly useful for the location and potential conservation of species of interest (Caveen *et al.*, 2014). This level of biological detail is often recommended in the literature in order to inform EBFM (Cogan *et al.*, 2009). However, currently management measures are focused on a broader scale (EUNIS level 3) (Fitzsimmons *et al.*, 2015). The relatively low accuracy of the of the EUNIS level 3-6 map is largely due to the topographic, hydrographic and biological homogeneity of the study site. The biological communities present are similar in composition and ecological requirements, and grade into one another rather than having abrupt boundaries. These methods of analysing fisheries-habitat interactions hold great potential for wider application, particularly in areas of greater spatial heterogeneity where the prediction of biological communities from acoustic data can be achieved with higher accuracy (Hill *et al.*, 2014; Sotheran *et al.*, 2014).

Successful implementation of MPAs have been questioned due to the incomplete knowledge of benthic assemblages, of fishing gear-habitat interactions and of fisher displacement from fishing restrictions (Caveen *et al.*, 2014). The present research has helped fill some of these knowledge gaps for the newly designated CQSM MCZ. Several authors have questioned the validity of using broadscale habitat maps (EUNIS level 3) for management since these contain insufficient detail to support robust planning; specific taxa only being introduced from level 4 (Cogan *et al.*, 2009; Caveen *et al.*, 2014). Level 3 - 6 habitat maps showed that fishers (potentially inadvertently) were selective in their use of biological communities. Potting impacts may be different between these different biological communities (Eno *et al.*, 2013). It is recommended that maps predicting the distribution of biological communities coupled with usage information are used for EBFM in order to allow adaptive management (Eno *et al.*, 2013). These spatial assessments will be particularly useful for prioritising protection of the most vulnerable or biologically important habitats (Eno *et al.*, 2013). However, the impact of closures on fishers' distribution and fishing effort is not well understood.

Results from this research suggest that fishers distribute themselves within their territories according to habitat type. This has implications for fisher redistribution due to fisheries closures – an aspect of marine spatial planning that has not been investigated for inshore fisheries in the UK to date. The effects of fishery closures will be gear specific (Campbell *et al.*, 2014) and displacement of pot fishers by areal closures may have greater impact than that of mobile gear users because potting is highly localised (Campbell *et al.*, 2014; Guenther *et al.*, 2015; Turner *et al.*, 2015). When fishery closures are enforced, spillover is frequently assumed to occur (McClanahan and Mangi, 2000). This assumes that the increase in target species abundance within the MPA will cause migration of individuals outside the closed area (Hoskin *et al.*, 2011) and that the displaced adjacent fisheries will focus fishing effort along the edges of the MPA as catches will be highest there (Guenther *et al.*, 2015). However, these models do not incorporate habitat specific fishing effort, habitat type or heterogeneity, or limits imposed by fishing methods/gear. For example, fishers showed preferences for rocky habitats over softer sediment habitats (Fig 3.10 and Fig 3.11). Thus habitat may not be suitable for the target species of the displaced fishers if these differ between the open and closed fisheries areas. In addition, fixed-gear fisheries where pots are set, soaked, pulled and reset in the same location results in areas that are “marked” or occupied, and stop other fisherman fishing in the same location. This has resulted in high levels of territoriality amongst pot fishers (Acheson, 1975; Turner *et al.*, 2012). Displacement of potting effort may increase disputes, the effects and severity of which will largely be determined by *a priori* territorial distributions and habitat quality (Guenther *et al.*, 2015). After inshore fishing closures in California, USA, lobster fishermen targeted areas based on competition with other fishers and habitat quality (i.e. catch) rather than targeting areas close to MPA borders (Guenther *et al.*, 2015). 771.9km^2 (54.9% of the NIFCA district) are currently designated as MPAs (CQSM MCZ and Berwickshire North Northumberland Coast EMS), including the heavily fished inshore areas (Fig 3.12). Any future changes in management of these MPAs could lead to displacement of fishing activity. This is currently poorly understood, but the ability to predict socio-economic, as well as environmental, outcomes of management measures is crucial to sustainable marine management. Multicriteria decision support tools such as Marxan are increasingly used to combine a variety of spatially explicit selection criteria in a GIS for marine spatial planning purposes (Baban and Parry, 2001; Villa *et al.*, 2002; Lieberknecht *et al.*, 2004; Bruce and Eliot, 2006; Prest *et al.*, 2007). These tools primarily focus on the distribution of the natural resources important to the development of conservation value, often ignoring fishing distribution data which can minimise environmental and economic costs of areal closures by displacement (Dinmore *et al.*, 2003; Lieberknecht *et al.*, 2004; Richardson *et al.*, 2006; Campbell *et al.*, 2014) resulting in wider acceptance, cost-effective and politically feasible protected areas (Richardson *et al.*, 2006; Campbell *et al.*, 2014). Although results of this research provide a crucial step towards a better understanding of long term fishing effort and spatial distribution in Northumberland, potentially enabling a predictive capacity for Marxan type modelling, a lack of access to cumulative uses, socio-economic information and habitat data outside the CQSM MCZ raises scientific and socio-economic concerns about the underpinning of marine spatial management decisions (Campbell *et al.*, 2014).

In addition, to being useful in MPA planning, fisheries – habitat interaction information can be used for monitoring and predicting anthropogenic impacts. This approach was successfully used to explore possible impacts of fishing gears, used at different intensities, over different habitats across Welsh waters (Eno *et al.*, 2013). However, distribution of fishing intensities was not available, changes over time were not investigated and seabed sensitivities to fishing gears were informed by expert opinion and not quantified data. Temporal fishing intensity distributions (chapter 2 - fishing pressure) and habitat interactions (current chapter), provided crucial information for the investigation of long-term (chapters 4 and 5) and short-term potting impacts (chapter 6).

3.5. Concluding remarks

Fisher distribution was not random and temporal clustering was observed. Pot fishermen are unlikely to conform to the Ideal free distribution hypothesis due to constraints of territoriality. Distances between clusters suggested fishermen may be targeting specific areas with high catches within their territories. Fishers were shown to consistently prefer rocky habitats over sediment habitats at both a broadscale and fine scale classification level across all study years. Fishing pressures were similar between rocky habitats but increased over time. Fisheries – habitat information is recommended for the implementation of successful spatial management, for example, in order to fully understand and predict the effects of closures of fisher displacement. Although broadscale habitats are currently used for management, fishers' interaction with fine scale habitats are particularly important when predicting differences in biological communities. This can allow prediction of (or testing of) and subsequently management of various anthropogenic impacts. Potting impacts on various habitats and at different classification levels are explored in chapters 4, 5 and 6.

Chapter 4 . Long-Term Potting Intensity Effects on Biotopes and Species Composition and Richness

4.1. Introduction

There is growing evidence that long-term changes in marine ecosystems may occur due to fishing impacts (Kaiser et al., 1996; MacDonald et al., 1996; Kaiser et al., 2006). As part of the continued focus of UK fisheries management for an evidence based approach (Marine and Coastal Access Act, 2009), Defra revised their approach to management of commercial fisheries in European Marine Sites (EMSs), now known as the fishing in MPAs project, which considers the potential impacts of fishing activities via Habitats Regulations Assessments on the designated features of each site (MMO, 2012). Fishing activities will only be allowed if they do not adversely affect site integrity.

Potting activity is generally assumed to have little physical impact on hard substrate habitats and epibenthos. The few potting impact studies undertaken to date have focused on assessing impacts using indicator species perceived to be sensitive to potting due to their life history traits (erect, fragile and sessile) (Eno *et al.*, 2001; Shester and Micheli, 2011; Coleman *et al.*, 2013). However, potting may alter trophic dynamics through the removal of keystone species incurring ecosystem changes (Siddon and Witman, 2004). No peer-reviewed research into long-term indirect impacts of potting has been conducted to-date, but these are crucial to understanding pot fisheries interactions with the environment (Siddon and Whitman, 2004).

The Berwickshire & North Northumberland Coast European marine site (BNNC EMS) (Fig 4.1) incorporates 635 km^2 of shoreline and coastal waters. The area includes Lindisfarne, St Abbs, the Farne Islands and the Eyemouth voluntary marine reserve (AONB Partnership and EMS Management Group, 2009), hosting a diverse range of marine ecosystems and habitats including rocky shore, intertidal mudflats and sand beaches, rocky reefs, sea caves and shallow bays and inlets (Brazier *et al.*, 1998). Monitoring reports of the BNNC EMS (Edwards, 1983; Foster-Smith and Foster-Smith, 1987; Connor, 1989; Holt, 1994; Brazier *et al.*, 1998; Mercer *et al.*, 2003; Mercer, 2012) provide information in the form of habitat and their associated biological assemblages, and in later years used the UK biotope classification system. Although methods of classification have changed, these reports have similar levels of information on habitat and species, with changes mainly in the increased accuracy of sample location in more recent reports. Common biotopes have persisted within the BNNC EMS (Mercer, 2012). Although epibenthic communities are influenced by biotic factors (recruitment, competition, predation, etc.), their overall character is predominately determined by their ambient abiotic conditions (Connor *et al.*, 2004). A key feature of the biotope classification is that it takes into account abiotic factors including: substratum, depth, exposure to wave action/tidal currents, salinity, topography, geology, oxygenation and scour/turbidity. Therefore biotope classifications provide added information on individual abiotic factors – eliminating the need for collecting explanatory environmental variables. In addition, biotopes and species richness are often the only data available from long-term EMS monitoring surveillance work. However, little work has been done on assessing impacts on biotopes in the academic literature. This provides an opportunity to assess whether data collected as part of routine marine monitoring and surveillance can be used to investigate long-term, chronic changes in ecosystems and whether these are linked to anthropogenic impacts.

This chapter aims to investigate biotope changes in the BNNC EMS over a ten year period, using previously collected condition monitoring data from 2002 and 2011 (Mercer *et al*., 2003; Mercer 2012), and to investigate the possibility that these changes are related to the intensity of shellfish potting activity. Data were extracted from previously collected video monitoring footage and grouped into biotopes. Using a frequency analysis of biotopes it was hypothesised that the number, composition and range of biotopes would differ between 2002/03 and 2011 (Moore and Bunker, 2001; Saunders *et al.*, 2011) (hypothesis 1). In addition, temporal changes in the number, composition and range of biotopes were hypothesised to be related to the intensity of shellfish potting activity (hypothesis 2). Results are discussed in light of the hierarchical classification system used, since these are the most frequently used monitoring method. This approach is then critically appraised.

4.2. Methods

4.2.1. Site selection

Three transect corridors, distributed in the southern half of the Berwickshire North Northumberland Coast (BNNC) European Marine Site (EMS), were selected by Natural England in 2002 (formerly English Nature), and surveyed using drop-down cameras in 2002/03 (Fig 4.1, a) and again in 2011 (Fig 4.1, b) as part of the regular condition monitoring required under the terms of the Habitats Directive (Council Directive 92/43/EEC).

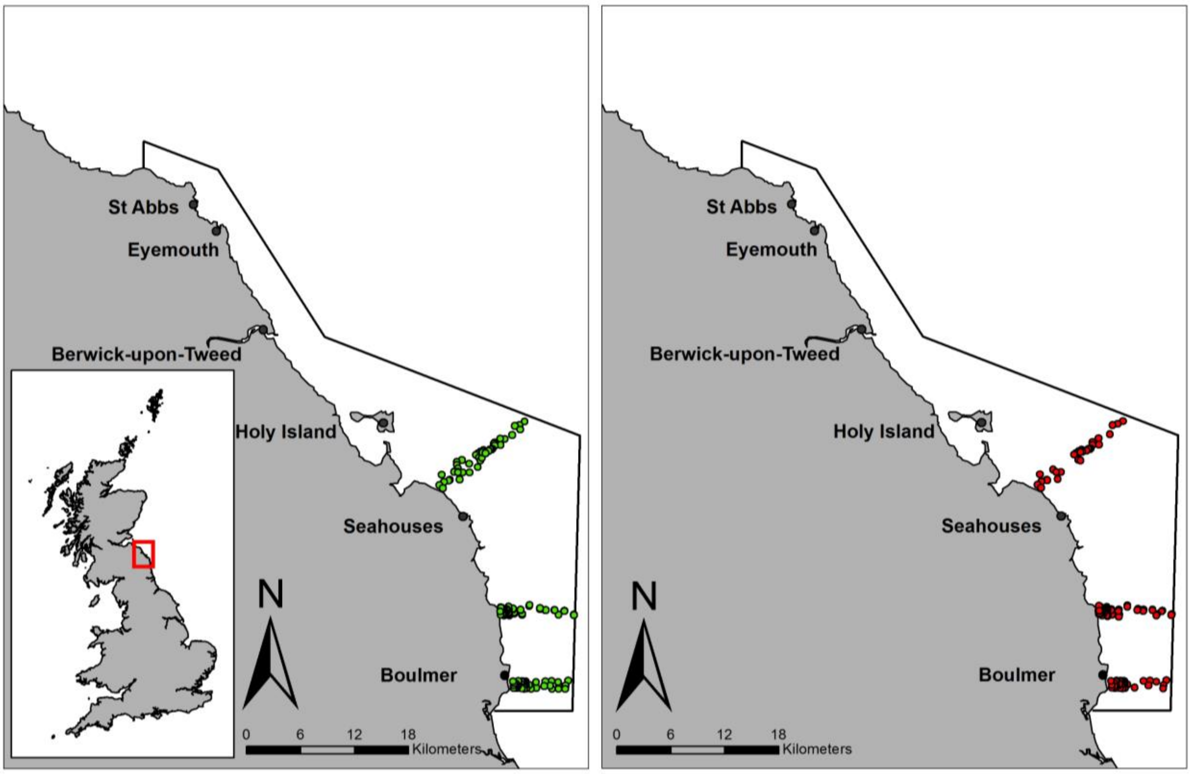


Fig 4.1. Berwickshire and North Northumberland Coast European Marine Site (black line) with a) 2002/3 corridor transects for the drop-down camera survey points b) 2011 corridor transects for the drop-down camera survey points.

4.2.2. Sampling protocol

In the original surveys, 2002/03 video fieldwork was undertaken during the weeks of 2nd - 7th August 2002, 30th August to 4th September 2002 and 4th - 18th June 2003. Field work in 2011 was undertaken between 23rd August and 22nd November 2011 (Mercer, 2012). Transect corridors were stratified by depth (0-10m, 10-20m and 20m+), each of which was randomly sampled using drop-down video (approximate tows of 100m) a maximum of 15 times. Fewer tows were performed if there was little variation in habitat and biotope (Mercer *et al.*, 2003). The drop-down video equipment and full methodology are described by Mercer *et al*. (2003) and Mercer (2012).

4.2.3. Data analysis

Prior analysis of this monitoring data was undertaken at a broad scale, for the purposes of Natural England condition monitoring. Biotopes were identified across the 100m tows and change between years was described by comparing biotope composition in transect corridors using only expert judgement (Mercer, 2012). This study reanalysed the data, using frequency counts of biotopes, to allow statistical analysis of change between years. This permitted investigation into the possibility that changes are related to the intensity of shellfish potting activity (described in chapter 2).

For the purposes of this study the video footage of the 100m tows was randomly sampled every 10–30 seconds (mean sampling time 17.2s, mean samples per tow = 9.1). When the video footage was paused the image was blurred due to the movement of the camera and low display resolution. Therefore, 2-second samples were used to identify substrate and species. When random samples did not yield usable footage, i.e. if bottom substrate was not visible, the next random time generated was used. Video footage from three transects was analysed for this project (Dunstanburgh, Boulmer and Farne Islands). The original monitoring reports (Mercer *et al.*, 2002; Mercer, 2012) used five transect corridors but the two most northern transects (Marshall Meadows and St Abbs) were not used because fishing pressure maps were more detailed and reliable in the South of the NIFCA district (see chapter 2). A total of 1724 samples were generated from 189 video tows.

Species present were recorded to the lowest taxonomic level using up-to-date identification manuals (Cornelius, 1995; Hayward and Ryland, 1995; Foster-Smith and Foster-Smith, 2000), online resources (MarLIN), and expert advice (Foster-Smith, J., pers comm. 2013). Biotope classifications were then assigned using identified species with the aid of ‘The Marine Habitat Classification for Britain and Ireland, Version 04.05’ (Connor *et al.,* 2004). Due to a lack of scale on the video footage, abundance and semi-quantitative scales (SACFOR) could not be used to inform the assignment of biotopes. Expert judgement was therefore used for biotope identification. This could be problematic when identifying certain biotopes at a finer scale from a single still image (e.g. determining between urchin grazed and ungrazed *Laminaria* spp dominated biotopes). In addition, samples were frequently disrupted as the camera rotated away from the benthos, either due to water movement or through contact with objects on the seabed; sampling times therefore varied to account for this. This approach accounts for the need for a biotope to be assessed at a scale of at least 5 x 5m (Connor *et al*., 2004) (a single static frame shows < 2 x 2m) as well as allowing time for identification of more cryptic species (i.e. if the camera rested on the seafloor, smaller species such as barnacles, small hydroids and encrusting bryozoan could be identified).

The maintenance of the number and range of biotopes was investigated between the years, focussing on the persistence of dominating community assemblages through the analysis of biotope frequency. The Wilcoxon signed rank test (non-parametric) was used with individual biotope frequencies from the two survey years treated as matched pairs (Moore and Bunker, 2005).

The sampling effort required to observe maximum biotope richness was investigated by producing an asymptotic sampling effort curve in Sigmaplot (v11). The Boulmer and Farne Island transects were chosen as they represent areas with low and high biotope richness respectively. Randomly selected biotopes for each sampling period in the Boulmer and Farne Island transect were counted to estimate biotope richness. This was repeated 15 times for each sampling event in order to take into account different biotope richness combinations. Mean biotope richness for any given sampling effort was then calculated using a second degree polynomial line of best fit.

4.3. Results

4.3.1. Biotope richness

Biotope richness was the same between survey periods for the Boulmer transect (Table 4.1). There were small differences in biotope composition between years although these were primarily due to sediment biotopes that are not easily identifiable using videographic methods. Overall, biotope composition and richness in the Boulmer transect were similar between survey periods, despite sample number (*n*) being higher in 2002/03 (Table 4.1).

Biotope richness for the Dunstanburgh transect was higher in 2002/03 than in 2011 for both classification levels (Table 4.1). However, *n* was also higher in 2002/03 than in 2011. Biotope composition and richness was similar between Boulmer and Dunstanburgh transects.

Biotope richness at the Farnes was higher in 2011 than in 2002/03 (Table 4.1), and was higher than in Boulmer and Dunstanburgh transects. Biotope richness of circalittoral biotopes was identical in both survey periods, differences being predominately driven by infralittoral biotopes found solely in 2011 (Table 4.1). At biotope classification level 5 (Table 4.1) this difference remains obvious; several infralittoral biotopes were found solely in 2011.

Biotope richness and composition was similar when classifying biotopes to level 6 or to level 5 for all transects (Table 4.1), albeit biotope richness was much lower in the latter. This was due to a high number of sub-biotopes identified for *Laminaria* spp dominated biotopes (IR.MIR.KR.Lhyp abbreviated to Lhyp) and Faunal and algal crust dominated biotopes (CR.MCR.EcCr.FaAlCr abbreviated to FaAlCr).

Table 4.1. Biotope and sub-biotope composition and standardised frequency data from two survey periods (2002/03 and 2011) for the Boulmer, Dunstanburgh and Farne Island transects. Cells highlighted in grey represent biotopes that were found solely in one survey period.

Transect	**Sub biotope**	**2002/03**	**2011**	**Biotope**	**2002/03**	**2011**
		Count	Count		Count	Count
Boulmer	IR.MIR.KR.Lhyp.Ft	6.1	7.5	IR.MIR.KR.Lhyp	12.2	17.5
	IR.MIR.KR.Lhyp.Pk	2.0	5.0			
	IR.MIR.KR.Lhyp.GzPk	4.1	5.0			
	CR.MCR.EcCr.FaAlCr	30.6	27.5	CR.MCR.EcCr.FaAlCr	55.1	50.0
	CR.MCR.EcCr.FaAlCr.Adig	6.1	0.0	CR.MCR.CSab.Sspi	2.0	2.5
	CR.MCR.EcCr.FaAlCr.Flu	16.3	10.0	SS.SCS.CCS	10.2	5.0
	CR.MCR.EcCr.FaAlCr.Pom	2.0	10.0	SS.Smu.CSaMu	10.2	0.0
	CR.MCR.EcCr.FaAlCr.Sec	0.0	2.5	SS.SBR.PoR.SspiMx	10.2	0.0
	CR.MCR.CSab.Sspi	2.0	2.5	SS.SMX.CMx	0.0	10.0
	SS.SBR.PoR.SspiMx	10.2	0.0	SS.SSA.CFiSa	0.0	15.0
	SS.SCS.CCS	10.2	5.0			
	SS.Smu.CSaMu	10.2	0.0			
	SS.SMX.CMx	0.0	10.0			
	SS.SSA.CFiSa	0.0	15.0			
Biotope richness		11	11		6	6
n		49	40		49	40
Dunstanburgh	IR.MIR.KR.Lhyp.Ft	4.5	5.1	IR.MIR.KR.Lhyp	18.2	7.7
	IR.MIR.KR.Lhyp.Pk	2.3	0.0	IR.MIR.KR.XFoR	2.3	0.0
	IR.MIR.KR.Lhyp.GzPk	11.4	2.6			
	IR.MIR.KR.XFoR	2.3	0.0			
	CR.MCR.EcCr.FaAlCr	31.8	46.2	CR.MCR.EcCr.FaAlCr	59.1	71.8
	CR.MCR.EcCr.FaAlCr.Adig	2.3	0.0	CR.MCR.EcCr.CarSp	2.3	0.0
	CR.MCR.EcCr.FaAlCr.Flu	15.9	10.3	CR.MCR.EcCR.UrtScr	0.0	2.6
	CR.MCR.EcCr.FaAlCr.Pom	6.8	12.8	CR.HCR.Xfa.SpNemAdia	2.3	0.0
	CR.MCR.EcCR.FaAlCr.Bri	2.3	2.6	SS.SBR.PoR.SspiMx	4.5	0.0
	CR.MCR.EcCr.CarSp.Bri	2.3	0.0	SS.SCS.CCS	9.1	10.3
	CR.MCR.EcCR.UrtScr	0.0	2.6	SS.SCS.CCS.Pom	2.3	7.7
	CR.HCR.Xfa.SpNemAdia	2.3	0.0			
	SS.SBR.PoR.SspiMx	4.5	0.0			
	SS.SCS.CCS	9.1	10.3			
	SS.SCS.CCS.Pom	2.3	7.7			
Biotope richness		14	9		8	5
n		44	39		44	39
Farne Islands	IR.MIR.KR.Lhyp.Ft	9.1	0.0	IR.MIR.KR.Lhyp	25.0	9.5
	IR.MIR.KR.Lhyp.GzFt	2.3	2.4	IR.MIR.KR.LhypT	4.5	4.8
	IR.MIR.KR.Lhyp.GzPk	4.5	4.8	IR.MIR.Ksed.DesFilR	0.0	2.4
	IR.MIR.KR.Lhyp.Pk	9.1	2.4	IR.MIR.Ksed.XKScrR	0.0	2.4
	IR.MIR.KR.LhypT.Ft	2.3	2.4	IR.MIR.KT.XKTX	0.0	2.4
	IR.MIR.KR.LhypT.Pk	2.3	2.4	IR.HIR.Ksed.LsacSac	0.0	2.4
	IR.MIR.Ksed.DesFilR	0.0	2.4	SS.SCS.ICS	0.0	2.4
	IR.MIR.Ksed.XKScrR	0.0	2.4	SS.SSA.IFiSa	2.3	2.4
	IR.MIR.KT.XKTX	0.0	2.4			
	IR.HIR.Ksed.LsacSac	0.0	2.4			
	SS.SCS.ICS	0.0	2.4			
	SS.SSA.IFiSa	2.3	2.4			
	CR.MCR.EcCr.FaAlCr	20.5	2.4	CR.MCR.EcCr.FaAlCr	59.1	59.5
	CR.MCR.EcCR.FaAlCr.Adig	9.1	16.7	SS.SCS.CCS	6.8	7.1
	CR.MCR.EcCr.FaAlCr.Pom	13.6	26.2	SS.SCS.CCS.PomB	2.3	4.8
	CR.MCR.EcCR.FaAlCr.Bri	15.9	14.3			
	SS.SCS.CCS	6.8	7.1			
	SS.SCS.CCS.PomB	2.3	4.8			
Biotope richness		14	9		8	5
n		44	39		44	39

4.3.2. Biotope frequency occurrence

Lhyp and FaAlCr biotopes and sub-biotopes dominated composition of all transects. These had similar frequencies of occurrence between sampling periods across all transects (Table 4.1). FaAlCr was the dominant biotope in all transects, in both survey periods and at both classification levels (Table 4.1). Biotope composition and frequency occurrence were similar between Dunstanburgh and Boulmer transects. Differences in biotope composition between years were driven by the low frequency occurrence of less dominant biotopes (< 10 in 100 samples) with many found uniquely in a single sampling year (Table 4.1). The Farne Islands transect differed slightly in biotope composition to both Boulmer and Dunstanburgh transects, with a greater diversity in infralittoral biotopes. However, similarly to other transects, Farne Island biotope composition was similar between years with differences in biotope richness between years driven by the low frequency occurrence of biotopes (< 3 in 100 samples) found uniquely in a single year (IR.MIR.Ksed.DesFilR, IR.MIR.Ksed.XKScrR, IR.MIR.KT.XKTX, IR.HIR.Ksed.LsacSac and SS.SCS.ICS).

There was no evident pattern in frequency occurrence of FaAlCr and Lhyp biotopes between years in any transects, with some increasing and others decreasing (Table 4.1). Classification to level 5 accentuated the dominance of Lhyp biotopes and FaAlCr biotopes for both survey periods across all transects. At level 5 other biotopes were less frequently recorded.

Individual biotope frequencies were not significantly different between survey years (treated as matched pairs) for sub-biotopes classified to level 6 or biotopes classified to level 5 for: Boulmer (Wilcoxon signed rank test, w=208, p=0.505 and w=69, p=0.554 respectively); Dunstanburgh (Wilcoxon signed rank test, w=258.5, p=0.556 and w=93.5, p=0.674 respectively); Farne Islands (Wilcoxon signed rank test, w=327.5, p=0.834 and w=108.5, p=0.674 respectively). As no changes in biotopes were observed between years fishing pressure as cause of biotope change was not investigated.

4.4. Discussion

4.4.1. Change in biotope composition and richness between years

Biotope richness varied slightly between years and between transects. Biotope composition (including frequency counts) was similar between years (highly non-significant Wilcoxon signed rank test) and between transects. Despite the uneven sampling effort between time periods (Table 4.1) the null-results were conservative as only uncommon biotopes with low frequency counts of less than 5 per 100 differed between time periods. These biotopes were not representative of the area as a whole and arguably could have been excluded from the analysis further reducing differences between years.

The number and range of biotopes was similar between transects with only minor differences between the Farne Islands and both Boulmer and Dunstanburgh transects. This difference was expected, as the Northern sites of the EMS have been described as particularly diverse due to being subject to low sedimentation and high water flow (Birkett *et al.*, 1998; Hartnoll, 1998). The non-significant fluctuations in frequency counts of biotopes between years can be attributed to natural variability. Long-term studies from the Swedish west coast have shown regular annual and seasonal fluctuations in circalittoral and infralittoral biotopes (Lundälv and Christie, 1986). In general, the number and range of biotopes was maintained between years through the persistence of a few biotopes, particularly infralittoral kelp (Lhyp) and circalittoral faunal and algal crust (FaAlCr) biotopes (rejection of hypothesis 1). As no changes in biotopes were observed between years fishing pressure as cause of biotope change was not investigated (rejection of hypothesis 2).

Both the maintenance of the number and range of biotopes are deemed important when investigating environmental change (Saunders *et al.*, 2011). However, one of the primary uses for the biotope classification system (in addition to providing a consistent description of habitat types), is mapping geographical distribution of biotopes to aid spatial management of marine ecosystems (Connor *et al.*, 2004; Crowder and Norse, 2008; Salomidi *et al.*, 2012). Spatial representations of biotopes are particularly useful as they may show changes that otherwise are not detected using biotope frequency alone. This is particularly important with regards to investigating changes induced by spatially variable anthropogenic impacts such as fishing (Diesing *et al.*, 2009), but a lack of acoustic data for large areas of the EMS meant that the geographical distribution of biotopes could not be mapped.

The investigation of biotope richness and persistence of dominant biotopes between years is a simple analysis both in terms of the data required and in the analysis of the data. However, in this particular case, conclusions that can be drawn from this data are limited due to the broad nature of biotope analysis and the low number of sampling years (2002/03 and 2011). These issues are further discussed in the following sections.

4.4.2. Sampling effort for biotope richness

Biotope richness increased rapidly as number of samples increased following a negative exponential curve until a plateau was reached (Fig 4.2, a and b). Sampling effort curves differed as expected between transects with differing biotope richness. For the Boulmer transect, with relatively low biotope richness (max 11) this rapidly increased up to 10 samples and started to plateau at approximately 25 samples (Fig 4.2, a). In contrast, the Farne Island transect with high biotope richness (max 17) increased more slowly at lower sampling efforts and did not fully plateau even at the maximum number of samples (n = 40) (Fig 4.2, b).

Theoretical biotope richness (samples 40 – 60) for the Boulmer transects (Fig 4.2, a) showed that there was no predicted increase in biotope richness with increased sampling effort. Thus the sampling protocol employed for the Boulmer transect adequately estimated the number of biotopes in the area in 2002. Fewer samples would have been sufficient for the task; only 25 compared to the 42 samples collected were needed. However, the theoretical biotope richness (samples 40 – 60) for the Farne Island transects (Fig 4.2, b) showed that estimates of biotope richness may increase with increased sampling effort. This suggests that the sampling undertaken for this study may not have adequately sampled the maximum number of biotopes in the area in 2011.

Although biotope richness is an important metric used to determine ecological health of an area (JNCC, 2013), the frequency of occurrence of biotopes is also important; it permits investigation of the persistence of dominating community assemblages (Saunders *et al.,* 2009). Depending on the nature of work and the time and funds available it may not be necessary to find the maximum biotope richness of the area. For example, previous monitoring reports from BNNC EMS did not aim to maintain specific biotope richness but rather to maintain the persistence of dominant communities (Mercer et al., 2003; Mercer 2012). The increased sampling that may be necessary in order to sample the maximum biotope richness may be costly and add little value to the study as high temporal and spatial sampling effort would be necessary to distinguish between natural variance in biotope composition and variance due to anthropogenic impacts. There are evidently no published condition monitoring programmes that specifically use maximum or maintenance of biotope richness as evidence of change, although this is mentioned as one of several recommended methods for condition monitoring (Davies *et al.*, 2001; Saunders *et al.*, 2011). However, as monitoring methods change and classification systems evolve biotope richness may be the only data that can be accurately extracted from historical records.

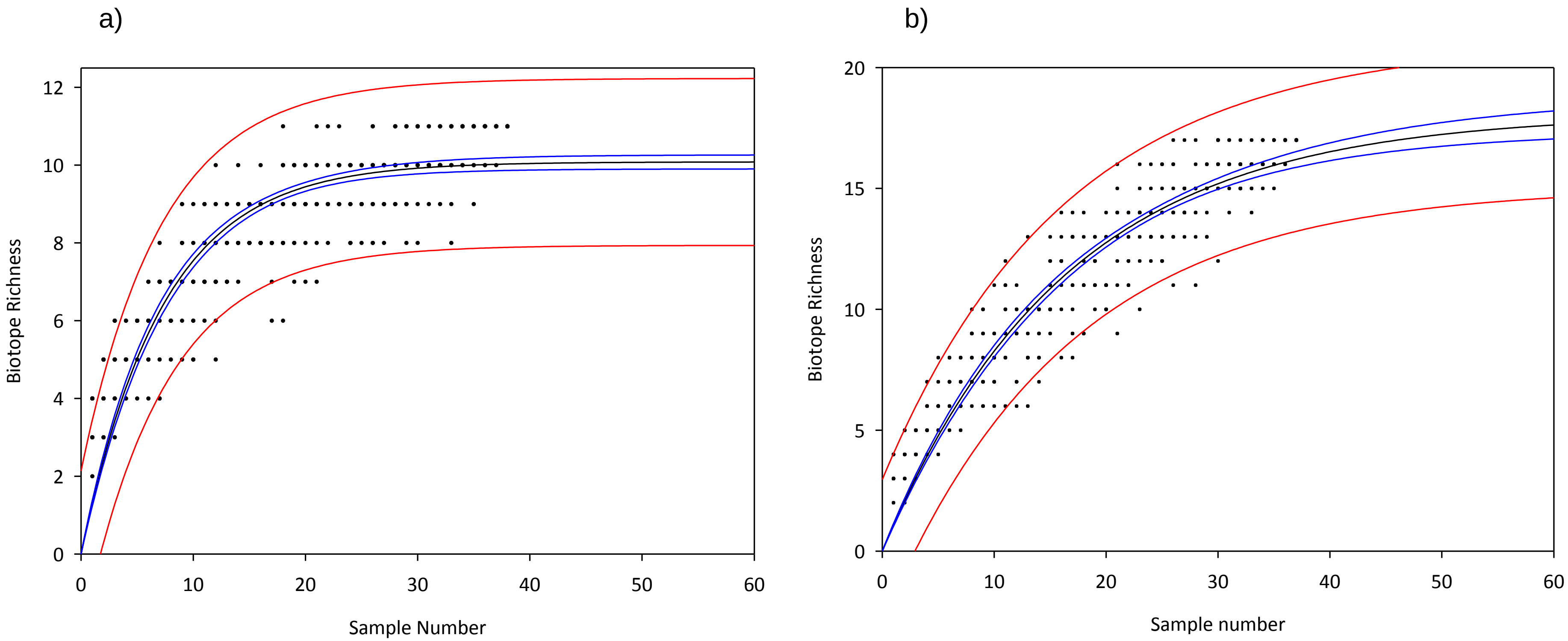


Fig 4.2. a) Boulmer transects (2002) and b) Farne Island transect (2011) biotope richness sampling effort curve (black line) 95% prediction band shown in red band and 95% confidence band shown in blue.

4.4.3. Scale of biotope identification

Potting impacts on benthic communities are sparsely documented. The few studies that exist have focused on impacts on a small group of locally important species perceived as fragile (Coleman *et al.,* 2013). No studies have looked at the impact of potting on communities and habitats as a whole. However, ecological monitoring, using biotopes is often undertaken at this broader scale, resulting in discrepancies between conclusions from peer-reviewed literature and those from monitoring and surveillance reports (Olenin and Ducrotoy, 2006). This section will critically appraise the use of hierarchical classification systems for use in robust scientific investigation.

The UK biotope classification has the advantage of integrating the surrounding abiotic conditions within the classification system. However, hierarchical classifications may also pose problems, specifically with regards to scale. The appropriate identification level used for ecological work will depend on the aim of that work and the sampling methods used (Connor *et al.,* 2004). For example, broad scale maps for management of SACs on a national scale may only need ‘main habitats’ (EUNIS level 3), for example the CQSM MCZ (Fitzsimmons *et al.*, 2015). Whereas biological descriptions of a smaller area, such as single transects, may need to use ‘sub-biotopes’ (level 6). Level 5 (biotope) and level 6 (sub-biotope) scales were used for this analysis. Scale and accuracy of habitat maps for management is critically appraised in chapter 3.

Working with level 5 data requires much less effort than level 6 both in the field and during the identification process (Holt and Sanderson, 2001); it is consequently cheaper and less time consuming than level 6 (Moore and Bunker, 2001). However, a large difference in recorded biotope richness occurred between these two scales, as the multitude of Lamanaria biotopes (6 sub-biotopes) and faunal and algal crust biotopes (6 sub-biotopes) fall into only 2 groups (IR.MIR.KR.Lhyp and CR.MCR.EcCr.FaAlCr, respectively) (Table 4.1). Further, the reduction in biological information means that impacts may not be detected (see chapter 6). Unless very large-scale changes are expected, then increased biological information is usually necessary to detect change (Davies *et al.*, 2001).

Fraschetti *et al.* (2008; 2011) concluded that marine community classification systems in Europe over the past 50 years are either too vague, focussing primarily on broad scale geological features, or too detailed, too fine scale definition of biodiversity at species levels, which is often biased towards the plant component in the infralittoral. In particular, Fraschetti *et al.* (2008) warn against overcomplicating classification levels, stating that ‘species intra-biotope natural variability can in fact cause a misleading qualitative interpretation of small-scale biodiversity’ (Fraschetti *et al.*, 2008). Only some habitats have conspicuous dominant species (e.g. kelp forests, mussel beds, maerl beds), but many support a mosaic of less dominant species (Connor, 1995). This fine-scale spatial variation (patchiness) can significantly vary over time. Infralittoral areas tend to be more predictable whereas circalittoral rock tends to be a mosaic of different species patches (Hartnoll, 1998), but little information is available on natural successions of community assemblages in the North Sea (this is further discussed in chapter 3). Results suggest that classification at the biotope scale (level 5) may be most appropriate for research purposes, as poorly understood natural changes in sub-biotopes (level 6) may result in increased variation and may prevent detection of impacts. Sub-biotope patchiness resulted in several levels of the classification system being used simultaneously in this research. Direct comparison of biotopes that are linked, for example: CR.MCR.EcCR.FaAlCr (biotope) and CR.MCR.EcCR.FaAlCr.Flu (sub-biotope belonging to the latter) may not be adequate. However, the nature of the classification system makes this very difficult to avoid.

There are often no distinct boundaries between biotopes. Gradual transitions between two biotopes mean that classification may have variable outcomes due to observer bias (Connor *et al*., 2004). Additionally, some communities may be temporary or transitional and represent a stage between two or more ‘stable’ biotopes (Sutherland, 1974). This could be due to periodic abiotic or biotic disturbance e.g. winter storms or increased grazing during summer. In addition, certain habitats, for example FaAlCr sub-biotopes, may be so variable that the position of a biotope along a gradient cannot be accurately defined (Connor, 1995). In most rocky reef habitats, substratum space is fully occupied, and the availability of space is a controlling resource. Depending on availability of free space and which species are recruiting, different species assemblages can develop under the same physicochemical conditions (Sebens, 1985). In this research, when sampling boundaries between biotopes or biotopes in a state of transition, biotopes were conservatively identified to a lower level. For example, distinction between CR.MCR.EcCR.FaAlCr.Flu and CR.MCR.EcCR.FaAlCr.Adig where neither *Flustra* nor *Alcyonium* dominated, were both classified at a lower level

(CR.MCR.EcCR.FaAlCr). Although this is a conservative approach, the loss of accuracy (being unable to identify samples at a higher biotope level), may make detection of impacts difficult.

Conclusions from this work are similar when using either biotopes (level 5) or sub-biotopes (level 6) with no change detected between years. However, the use of biotope data for low-level fishing impact studies remains problematic and a null result in this case may be due to methodological issues rather than a lack of change. Use of a broader biological classification (e.g. level 5) is only likely to detect larger impacts which is less useful when investigating small-scale impacts of static gears, although results indicating change may be more robust as small-scale natural variability is already largely accounted for. In contrast, the use of a more detailed biological classification (level 6) may result in detection of smaller scale impacts, but much larger data sets are required to allow for natural spatial and temporal variability as little information exists for UK habitats (Hartnoll, 1998). For the investigation of small-scale fishing impacts, such as potting, the use of a biotope classification system alone is not recommended. However, a biotope based approach may provide a useful overview and help identify particular habitats or species as worthy of further research.

4.5. Concluding remarks

Biotopes cannot be used in all ecological studies and the limitations of the classification system must be understood. There are issues related to simplifying multivariate data sets into univariate data sets (further discussed in chapter 5) resulting in a loss of detail in the data which therefore increases variability, in turn affecting the detection of impacts on species (Clarke and Warwick, 2001).

Variation of species within biotopes and frequency of biotope occurrence are expected under natural marine environmental conditions (Fraschetti *et al.*, 2008). At two scales (transect level and area wide) biotopes have scarcely changed, however, this is a simplistic method for investigating ecological health of an area; it does not allow changes in species abundances, diversity or composition for each biotope to be taken into account. Ecological quality of biotopes was not assessed using this method. The maintenance of quality of features of interest is likely to be a critical aspect of future monitoring programmes as part of Defra’s revised approach to fisheries management in EMSs. However, this has not been considered in condition monitoring to date. Biotope classifications are relevant for examining geographical and distributional differences between years (i.e. chapter 3), but in order to investigate biotope health, further work is required on species diversity and abundance levels (chapter 5) which may be a more sensitive indicator of change, enabling small-scale impacts to be detected.

Chapter 5 . Long-Term Potting Intensity Effects on Species Composition and Richness.

5.1. Introduction

The recent drive to maintain global biodiversity through protection and preservation of the marine environment has resulted in an increased need for data on impacts of human activities in the marine environment (Heslenfeld and Enserink, 2008). The Berwickshire North Northumberland Coast (BNNC) Special Area of Conservation (SAC), part of the BNNC European Marine Site (EMS) has been repeatedly surveyed as part of regular SAC condition monitoring (Mercer *et al.*, 2003; Mercer, 2012), and long-term habitat mapping and monitoring (Edwards, 1983; Foster-Smith and Foster-Smith, 1987; Connor, 1989; Holt, 1994; Brazier *et al.*, 1998). However, these surveys were designed to report on habitats and biotopes in the area over time from before designation of the SAC, and as part of condition monitoring, rather than a focused study on impacts affecting habitats and biotopes. They may not provide appropriate data to support the evidence-needs required as part of Defra’s revised approach to commercial fishing within European Marine Sites (EMSs) (Chapter 4). In line with the UK’s agreement to adhere to a precautionary approach (EU Habitats Directive, Council Directive 92/43/EEC), current fishing activities in the BNNC EMS will only be allowed to continue if empirical evidence can show that these fishing activities do not adversely impact SAC and SPA features (MMO, 2012). As part of Defra’s “revised approach” interactions of fishing gear with marine features of SACs and SPAs were assessed on a Red (highest risk), Amber, Green and Blue (lowest risk) scale of likelihood of damage to the feature to determine types of management of activities required (MMO, 2012). Northumberland IFCA introduced a byelaw prohibiting the use of mobile fishing gear within the English section of the Berwickshire & North Northumberland Coast Special Area of Conservation (SAC) on 1 January 2014, as there was evidence that mobile fishing gear were likely to impact Annex 1 rocky reef habitats (NIFCA, 2014).

Less destructive fishing methods such as potting will also require monitoring and investigation of impacts. Currently colour coded in amber in the Draft Fisheries in EMS Populated Matrix (2013), impacts are poorly understood at present and supporting evidence is lacking (Eno *et al.,* 2001; Coleman *et al.,* 2013). Previous chapters found no evidence of changes in biotope composition attributable to potting impacts, or between years, using a conventional condition monitoring biotope based approach (Moore and Bunker, 2001; Saunders *et al.*, 2011). Methods for detection of lower-level impacts from potting must be improved as part of Defra's "revised approach". Previously discussed methods include analyses of indicator / sensitive species (Eno *et al*., 2001; Shester and Michelli, 2011, Hiscock and Tyler-Walters, 2006) and community composition (Coleman *et al.,* 2013).

Analyses using indicator species compare change over time in particular species that are thought to be vulnerable to impacts (Shester and Michelli, 2011). In the case of potting, the focus would be on displacement, abrasion and physical disturbance from direct interaction with pots (Eno *et al.*, 2001), in which case, sessile, erect and brittle species are thought to be the most appropriate indicators of change (Eno *et al*., 2013). However, these species lists are often only available and valid for small geographical areas (Coleman *et al*., 2013) and selection is based on expert opinion rather than observed potting impacts (Eno *et al*., 2013).

Analyses using community assemblages compare all the species identified and their proportions between sites (Coleman *et al*., 2013). Exact number of species, level of species identification and accuracy of abundance will depend on the method used to collect the data (see Chapter 1, for review of methodologies). For measurement of change due to potting impacts, the use of community assemblages over indicator species may be beneficial since species sensitivity to the relatively low degree of force associated with static gear is scarcely known (Coleman *et al*, 2013). Considering the whole assemblage may allow impacts on unexpected species to be detected (Clarke and Warwick, 2001).

This chapter seeks evidence of finer scale potting impacts on benthic features of interest. Species compositions, indicator species and species richness are reanalysed from monitoring data gathered in 2002 and 2012. This has the potential to facilitate the development of protocols needed for the detection of small-scale impacts, which would otherwise require extensive costly fieldwork. Use of this long-term monitoring data may provide a unique opportunity to investigate whether ecological condition monitoring data, analysed in novel ways, is sufficiently accurate and precise for robust scientific investigations. To achieve this, several metrics for the detection of small-scale impacts were further explored.

This chapter aimed to investigate changes in benthic community composition within biotopes in the BNNC EMS between two sampling periods and whether relationships existed between biotic changes and potting pressure. Drop-down video data collected for regular condition monitoring (2002/03 and 2011) were again employed (chapter 4). It was hypothesized that changes undetectable using biotope classification alone could be detected using species presence/absence data (hypothesis 1). This was investigated by analysing change of: taxonomic composition of assemblages, species diversity and ecologically important species between years in three transects (Fig 5.1). Community assemblage data from transects were pooled and change across the whole BNNC EMS area explored. It was also hypothesized that relationships existed between biotic changes and potting pressure (hypothesis 2). This was tested by investigating the effects of potting pressure (chapter 2) on the change in benthic community structure of individual biotopes across the EMS between years (2002/03 – 2011).

5.2. Methods

5.2.1. Sampling protocol

Methods for site selection, videographic sampling and selection of video data are detailed in chapter 4 (sections 4.2.1 and 4.2.2 respectively). In summary, within each of three transects corridors identified by Natural England, three depth ranges were selected and up to fifteen 100m tows of a drop-down video camera recorded in each (Fig 5.1). Footage from each tow was randomly sampled every ~10 – 30 seconds, generating 1724 discrete samples (2-second clips) from 189 video tows. Species present were recorded to the lowest taxonomic level using identification manuals (e.g. Cornelius (1995); Hayward and Ryland (1995); Foster-Smith and Foster-Smith (2000)) and online resources (MarLIN). Expert advice was sought where necessary (J Foster-Smith, pers comm. 2013). Lack of scale on the drop-down camera precluded collection of abundance data from the footage, so species presence/absence data were used exclusively to describe communities. A biotope class (minimum level 5) was assigned to each sample (Connor *et al.,* 2004) following the methodology outlined in chapter 4. This was taken to encapsulate the relevant range of abiotic factors, and the biotope classification was used instead of individual abiotic factors in the analysis.

Although not integrated into the classification system, water visibility may affect the species that are identifiable from the videographic imagery (Mercer *et al.*, 2003). Low visibility may result in smaller or cryptic species being recorded less frequently in video footage or not at all (Moore and Bunker, 2001). The influence of visibility was tested using subjective scales (1 – 3) of water turbidity (where 1 was clear and 3 was highly turbid), light levels (where 1 was high and 3 was low levels of light) and camera movement (where 1 was a stationary camera and 3 was ≥ 2m camera travel). This 27 point scale of visibility (turbidity x light levels x distance travelled by the camera) was then categorised into 6 levels for ease of analysis. Underwater visibility enhancing hardware (Lynn-hawk) was used to filter out turbidity. Although footage with and without the visibility enhancing hardware did not differ in number of species identified, it was subjectively felt by observers to contribute to the ‘ease’ of species identification, therefore increasing efficiency and potentially reducing identification error.

To assess the ecological stability of individual biotopes, analysis of change between years (2002/03 and 2011) using full taxonomic composition, sensitive species and species richness of biotopes was undertaken at individual transect level. Transect data were then pooled and BNNC EMS area-wide change was investigated. Area-wide data also allowed fishing pressure to be investigated as a factor affecting change between 2002/03 and 2012.

5.2.2. Taxonomic composition of assemblages

Exploratory statistics including multivariate analysis (cluster dendograms, multi-dimensional scaling (MDS) plots and SIMPROF) and SIMPER were conducted using PRIMER (v.6). The Jaccard similarity measure was used for species presence/absence data.

Change in species presence/absence within individual transect corridors between 2002/03 - 2011 was investigated using a mixed model in PERMANOVA (v. 1.0.5) (type III sum of squares, under a reduced model with 9999 permutations) following a 3 factor design with interaction (visibility as a random factor with 6 levels, biotope as a random factor with 11 levels and year as a fixed factor: 2002 and 2011). Interaction terms ‘visibility * year’ and ‘visibility * biotope * year’ were removed from the model by pooling as they had negative estimates of components’ variations for all transect corridors (Anderson *et al.*, 2008). Biotopes on soft sediments rely on infauna for identification and were excluded from the analysis as the video footage did not provide enough biological information. Tests of homogeneity of dispersion (PERMDISP routine in PERMANOVA) were used to test the null hypothesis of no difference in dispersion among *a priori* groups. Confounding effects of dispersion among groups were investigated using principal coordinate (PCO) analysis of the centroids for presence/absence survey data grouped according to biotope and year in PERMANOVA (v. 1.0.5).

Change in species presence/absence across all transect corridors (BNNC EMS area-wide data) between years was investigated using a mixed model in PERMANOVA (v. 1.0.5) (type III sum of squares, under a reduced model with 9999 permutations) following a 3 factor design with interaction (visibility as a random factor with 6 levels, biotope as a random factor with 20 levels and year as a fixed factor: 2002 and 2011). As with individual transects, soft sediment biotopes were excluded from the analysis, tests of homogeneity of dispersion (PERMDISP routine in PERMANOVA) were used to test the null hypothesis of no difference in dispersion among *a priori* groups and confounding effects of dispersion among groups were investigated using principal coordinate analysis (PCO).

Simper analysis (PRIMER, v.6) was undertaken for both transect level and BNNC EMS area-wide analysis to compare each species' presence/absence between years (SIMPER on species grouped by biotope with factor year).

5.2.3. Fishing pressure

Northumberland potting effort distribution (chapter 2) was categorised into 2 levels for use in a species presence/absence mixed model (low = 0 – 226 and high = 227 – 770 pots month^{-1} km^{-2}) using the natural breaks (jenks) function in ARC ESRI GIS (v. 10.0) (Fig 5.1). Since video data were obtained during 2 seasons (summer and autumn), annual fishing pressure was used to remove seasonal effects of changing fishing effort (Fig 5.1).

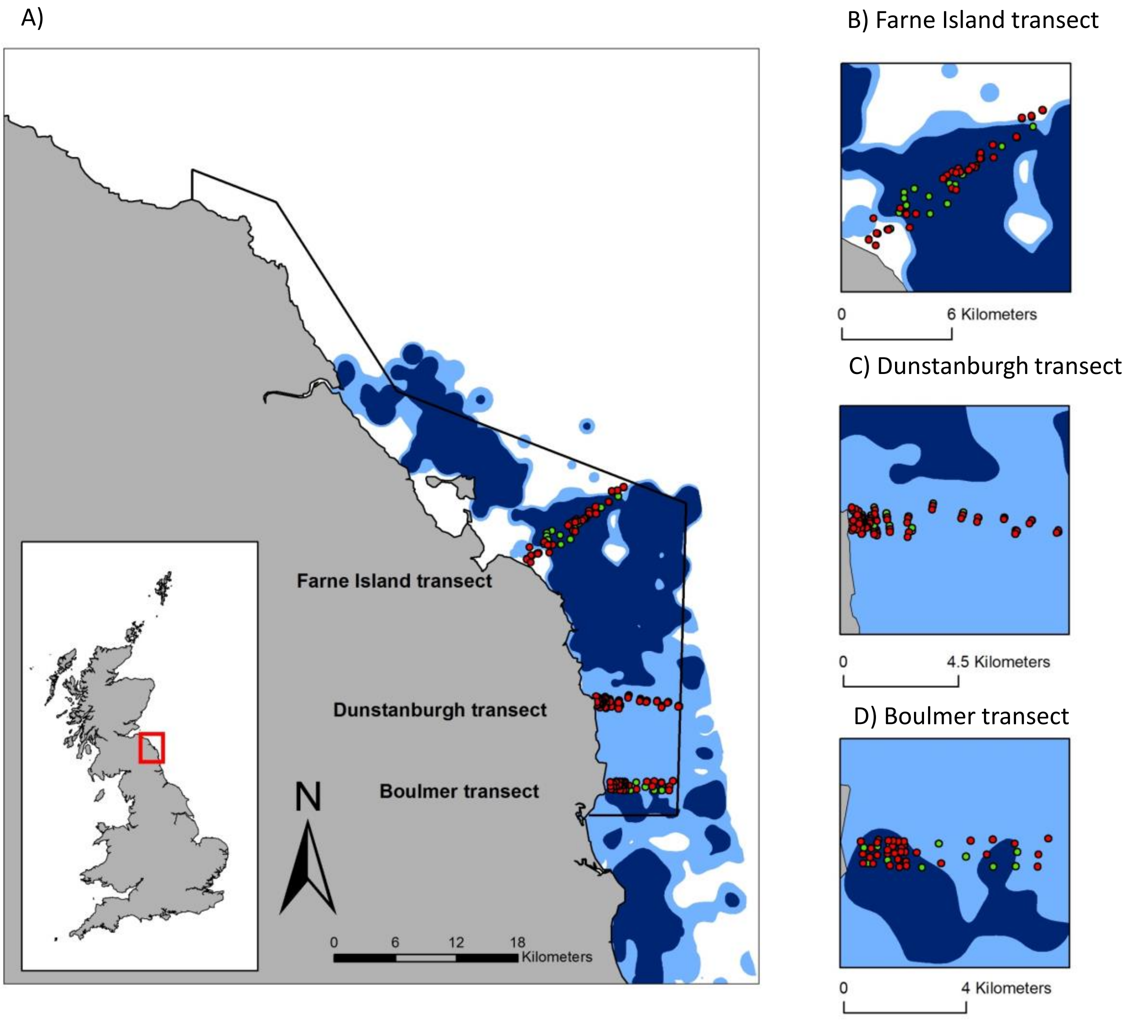


Fig 5.1. Drop-down camera sites at the Farne Island (B), Dunstanburgh (C) and Boulmer (D) transect corridors within the Berwickshire and North Northumberland Coast (A) European Marine Site (black line) in 2003 (green points) and 2011 (red crosses)). Potting intensity categorised into two levels (low, light blue; high, dark blue; No data, white).

It was not possible to test the effect of potting pressure on species presence/absence between years on individual transects due to low biotope numbers in each fishing pressure. For example, across the Dunstanburgh transect, biotopes of interest were only present in areas of low potting intensity and therefore could not be compared to the same biotopes in areas of high fishing pressure within the same transect (Fig 5.1). Therefore, the effect of potting pressure on species presence/absence between years was investigated using BNNC EMS area-wide data, with a mixed model in PERMANOVA (type III sum of squares, under a reduced model with 999 permutations) following a 4 factor design with interaction (fishing pressure as a random factor with 2 levels, visibility as a random factor with 6 levels, biotope as a random factor with 20 levels and years as a fixed factor with 2 levels: 2002 and 2011). Interaction terms ‘Year * Fishing’, ‘Visibility * Fishing’ and ‘Visibility * Year *

Fishing' were removed from the model by pooling as they had negative estimates of components variations. PERMDISP was also used to test the null hypothesis of no difference in dispersion among *a priori* groups. Confounding effects of dispersion among groups was investigated using PCO of the centroids for presence/absence survey data grouped according to biotope and year in PERMANOVA (v. 1.0.5).

5.2.4. Species diversity

Species richness (*S*) per discrete video sample was chosen as the diversity metric, due to the lack of abundance data (Clarke and Warwick, 2001). Data were not normally distributed (even following Log transformation), so were analysed using semi-parametric techniques in PERMANOVA (v. 1.0.5). A Bray-Curtis resemblance matrix of untransformed *S* values was used in a mixed model. Differences in *S*, grouped by biotope, between years and fishing pressure were tested using PERMANOVA (v. 1.0.5) (type III sum of squares, under a reduced model with 9999 permutations) following a 3 factor design with interaction (biotope as a random factor with 11 levels, year as a fixed factor: 2002 and 2011 and fishing pressure as a random factor with 2 levels: low = 0 – 240 and high = 241 – 770 pots km^{-2} year^{-1}). Factor 'Visibility' was excluded from the analysis as this factor and all its interaction terms were non-significant ($p > 0.15$) with negative or very low estimates of components of variation.

5.2.5. Sensitive species

Information on important species of biotopes in transects were obtained from the Marine Life Information Network website (MarLIN, 2012). Change or loss of these species can be taken to indicate deterioration of biotope health (Tyler-Walters *et al.*, 2001). Descriptions of species roles and selection criteria are described in Table 5.1. Important species and descriptions of their roles for each biotope are provided in Appendix 3. Differences in presence/absence of chosen important species, grouped by biotope, between years was tested using PERMANOVA (v. 1.0.5) (type III sum of squares, under a reduced model with 9999 permutations) following a 2 factor design with interaction (biotope as a random factor with 11 levels and year as a fixed factor; 2002 and 2011). However, results using a reduced species list were similar to those using the full species list (described in section 5.2.2) therefore further analyses using only sensitive species were not undertaken.

Table 5.1. Categorisation of species indicative of biotope sensitivity (MarLIN, 2012)

Rank	Selection criteria
Key structural	The species provides a distinct habitat that supports an associated community. Loss/degradation of this species population would result in loss/degradation of the associated community.
Key functional	The species maintains community structure and function through interactions with other members of that community (for example, predation, grazing, competition). Loss/degradation of this species population would result in rapid, cascading changes in the community.
Important characterizing	The species is/are characteristic of the biotope (dominant, highly faithful and frequent) and are important for the classification of that biotope. Loss/degradation of these species populations could result in loss of that biotope.
Important structural	The species positively interacts with the key or characterizing species and is important for their viability. Loss/degradation of these species would likely reduce the viability of the key or characterizing species. For example, these species may prey on parasites, epiphytes or disease organisms of the key or characterizing species.
Important functional	The species is/are the dominant source of organic matter or primary production within the ecosystem. Loss/ degradation of these species could result in changes in the community function and structure.
Important other	Additional species that do not fall under the above criteria but where present knowledge of the ecology of the community suggests they may affect the sensitivity of the community.

5.3. Results

5.3.1. Taxonomic composition of assemblages

Non-metric multi-dimensional scaling plots showed species presence/absence survey data, grouped according to biotope, for three transects (Farne Island, Dunstanburgh and Boulmer) (Fig 5.2). Species presences differed between circalittoral and infralittoral rock on all transects (Fig 5.2). Species show grouping by biotope, although there was variability within all groups. Biotope groupings showed less variability for the Farne Island transect data (including lower stress values, Fig 5.2, A) than the Dunstanburgh (Fig 5.2, B) and Boulmer (Fig 5.2, C) transect data. In all locations the biotope FaAlCr was particularly variable.

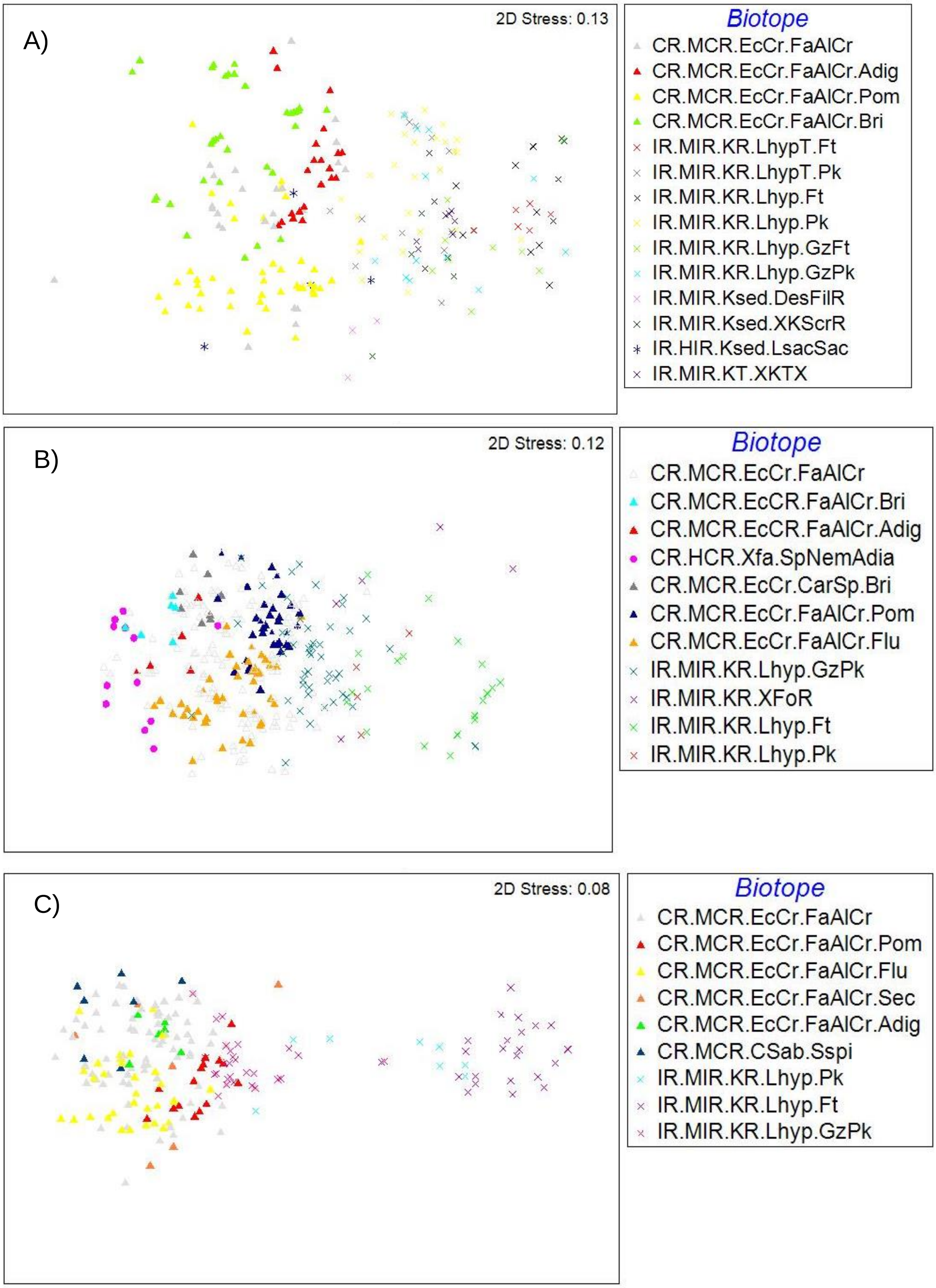


Fig 5.2. Two dimensional non-metric multi-dimensional scaling plot of species presence/absence survey data grouped according to biotope (A) Farne Island (2D stress = 0.13); B) Dunstanburgh (2D stress = 0.12); C) Boulmer (2D stress = 0.08)). Triangles represent circalittoral rock with medium tide/wave exposure. Crosses represent infralittoral rock with medium tide/wave exposure. Circles represent circalittoral rock with high tide/wave exposure.

5.3.2. Dunstanburgh taxonomic composition of assemblages

For the Dunstanburgh transect (Fig 5.2, C) all factors and interaction terms in the model were significant except year (Table 5.2). Factor ‘Biotope’ explaining the most variation, with others explaining much less variation (Table 5.2). The overall proportion of variation explained by the model was approximately 40%.

Table 5.2. PERMANOVA results with estimates of components variation for the Dunstanburgh transect. Interaction terms ‘Visibility * Biotope * Year’ and ‘Visibility * Year’ were not included in the model (pooled) due to negative estimates of components of variation.

Terms	**P (perm)**	**Estimates of components of variation**	**Percentage contribution to total variation (%)**
Visibility	0.0307	99.787	2.34
Biotope	0.0001	1283.70	30.06
Year	0.077	122.35	2.87
Visibility * Biotope	0.0004	104.18	2.44
Biotope * Year	0.0001	174.87	4.10
Residuals		2484.90	58.20

Post-hoc testing of ‘Biotope * Year’ showed that species assemblages grouped by biotope differed between years (all, $p < 0.015$) except for biotope CR.MCR.EcCR.FaAlCr.Bri, $p = 0.115$ (Table 5.3). Only 6 biotopes were present in both 2003 and 2011 for comparison: FaAlCr, FaAlCr.Bri, FaAlCr.Pom, FaAlCr.flu, Lhyp.GzPk, Lhyp.Ft (Table 5.3). However, pairwise comparison of ‘Biotope * Year’ using PERMDISP were significant for all biotopes except for IR.MIR.KR.Lhyp.Ft ($p = 0.14$) and IR.MIR.KR.Lhyp.GzPk ($p = 0.666$) suggesting dispersion effects may be confounding the PERMANOVA results (Table 5.3).

Table 5.3. Pairwise comparisons for PERMANOVA and PERMDISP routines of ‘Biotope * Year’ for factor year. Dunstanburgh transect.

Biotope	**PERMANOVA result (indicates potential location effect)**	**PERMDISP result (indicates dispersion effect)**
CR.MCR.EcCr.FaAlCr	Significant	Significant
CR.MCR.EcCR.FaAlCr.Bri	Not Significant	Significant
CR.MCR.EcCR.FaAlCr.Pom	Significant	Significant
CR.MCR.EcCR.FaAlCr.Flu	Significant	Significant
IR.MIR.KR.Lhyp.GzPk	Significant	Not Significant
IR.MIR.KR.Lhyp.Ft	Significant	Not Significant

A difference in species assemblages was detected between 2002 and 2011 within biotopes IR.MIR.KR.Lhyp.GzPk and IR.MIR.KR.Lhyp.Ft. This difference was not confounded by dispersion effects. It is unclear whether species differed between 2002 and 2011 within biotopes CR.MCR.EcCr.FaAlCr, CR.MCR.EcCR.FaAlCr.Pom and CR.MCR.EcCR.FaAlCr.Flu because dispersion effects were detected.

PCO showed centroid values for species composition grouped by biotope were not well separated between years except for the IR.MIR.KR.Lhyp.GzPk biotope (Fig 5.3), indicating that differences in species compositions (for biotopes CR.MCR.EcCr.FaAlCr, CR.MCR.EcCR.FaAlCr.Pom and CR.MCR.EcCR.FaAlCr.Flu) between years detected by the PERMANOVA routine were more likely to be dispersion and not location effects.

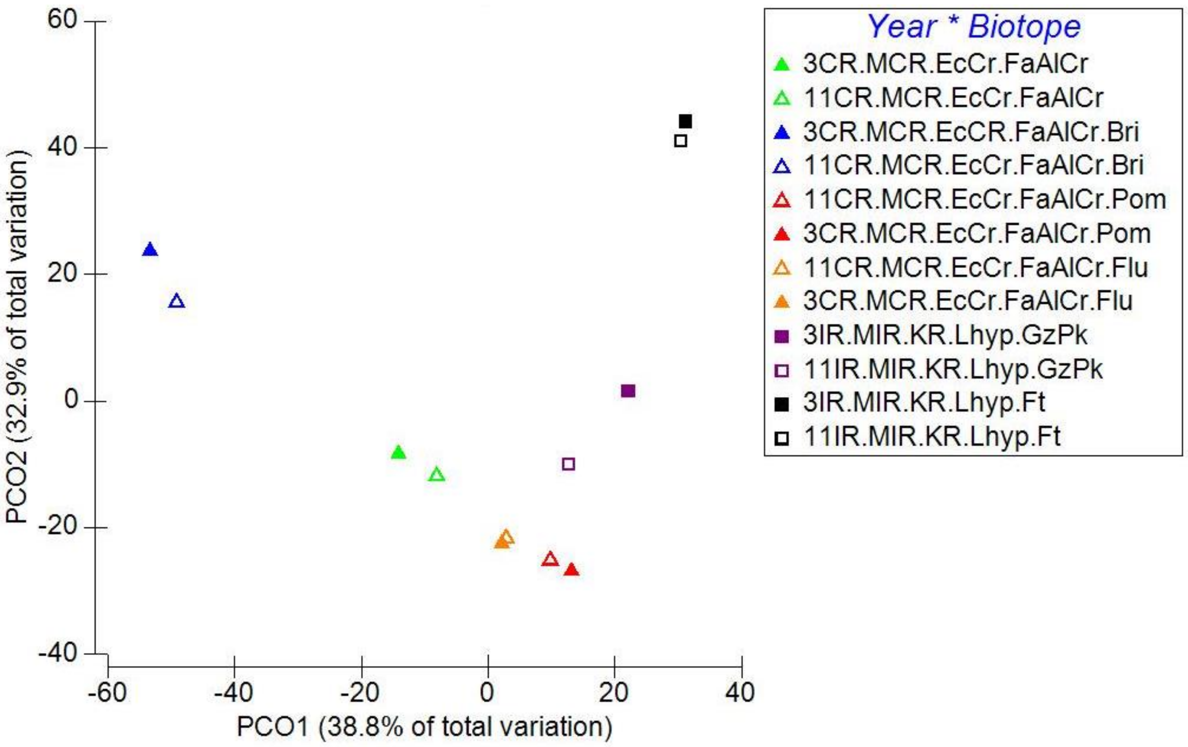


Fig 5.3. Principal coordinate analysis (PCO) of the centroids for presence/absence survey data grouped according to biotope and year for the Dunstanburgh transect. Circalittoral rock with medium tide exposure (triangles) and infralittoral rock with medium tide/wave exposure (squares) are grouped by year (2011 samples have a colour filling and 2002/03 samples lack a colour filling).

SIMPER analysis for biotopes that showed significant species change between years, unconfounded by dispersion effects (Table 5.3) are shown in Table 5.4 (IR.MIR.KR.Lhyp.Ft) and Table 5.5 (IR.MIR.KR.Lhyp.GzPk).

Average species composition similarity between samples of IR.MIR.KR.Lhyp.GzPk was 41.9% in 2002 (n = 56) and 18.1% in 2011 (n = 8) indicating that samples from 2011 were more variable in species composition than those in 2002. The average similarity of species composition between years was very low at 26.2%. Large differences between years were driven by *Echinus esculentus* and *Laminaria hyperborea* (29% cumulative contribution to dissimilarity between years (Table 5.4)). Smaller differences in average presence between years included smaller sessile species.

Table 5.4. Species contributing more than 5% towards the SIMPER dissimilarity (%) of IR.MIR.KR.Lhyp.GzPk between years at the Dunstanburgh transect.

Species	**2002/03**	**2011**			
	Average presence	**Average presence**	**Average dissimilarity**	**Percent contribution to dissimilarity**	**Cumulative percent contribution**
Echinus esculentus	0.77	0.38	10.57	14.32	14.32
Laminaria hyperborea	0.54	0	10.52	14.26	28.58
Balanus spp	0.57	0.5	8.96	12.14	40.71
Hydroid	0.43	0.25	7.66	10.38	51.09
Pomatoceros triqueter	0.36	0.25	7.57	10.26	61.35
Corallinaceae	0.23	0.38	7.19	9.74	71.08
Foliose red alga	0.09	0.13	3.68	5	76.07

Average species composition similarity between samples of IR.MIR.KR.Lhyp.Ft was higher (59.2% in 2002, n=22) (52% in 2011, n = 20) than IR.MIR.KR.Lhyp.GzPk. Average similarity of species composition between years was 51.8%. In IR.MIR.KR.Lhyp.Ft, 62% of the cumulative contribution difference between years were driven by differences in average presence of five epiphytes (*Membranoptera alata, Electra pilosa, Membranipora membranacea*, *Obelia* sp. and *Plocamium* sp.)(Table 5.5).

Table 5.5. Species contributing more than 5% towards the SIMPER dissimilarity (%) of IR.MIR.KR.Lhyp.Ft between years at the Dunstanburgh transect.

Species	**2002/03**	**2011**			
	Average presence	**Average presence**	**Average dissimilarity**	**Percent contribution to dissimilarity**	**Cumulative percent contribution**
Membranoptera alata	0.32	0.45	10.15	21.05	21.05
Electra pilosa	0.41	0.2	7.58	15.72	36.76
Plocamium spp	0.41	0	5.96	12.37	49.13
Membranipora membranacea	0.27	0	3.51	7.28	56.41
Obelia spp	0.14	0	3.03	6.29	62.7
Echinus esculentus	0.09	0.1	2.94	6.1	68.8
Ceramium spp	0	0.15	2.79	5.8	74.59
Delesseria sanguinea	0	0.15	2.45	5.09	79.68

5.3.3. Boulmer taxonomic composition of assemblages

For the Boulmer transect (Fig 5.2, D) not all factors and interaction terms were significant in the PERMANOVA mixed model (Table 5.6). Species composition for factor ‘Year’ and interaction term ‘Biotope * Year’ did not differ (p > 0.1). Factor ‘Biotope’ explained the most variation (Table 5.6), the overall amount of variation explained being ≈ 45%. Interaction term ‘Biotope * Year’ was non-significant (p = 0.124) making post-hoc testing unnecessary.

Table 5.6. PERMANOVA results with estimates of components variation for the Boulmer transect. Interaction terms ‘Visibility * Biotope * Year’ and ‘Visibility * Year’ were removed from the model by pooling as they had negative estimates of components variations.

Terms	**P (perm)**	**Estimates of components of variation**	**Percentage contribution to total variation (%)**
Visibility	0.0241	21.823	0.54
Biotope	0.0001	1536.6	38.36
Year	0.3667	22.998	0.57
Visibility * Biotope	0.0002	116.41	2.91
Biotope * Year	0.1239	126.06	3.15
Residuals		2181.4	54.46

5.3.4. Farne Islands taxonomic composition of assemblages

For the Farne Island transect (Fig 5.2, B) significant differences in species composition were only detected between the factor ‘Biotope’ and interaction term ‘Visibility * Biotope’, with factor ‘Biotope’ explaining the most variation, other factors explaining little (Table 5.7). Overall variation explained by the model (~60%) was highest for this transect. The interaction term ‘Biotope * Year’ was again non-significant (p = 0.1125), and no post-hoc testing was required.

Table 5.7. PERMANOVA results with estimates of components variation for Farne Islands transect. Interaction terms ‘Visibility * Biotope * Year’ and ‘Visibility * Year’ were removed from the model by pooling as they had negative estimates of components variations.

Terms	**P (perm)**	**Estimates of components of variation**	**Percentage contribution to total variation (%)**
Visibility	0.2403	25.23	0.72
Biotope	0.0001	1593.70	45.75
Year	0.4251	55.93	1.61
Visibility * Biotope	0.0001	232.50	6.67
Biotope * Year	0.1125	190.81	5.48
Residuals		1385.70	39.77

5.3.5. BNNC EMS area-wide taxonomic composition of assemblages

Grouping of species by factor ‘Biotope’ (Fig 5.4) and a clear division between circalittoral and infralittoral classifications was observed (Fig 5.4). Groupings of samples by biotope were more variable than in any individual transect corridor (Fig 5.2), suggesting spatial-scale effects within biotopes, particularly CR.MCR.EcCr.FaAlCr, IR.MIR.KR.Lhyp.GzPk and CR.MCR.EcCr.FaAlCr. Flu.

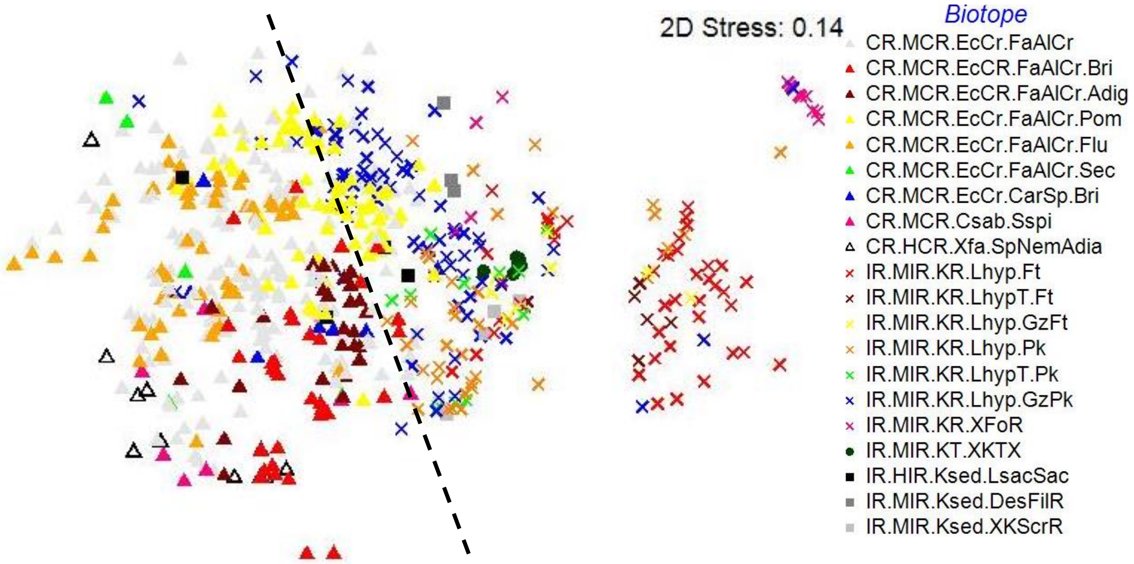


Fig 5.4. Two dimensional non-metric multi-dimensional scaling plot of species presence/absence survey data grouped according to biotope (all transects). Triangles represent Circalittoral rock with medium tide exposure. Crosses represent infralittoral rock with medium tide/wave exposure. Squares represent infralittoral rock with high sediment loads. Dashed black line shows approximate separation of infralittoral and circalittoral habitats. 2D stress = 0.14.

Factor ‘Biotope’ explained the most variation (Table 5.8) and overall variation explained by the model was ≈ 43%, lower than for Boulmer and Farne Islands transect data. The interaction term ‘Biotope * Year’ was non-significant (p = 0.106) therefore no further post-hoc testing was undertaken. The estimates of components of variation for factors included in the overall model (Table 5.8) were positive and much greater than zero, suggesting that the non-significant p-values were robust (Anderson *et al.,* 2008).

Table 5.8. PERMANOVA results with estimates of components variation for all transects (Farne Island, Dunstanburgh and Boulmer). Interaction terms ‘Visibility * Biotope * Year’ and ‘Visibility * Year’ were removed from the model by pooling as they had negative estimates of components variations.

Terms	**P (perm)**	**Estimates of components of variation**	**Percentage contribution to total variation (%)**
Visibility	0.121	28.5	0.8
Biotope	0.001	1347.3	35.6
Year	0.037	30.2	0.8
Biotope * Year	0.106	52.6	1.4
Visibility * Biotope	0.001	142.9	3.8
Residuals		2178.5	57.6

5.3.6. *Change in taxonomic composition of assemblages with fishing intensity*

Potential effects of fishing pressure on taxonomic composition of assemblages were investigated using the BNNC EMS area-wide data. In addition, this model allowed investigation into whether the addition of factor 'Fishing pressure' explained a higher amount of variation (Table 5.9).

The null hypothesis that fishing pressure does not affect species composition between years was investigated and only results that are directly relevant to the investigation of that hypothesis are shown in Table 5.9. No other interaction terms ('Visibility * Biotope', 'Visibility * Year', 'Biotope * Fishing', 'Visibility * Biotope * Year' and 'Visibility * Year * Biotope * Fishing') were significant (all p > 0.14).

Table 5.9. PERMANOVA results with estimates of components variation for all transects (Farne Island, Dunstanburgh and Boulmer) including fishing pressure. Interaction terms 'Year * Fishing', 'Visibility * Fishing' and 'Visibility * Year * Fishing' were removed from the model by pooling as they had negative estimates of components variations.

Terms	**P (perm)**	**Estimates of components of variation**	**Percentage contribution to total variation (%)**
Visibility	0.127	34.7	0.865164
Biotope	0.001	1234.1	30.76942
Year	0.14	110.0	2.742595
Fishing pressure	0.216	24.3	0.605864
Biotope * Year	0.917	229.7	5.727037
Biotope * Year * Fishing pressure	0.142	310.7	7.746584
Residuals		2067.3	51.54333

P-values and variation explained by the factors 'Visibility', 'Biotope', and 'Year' differed from those of individual transects and BNNC EMS area-wide models. 'Biotope' still explained the most variation while factor 'Fishing pressure' explained negligible variation in the model (0.6%) (Table 5.9). Overall variation explained by the model was 48.5%, higher than the BNNC EMS area-wide model without fishing (Table 5.8).

Interaction term 'Biotope * Year' was highly non-significant (p = 0.917). However, post-hoc testing was undertaken in order to examine if changes in species composition of specific biotopes occurred. Pair-wise testing of 'Biotope * Year' for factor year showed that species assemblages only differed in biotope CR.MCR.EcCR.FaAlCr (p = 0.01) which was not affected by dispersion effects (PERMDISP, p = 0.16). All other biotopes had p > 0.17. Interaction term ‘Biotope * Year * Fishing pressure’ was also not significant (p = 0.142). No further post-hoc testing was required.

5.3.7. Change in Species richness (S) between years and fishing pressures

There were no differences in species richness (*S)* between years, except for biotope IR.MIR.KR.LhypT.Pk (Table 5.10). No obvious pattern in *S* across biotopes was observed (Fig 5.5), athough *Laminaria* spp dominated biotopes (Lhyp) had slightly higher *S* than faunal and algal crust circalittoral rock (FaAlCr).

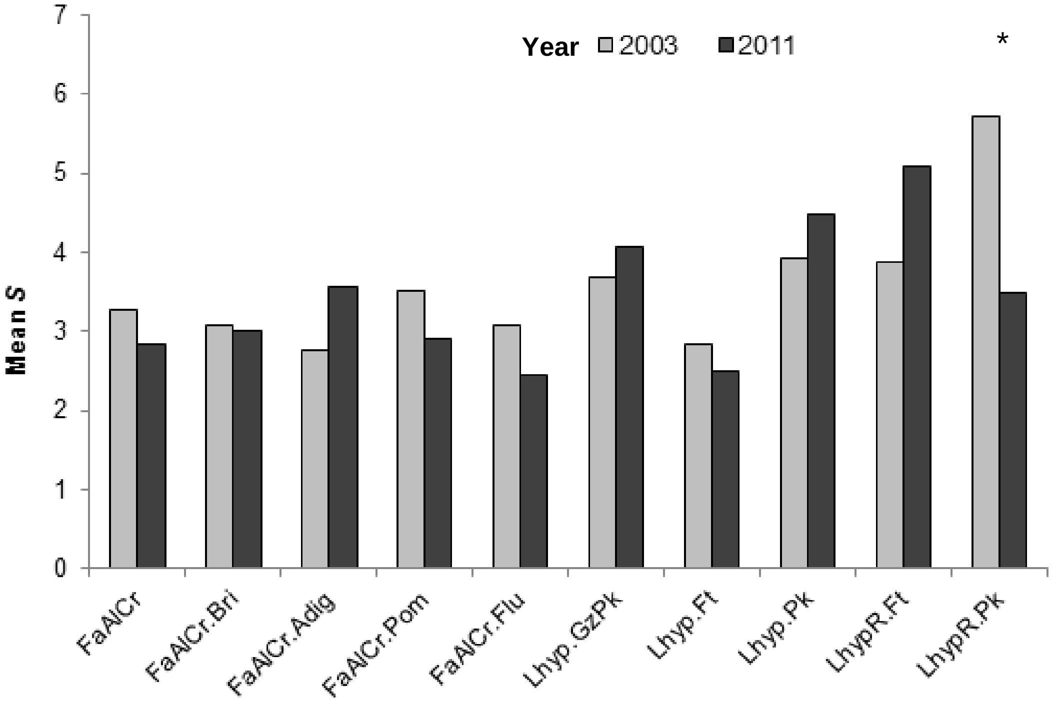


Fig 5.5. Mean S grouped by biotope in different years (2003; light grey and 2011; dark grey). Significant differences of S between years (for the same biotope) are shown with * symbol.

S differed between fishing pressures levels in three of the 10 biotopes tested (CR.MCR.EcCr.FaAlCr, CR.MCR.EcCR.FaAlCr.Bri and CR.MCR.EcCR.FaAlCr.Flu) (Table 5.10). Dispersion effects were not detected, confirming that these were location effects. *S* was higher at low fishing pressures within biotopes in all biotopes except CR.MCR.EcCR.FaAlCr.Bri (Fig 5.6).

The *S* of *Laminaria* dominated biotopes (Lhyp) did not differ with fishing pressure. Three out of the five faunal and algal crust circalittoral rock (FaAlCr) biotopes differed with fishing pressure, FaAlCr and FaAlCr.Bri diversity being reduced, and FaAlCr.Flu enhanced, at the highest level of fishing pressure.

Table 5.10. Species number (S) PERMANOVA table of results grouped by biotope for factors ‘Fishing Pressure’ and ‘Year’. Significant results are shown in grey shaded cells.

	FaAlCr	**FaAlCr.Bri**	**FaAlCr.Adig**	**FaAlCr.Pom**	**FaAlCr.Flu**	**Lhyp.GzPk**	**Lhyp.Ft**	**Lhyp.Pk**	**LhypT.Ft**	**LhypT.Pk**
‘Year’	0.55	0.914	0.33	0.26	0.33	0.14	0.34	0.27	0.086	0.026
‘Fishing pressure’	< 0.01	< 0.01	0.20	0.08	< 0.01	0.16	0.18	0.46	No test	0.71
‘Year * Fishing pressure’	< 0.01	< 0.01	0.34	0.14	0.13	0.18	0.54	0.58	No test	No test

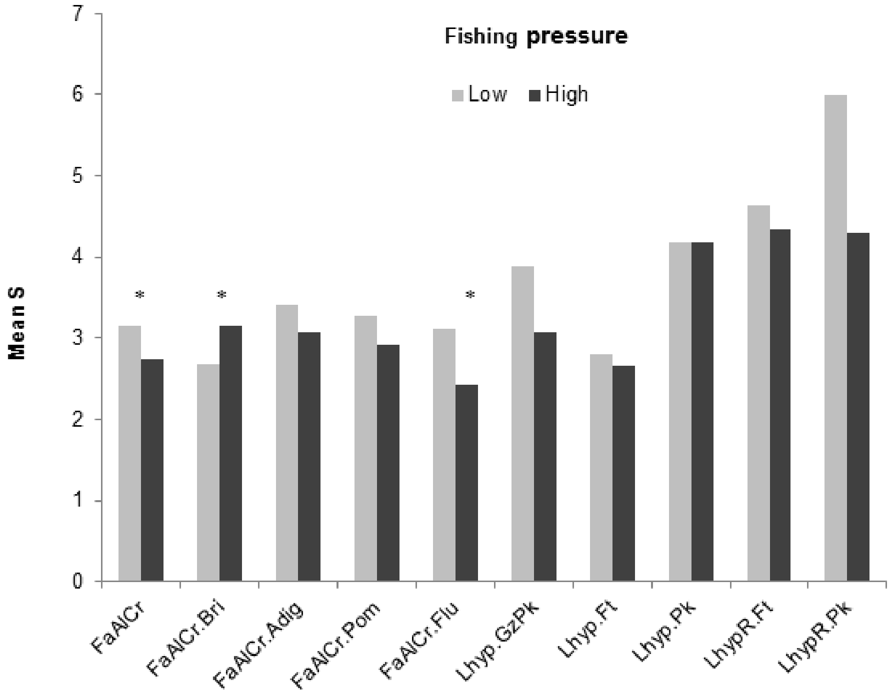


Fig 5.6. Mean species number (S) grouped by biotope for low (light grey) and high (dark grey) fishing pressures. Significant differences of S between fishing pressures (for the same biotope) are shown with * symbol.

The interaction term ‘Year * Fishing pressure’ was significant for *S* in biotopes CR.MCR.EcCr.FaAlCr, CR.MCR.EcCR.FaAlCr.Bri and post-hoc testing showed that *S* at low fishing pressure (Fig 5.7) differed between years ($p < 0.006$) for both CR.MCR.EcCr.FaAlCr, CR.MCR.EcCR.FaAlCr.Bri. However, *S* at high fishing pressures (black columns in Fig 5.7) did not differ between years ($p > 0.158$) for both CR.MCR.EcCr.FaAlCr, CR.MCR.EcCR.FaAlCr.Bri. The *S* of both CR.MCR.EcCr.FaAlCr, CR.MCR.EcCR.FaAlCr.Bri differed between fishing pressures in 2003 ($p < 0.001$) but did not in 2011 ($p > 0.247$).

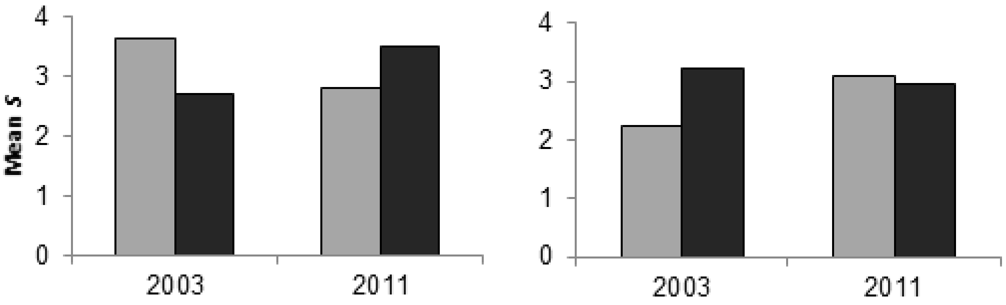


Fig 5.7. Mean S for each year (2003 and 2011) at low (light grey) and high (dark grey) fishing pressures. A) Biotope CR.MCR.EcCr.FaAlCr B) Biotope CR.MCR.EcCR.FaAlCr.Bri.

5.4. Discussion

This section discusses the usefulness of BNNC EMS data in investigating potential potting impacts. Observed changes in taxonomic composition of assemblages between years, both within individual transects and more broadly, across the BNNC EMS were discussed. Effects of fishing intensity on any change detected and results regarding change in species diversity and ecologically important species were explored. Finally, the use of monitoring data for this research was critically appraised.

5.4.1. Change in taxonomic composition of assemblages between years

As would be expected, for individual transect data the factor ‘Biotope’ explained the most variation in models. Other factors and interaction terms differed between transects (Table 5.2, Table 5.6, Table 5.7) but had low explanatory power overall. The amount of variation explained by individual transect models varied (Dunstanburgh ~ 41%, Boulmer ~ 45% and Farnes ~ 60%). The Farne Island’s model best explained the species and biotopes observed in the transect surveys. Other studies investigating epibenthic community structure explained similar levels of overall variation. In the Belgian North Sea 40.5% of the variation was driven by site specific conditions, such as distance to shore, depth, substrate, salinity and temperature (De Backer *et al.*, 2010). Here too, the low level of variation captured and the large scale of the study area mean that the impacts of human activities (if any) may not be discernible among natural variations (De Backer *et al.*, 2010).

Overall results indicate no changes in species composition of biotopes in the BNNC EMS across the study period; the interaction term ‘Biotope * Year’ being non-significant for all transects except Dunstanburgh. Therefore changes in species composition and patterns of presence/absence in specific species within biotopes between years could not be investigated with any certainty. However, *post-hoc* testing was undertaken to inform future research. Biotopes which showed differences in species composition between years and for more than two transects are summarised in Table 5.11.

Table 5.11. Biotopes with significant differences in species composition between years for all transects. Grey cells show results that are influenced by dispersion and perhaps (although not necessarily) a location effect as well.

Dunstanburgh transect	**Boulmer transect**	**Farne Island transect**
IR.MIR.KR.Lhyp.GzPk	Non-significant	IR.MIR.KR.Lhyp.GzPk
CR.MCR.EcCr.FaAlCr	CR.MCR.EcCr.FaAlCr	CR.MCR.EcCr.FaAlCr
IR.MIR.KR.Lhyp.Ft	IR.MIR.KR.Lhyp.Ft	Non-significant
CR.MCR.EcCR.FaAlCr.Flu	CR.MCR.EcCR.FaAlCr.Flu	Non-significant

The only biotope that showed change in species composition between years across all three transects was CR.MCR.EcCr.FaAlCr, although dispersion effects were detected in samples from Dunstanburgh (Table 5.11). This suggests that community assemblages across the BNNC EMS scarcely changed overall between 2002 and 2011.

For the two biotopes where differences in species compositions between years were detected in at least 2 transects, and were not confounded by dispersion effects (IR.MIR.KR.Lhyp.GzPk and CR.MCR.EcCr.FaAlCr – see Table 5.11), SIMPER showed that similar species contributed to differences between years. The level of influence and changes in the frequency of species presence did not show any patterns between transects. The random nature of the observed change makes identifying cause difficult (Clarke, 1993) but suggests location differences, most probably linked to natural variation, differences in visibility or local scale impacts not quantified in this research (Clarke and Warwick, 2001).

As discussed in chapter 4, natural variation in species composition of biotopes is very difficult to quantify. No work to date has quantified natural variability of either Faunal and Algal crusts (FaAlCr) or grazed *Laminaria* spp parks (GzPk) (Birkett *et al.*, 1998; Hartnoll, 1998). However, FaAlCr is a particularly variable biotope in terms of assemblage structure (Fig 5.2) and GzPk is by definition a transitional state, making differences in species assemblages between areas or years highly likely (Connor, 1995).

The present video data were collected in different seasons: the 2002 in early – mid August, whereas 2011 data were collected in mid – late October. Seasonal variations within individual biotopes also differ by location and inter-annually, albeit not quantified in any way in the UK biotope classification (Connor *et al.*, 2004; Sundblad *et al.*, 2013). However, seasonality is understood to have a major role in species composition and diversity (Connor *et al.*, 2004). It is expected that species richness will increase within reef habitats at the height of the summer due to the growth of ephemeral species and reach a low point during winter (Dring and Dring, 1991; Dayton *et al.*, 1998; Connor *et al.*, 2004). Using the data available, there is no way of quantifying the difference that sampling in different seasons may have had on species composition results. This was not an issue for monitoring, where the maintenance of characteristic biotopes was required and seasonal variation in species composition was not taken into account (chapter 4). Large differences in visibility are observed between years, believed to be commensurate with change in season. Lower visibility is apparent on all video samples from autumn 2011 compared to summer 2002. This is discussed further in section 5.4.6. Any conclusions derived from model results presented here should be treated with caution as inter-annual effects are likely confounded by seasonality.

Some evidence suggests that species composition may have changed between years within biotope CR.MCR.EcCr.FaAlCr. Significant differences were observed for all transects except Dunstanburgh, which may be confounded by dispersion effects. This is a surprising result because the species assemblages were particularly variable spatially (MDS plots, Fig 5.2) making changes difficult to detect. High variability can be explained in several ways. Firstly, in the hierarchical classification system, FaAlCr is not the most detailed level (biotope rather than sub-biotope), and is therefore found over a wide range of depths (10m - 50m). This results in individual samples being exposed to different light levels, turbidity and water movement (wave and current), all of which can affect species composition (Connor *et al.*, 2004). Much of the similarity across samples was driven by 5 species (*Alcyonium digitatum, Flustra foliacea, Pomatoceros triqueter, Echinus esculentus* and brittlestar spp (SIMPER analyses)). These easily identifiable species are found in the majority of biotopes in the BNNC EMS, occurring in all locations, with varying degrees of abundance, and could explain the perceived similarity of FaAlCr species composition with other biotopes. Secondly, a degree of error is associated with conservative identification of biotopes (Davies *et al.*, 2001). For example, biotopes identified as FaAlCr may in reality be FaAlCr.Adig. However, this is necessary where identification cannot be made with certainty, e.g. due to poor quality data (high sediment levels and poor visibility) (Foster-Smith, pers comm. 2013). Due to the high inherent variability in species composition within the more coarsely described FaAlCr biotope it is unusual that change was detected. Given the lack of specific patterns in species presence frequency and contribution between years, further investigation into potential change in species between years is required for this biotope.

5.4.2. BNNC EMS area wide data

In the area wide (BNNC EMS) model the wider spread of data observed in an MDS (Fig 5.4) compared to individual transects (Fig 5.2) suggests that small scale spatial effects may occur in species composition of biotopes CR.MCR.EcCr.FaAlCr, IR.MIR.KR.Lhyp.GzPk and CR.MCR.EcCr.FaAlCr.Flu (Fig 5.2). The greater variability in the area-wide data may reduce the possibility of detecting differences between years. The overall model explained less of the variation than two of the three transects (Boulmer and Farne Island) although factors and interaction terms of the overall model have similar percentage contributions to total variation compared to individual transects. In addition, p-values for factors and interaction terms of the BNNC EMS area-wide model differ compared to individual transect models, with higher p-values in the area wide model. For example, species composition differed with factor ‘Visibility’ at Boulmer and Dunstanburgh, but not in the overall model.

The choice of the spatial scale used to monitor and investigate marine community change over time is important (Jones *et al.*, 2000) (chapter 4). This research has shown that there are differences in species composition of biotopes at relatively small scales (between transects). Models using individual transects explained more of the variability and revealed potential differences in community composition between years (Dunstanburgh transect) compared to models using the area-wide data.

5.4.3. Change in taxonomic composition of assemblages with fishing pressure

The BNNC EMS area-wide model investigated species community change between years. It explained a low overall variation using three factors (‘Visibility’, ‘Biotope ‘and ‘Year’). A higher overall explained variation may be achieved by integrating further factors that affect species composition in to the model (Anderson *et al.*, 2008; De Backer *et al.*, 2010). Factor ‘Fishing pressure’ was included in the model to investigate if: 1) Potting pressure affected species composition of biotopes between years; 2) The addition of this factor increased the amount of overall variation in species composition explained by the model. There is no evidence that fishing pressure affects species composition. The inclusion of factor ‘Fishing pressure’ increased the variation explained by the model from approximately 43% to 48.5%. This increase was mainly attributable to interaction term ‘Biotope * Year * Fishing pressure’ explaining approximately 7.7% of the variability in the model (Table 5.9). Other variables and interaction terms only changed by small amounts.

Although interaction term ‘Biotope * Year’ was not significant, some post-hoc testing was undertaken to examine whether changes in species composition of specific biotopes across the EMS were similar to those seen in individual transects. Although the BNNC EMS area-wide model examines 13 biotopes for differences in species composition between years compared to 6 –8 biotopes for individual transect models, the inclusion of factor ‘Fishing Pressure’ reduced the number of biotopes in which species composition changes were detected between years compared to individual transect models. With the inclusion of factor ‘Fishing Pressure’, only biotope CR.MCR.EcCR.FaAlCr exhibited altered species assemblages between years. Even over large spatial scales and with the inclusion of fishing pressure as a factor, biotope CR.MCR.EcCR.FaAlCr had significantly differing species composition between years. Although caution should be used when interpreting these results, as only two time periods were sampled and abundance data was lacking (see section 5.4.6), temporal change is likely here. Further investigation into this biotope and its associated species to determine specific links to fishing pressure are recommended.

5.4.4. Change in species richness between years and fishing pressures

Very little evidence indicated that species richness (*S)* within biotopes differed between years. Differences were only detected in IR.MIR.KR.LhypT.Pk (Table 5.10), where replication was low (2002 n = 7 and 2011 n = 8).

In three out of ten biotopes, species richness differed between fishing pressure levels. Greater replication in each of these biotopes (n = 43–264) and lack of observed dispersion effects suggest this is a robust result. Nine out of the ten biotopes had higher species richness at low fishing pressures compared to the same biotopes at higher fishing pressures (Fig 5.6) although not all of these differences were significant. The only exception was biotope CR.MCR.EcCR.FaAlCr.Bri where low species richness in areas of high fishing pressure suggests that fishing may affect assemblage structure. However, these conclusions are speculative as further information is required; several fishing pressures levels, increased replication, investigation into spatial effects and more temporal data are all required to increase the reliability of any future work.

The intermediate disturbance hypothesis states that (Connell, 1978; Huston, 1979) both higher and lower species richness can exist under higher levels of disturbance (Townsend *et al.*, 1997). However, without extensive temporal sampling it is very difficult to determine the starting point of the community in relation to existing stress levels (Clarke and Warwick, 2001). In this case, higher species richness at lower fishing pressures seems logical, as these areas would be subject to lower amounts of disturbance. Differences in species richness with fishing pressure were only found in faunal and algal crust circalittoral rock (FaAlCr) biotopes: 3 out of 5 differed significantly, with a 4th only marginally non-significant (Fig 5.6). No differences in species richness were found in any of the five infralittoral *Laminaria* spp dominated (Lhyp) biotopes. Results indicate that biotopes most likely to be impacted by fishing pressure are deeper, faunal and algal crusts as opposed to shallower *Laminaria* spp biotopes. Deeper habitats in low energy systems generally have slower growing, relatively stable communities, but when damaged they are slow to recover original complexity (Hartnoll, 1998). These assemblages may be less able to withstand disturbances compared to those characterised by faster growing species found in the infralittoral zone (Connor, 1995; MacDonald *et al.*, 1996; Hartnoll, 1998). In general, longevity of species and community stability are believed to increase with depth, though data to support this are limited (Lundälv, 1985). Shallower habitats can be subject to harsher environment conditions such as increased wave action, tidal currents, sedimentation etc. compared to deeper habitats in the same area (Birkett *et al.*, 1998; Connor *et al.*, 2004)).

Both patchy areas of rock/sediment and more homogeneous rocky reef habitats may be classified as the same biotope (e.g. CR.MCR.EcCR.FaAlCr), they may host very different densities of lobster and crabs, and levels of species diversity. It is not known which habitats support the highest abundances of lobster and crab. However, habitat complexity and high species diversity are often believed to strongly influence successful settlement and recruitment (Eggleston *et al.*, 1999), therefore overall abundances of crab and lobster (Hartnoll, 1998; Hovel and Lipcius, 2001). Whether differences in biodiversity with different fishing pressures are a result of potting impacts or rather a result of variability in catchability within biotopes is not known. Longer time-series data would be required to determine between these.

Interaction term ‘Year * Fishing pressure’ was significant for biotopes CR.MCR.EcCr.FaAlCr and CR.MCR.EcCR.FaAlCr.Bri, however no clear pattern in *S* between years at different fishing pressures was observed (Fig 5.7). The low number of biotopes affected and the limited temporal data do not confirm whether potting pressure impacts the environment or not. Work has highlighted methods required for future research and areas for further investigation. Similarly to the investigation into species composition, *S* of biotope CR.MCR.EcCr.FaAlCr differed when tested using various factors and interactions terms. The consistency of change found within this biotope for both species composition and *S* merits further investigation (chapter 6).

5.4.5. Change in ecologically important species (reduced species list)

Analyses using indicator species compares change over time in particular species that are thought to be vulnerable to impacts. The species used in the reduced list of the present study were chosen since a loss or change in these could potentially indicate biotope sensitivity and were based on published studies of mobile fishing gear impacts (Hiscock and Tyler-Walters, 2006). A reduced species list may be more cost effective as the fewer biological data needed means increased samples can be collected compared to collection of full species list data (Sundblad *et al.*, 2013). Further, the choice of selected species for such a list will be dependent on the impact assessed (MacDonald *et al.*, 1996; Hiscock and Tyler-Walters, 2006) and can provide a focused approach, including for small scale impacts such as potting impacts (Eno *et al.*, 2001; Shester and Micheli, 2011). The reduced list used here produced similar results to the full species composition data. Specific to our null hypothesis, interaction term ‘Biotope * Year’ being non-significant for all transects shows that there has been no change in species indicative of biotope health within biotopes between years. This conclusion is slightly different to the previous section: there has been no change in species composition of biotope between years. There appears to have been no deterioration in ‘biotope health’ between 2002 and 2011 but we cannot make any conclusions on the state of health of these biotopes throughout this time. I.e. biotopes may be in a ‘healthy’ or indeed ‘unhealthy’ state but in either case there is no evidence to suggest that these have changed between years.

Although the ecologically important species showed no differences from the full species list, using a reduced species list has the potential to reduce cost, effort, time and increase sampling efficiency, i.e. more samples can be obtained in the same time for the same cost as using a full species list. Deciding between using a reduced species list and a full species list for scientific studies, should include weighing up the level of biological detail needed for the study, and the number of replicates / power needed to provide robust evidence (Davies *et al.*, 2001). Previous studies have used reduced species lists, using presumed sensitive species specific to potting impacts (Eno *et al.,* 2001; Sheridan *et al.,* 2003; Coleman *et al.,* 2012). However, these lists are often only applicable to specific geographic areas and no change in abundance of these sensitive species were detected (Eno *et al.*, 2001; Coleman *et al.*, 2012).

The reduced species list used here was based on biotope sensitivity to a single dredging event (MarLIN, 2012), the force of which is on a much greater scale than that of potting (MarLIN, 2012). Thus, this reduced species list may not include species sensitive to specific potting impacts and therefore impacts may not be detected. Direct experimental observation of potting impacts on epibenthos would provide a more robust list of impacted species for investigation into the long-term impacts of potting (Dayton *et al.,* 1998) (chapter 6). As with the use of the full species list, there may be methodological issues with drop-down camera data (see section 5.4.7) that may hinder the detection of change.

5.4.6. Comparison of the effectiveness of methods for determining change

All methods used in this research to determine ecological change between years show similar patterns; some change in species composition or richness was measured for each factor explored, but this was not consistent across biotopes (rejection of hypothesis 1). Differences in significant factors between models preclude the possibility of identifying the cause for any observed impact (rejection of hypothesis 2). Overall, there was little evidence that biotopes have changed between years in terms of species richness, full or reduced species composition. Biotope CR.MCR.EcCr.FaAlCr, which had the most samples (n = 264), and was not affected by dispersion effects in most cases, suggests that further investigation would be beneficial, as consistent differences were demonstrated throughout. Species richness provided the most robust evidence, but evenness was not measured due to the lack of abundance data; diversity (*H'*) and dominance might have provided increased information on environmental stress and trends in environmental change (Clarke and Warwick, 2001).

5.4.7. Adequacy of monitoring data for scientific investigation

Overall, no consistent potting impacts were found. This is attributable largely to limitations of the monitoring data, which include a lack of scale on the video, hierarchical classification used (chapter 1 and 4), issues with seasonality and inconsistencies of the quality and visibility of the video data. These are further discussed below.

5.4.7.1. Lack of scale on the videographic data

Estimates of scale were not needed for the ecological monitoring program of the BNNC EMS as only the analysis of biotope frequency and composition (chapter 4) was required to ensure that conservation requirements were met through the maintenance of features of interest and by confirming that no large scale changes had occurred (Davies *et al.*, 2001; Saunders *et al.*, 2011). However, in order to investigate smaller scale impacts, detailed hypotheses and in-depth biological information are required.

Specifically for this research, the lack of scale available in the video data resulted in a loss of spatial and ecological resolution in the analysis. The presence/absence of species can result in certain biotopes looking identical on paper whereas in reality they may be very different. For example, kelp forests and kelp parks, both of which can be grazed or not, are characterised by similar species (*Laminaria hyperborea*, urchins and foliose red algae). Whereas the ungrazed kelp forest has dense *L. hyperborea*, low abundance of urchins and an understory of red algae, a grazed kelp park will only have very sparse *L. hyperborea* and red algae and a high abundance of urchins (Fig 5.8). The exaggerated similarity of *Laminaria* spp biotopes is highlighted in the MDS plots of section 5.3.1 (Fig 5.2) where biotopes show high levels of similarity to each other (shown by grouping of samples).

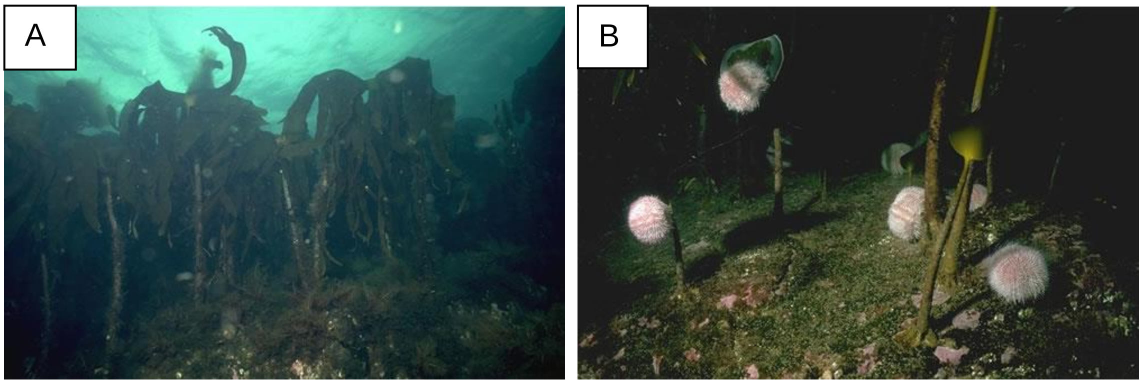


Fig 5.8. Photographs (at different scales) used to illustrate the differences in abundance of characterising species of A) kelp forest and B) grazed kelp park. (images from Paul Brazier, JNCC).

The lack of abundance data means that analysis of species composition change of biotopes between years, using only presence/absence, will only be detected if there is a consistent loss of species. Changes in species of this magnitude would only be likely to occur due to large disturbances occurring at a wide spatial scale such as trawling impacts (Kaiser *et al.*, 2000). Lack of abundance data also limits the statistical tests available to investigate subtle changes, and it may place too much emphasis on rare species (Clarke, 1993). This may increase variability and mask real impacts. Indeed, removal of rarer species is recommended for multivariate species assemblage analyses using abundance data (Clarke and Warwick, 2001). Determination of which species were rare and should be excluded from the analysis was not possible here, due to the lack of scale on video images. Future studies using drop-down cameras should have means of obtaining reliable abundance data in order to be able to provide evidence of lower-level impacts (Sotheran *et al.*, 2004). More recent habitat condition monitoring are collecting abundance data using videographic methods in order to improve accuracy of analysis (CEFAS, 2012;

Fitzsimmons *et al.*, 2015). For management purposes, changes in abundance may allow possible impacts to be identified and appropriate measures to be implemented in time to allow recovery to healthy levels.

The identification of biotopes from drop-down video data relies heavily on identification of characterising species. Cryptic species will not be detected (Saunders *et al.*, 2011) and in-situ identification by a taxonomist is often more accurate (Lam *et al.*, 2006). This is an important limitation associated with using videographic methods, and when coupled with the lack of abundance estimates, objective identification of biotopes is made more difficult. Defining physical biotope features (boulders, cobbles, particle size, etc.) could not be estimated furthering the subjective nature of biotope identification (Bullimore *et al.*, 2013).

5.4.7.2. Seasonality and annual issues

Detection of trends and changes in ecosystems requires a robust benchmark. This must be detailed enough to allow distinction between natural and anthropogenic changes (Dayton *et al.*, 1998; Lindenmayer and Likens, 2009). The monitoring data used in this research were collected in the summer of 2002 (with some Farne Island transect data in summer 2003) and in the autumn of 2011. This adds elements of both annual variation for the Farne Island transect 2002/03 data and seasonal variation between all the data from 2002 and 2011. These data cannot be used to distinguish between seasonal variation and other factors or impacts. The less stringent scientific requirements of monitoring data mean this may not necessarily be an issue for surveillance or monitoring. However, for the purposes of more in-depth scientific investigations, seasonality issues would need to be clarified in order to infer any causality to any change detected.

Due to centuries of fishing before designation in 2000, the BNNC EMS may already be a heavily modified marine environment. Undocumented removal of megafauna through fishing means there is little understanding of what the pristine natural community may have been like, and such a baseline cannot be established. The ability to separate anthropogenic impacts from natural ecosystem dynamics is therefore severely compromised (Dayton *et al.*, 1998).

Extensive baseline data are needed to distinguish between natural and anthropogenic change (Lindenmayer and Likens, 2009; De Backer *et al.*, 2010). For this work data from only two years separated by a decade were available making any detected changes difficult to attribute to either natural annual variation or anthropogenic impacts. The appropriate number of yearly sampling events needed to detect a change will vary depending on the scale of the impact and natural variations within community assemblages studied (Dayton *et al.*, 1998). The exact number of samples can only be determined after a preliminary study (Hartnoll, 1998). Results from this work do not provide abundance data, rendering this impossible.

5.5. Concluding remarks

Very little evidence of change in species composition or species richness of biotopes between years was found (rejection of hypothesis 1). It was not possible to effectively investigate the role of fishing pressure on community change (rejection of hypothesis 2). However, changes in species composition and richness of biotope FaAlCr were observed between years in most models and between fishing pressure levels. Further research could be beneficial. Issues with the data available reduced the robustness of results and conclusions. Abundance data and increased sampling across years and seasons are necessary to effectively determine whether any change in species composition or richness detected are a result of anthropogenic causes or natural variation. Results from this research suggest that on the scale of the BNNC EMS, which may already be in an altered state from millennia of fishing, small-scale potting impacts on epibenthic assemblages may not be detectable against the background of natural variability (De Backer *et al.*, 2010). This suggests that anthropogenic impacts are not pushing the ecosystem outside natural limits. However, to unravel the long term impacts, sliding community baselines and small scale impacts, more detailed studies with increased temporal sampling is required (Dayton *et al.*, 1998; De Backer *et al.*, 2010). In-situ observations of direct potting impacts through experimental fishing will provide further information on some of these issues and are discussed in chapter 6.

Chapter 6 . Direct Potting Impacts on Common Reef Habitats in the Berwickshire North Northumberland Coast European Marine Site

6.1. Introduction

Habitat quality and health are integral to management of sustainable fish stocks as part of an ecosystem based approach to fisheries management (EBFM) (Pikitch *et al.*, 2004; Armstrong and Falk-Petersen, 2008; Salomidi *et al.*, 2012). On temperate reefs epifauna are key ecosystem components (Hiscock and Tyler-Walters, 2006) providing a source of food and shelter for other species, including commercially important ones (Lambert *et al.*, 2011). Effects of high-impact mobile fishing gears on habitat and benthos have been well documented; epifauna are sensitive to both direct (physical disturbance) and indirect (changes of food web structure) fishing impacts (Kaiser *et al.*, 1996; MacDonald *et al.*, 1996; Collie *et al.*, 1997; Collie *et al.*, 2000a; Collie *et al.*, 2000b; Kaiser *et al.*, 2006; Lambert *et al.*, 2011). In contrast, physical impacts of static fishing gear such as baited pots on benthic habitats have been less studied and are debated (Eno *et al.*, 2001; Lewis *et al.*, 2009; Shester and Micheli, 2011; Coleman *et al.*, 2013) although further information is required for management (MMO, 2012; Defra, 2013). The present research aims to investigate direct potting impacts on epibenthos of common reef habitats using marine protected areas in the North East of England as a case study (BNNC EMS and CQSM MCZ, Fig 6.1, A).

Chapters 4 and 5 investigated the adequacy of using previously collected monitoring data to detect ecological change in the BNNC EMS between 2002/03 – 2012 and whether any changes found could be attributed to potting impacts. Although there was little evidence of potting impacts, this could be due to methodological issues (see chapter 4 and 5) and observations of direct impacts *in situ* through experimental fishing were recommended.

Previous potting impact studies did not find evidence that potting impacted habitat or epibenthos, however these studies also had limitations. Eno *et al*. (2001) conducted experimental fishing in the South West of England, but the low replication and analytical methods used reduced statistical power and led to other issues (Eno *et al*., 2001; Coleman *et al.,* 2013). For example, certain species appeared to significantly increase in abundance in response to the potting treatment although these species could not have settled and grown within the time period and their presence was attributed to sedimentation prior to the experiment (Eno *et al.*, 2001). Comparison of circalittoral assemblages in sites both in fished areas and a protected no-take zone (NTZ) in south west England showed there had been no change in sessile epifauna composition over four years between these sites (Coleman *et al.,* 2013). However, one of the major difficulties with assessing the localised impacts of potting is the small physical footprint of pots (≈ $1m^2$). Interactions between the seafloor and potting gear occur on a small scale compared to the large area that the potting gear covers (fleet of 12 pots ≈ $800m^2$). The low sample size compared to the large experimental area, coupled with control sites differing in depths (5 – 10m), exposures (controls more wave exposed than experimental areas) and substrate differences (boulder reefs compared to bedrock) compared to experimental sites (Coleman *et al.,* 2012), meant that small scale impacts may have been overlooked. No studies to date have tried to control for habitat differences when examining direct potting impacts. This may be particularly important in order to detect low-level small-scale impacts. Expert knowledge on sensitivities of specific habitats to fishing, including potting, suggests that certain reef habitats such as ‘Rock with erect and branching species’, ‘Rock with low lying fast growing faunal turf’ and ‘Shallow subtidal rock with kelp’ are sensitive to high levels of potting activity (Eno *et al.*, 2013), although experimental evidence is required to confirm this is the case.

The few studies of direct potting impacts have only been conducted in circalittoral habitats. Depth may play a significant role in pot movement in the spiny lobster (*Panulirus argus*) fishery in Florida, single pots impacted a greater area of benthos in shallow water compared to deep water due to increased wave action (Lewis *et al.*, 2009). The present research focuses on impacts on shallower infralittoral and circalittoral habitats, as these may be more susceptible to damage due to increased wave action. It aimed to experimentally fish pots in specific habitats with control sites in close proximity with indistinguishable environmental conditions. It also aimed to address issues of statistical power with high-density SCUBA diver sampling using a replicated symmetrical BACI experimental design. This allowed cumulative impacts to be investigated in a robust and controlled manner.

Based on recommendations from chapter 5, two subtidal habitats within the BNNC EMS (Mercer *et al.*, 2003; Mercer, 2012) were selected for investigation: ‘Faunal and algal crusts on exposed to moderately wave-exposed circalittoral rock’ (abbreviated as FaAlCr) and *Laminaria hyperborea* park with foliose red seaweeds on moderately exposed lower infralittoral rock (abbreviated as Lhyp.Pk). These habitats, which are commonly found across Europe and the UK (Connor *et al.*, 2006; MESH, 2008), have been identified as important within the BNNC EMS (Regulation 33(2) advice, English Nature & Scottish Natural Heritage 2000), as they are characterised by erect and presumably fragile species, likely to be impacted by even low levels of disturbance (Roberts *et al.*, 2010). There is some evidence that species composition and species richness of FaAlCr may have changed between 2002/03 – 2011 (chapter 5). In addition, Lhyp.Pk has high biodiversity (including erect species) and high biomass (Connor, 1989; Connor *et al.*, 2004) and may be particularly prone to impact due to higher levels of wave action that could result in underwater pot movement and abrasion (Lewis *et al*., 2009).

It was hypothesized that abundances of the erect and potentially fragile species would decline due to physical abrasion from pot fishing with increases in bare rock and greater percentage cover of encrusting species (hypothesis 1). For mobile gear, benthos of lightly fished areas has greater abundances, especially of erect fauna and flora, higher biomass and larger individuals compared to intensely fished sites (Collie *et al.*, 1997; Collie *et al.*, 2000b; Kaiser *et al.*, 2002; Lambert *et al.*, 2011); thus here, historically lightly fished sites were expected to be more sensitive to fishing impacts. Potting pressure for the NIFCA district was modelled using a combination of vessel sightings and landings data (chapter 2) and experimental sites within both historically intensively and lightly fished areas (herein referred to as intensively and lightly fished sites respectively) were subjected to the same amount of experimental potting. It was hypothesized that lightly fished areas would have greater abundances of erect long-living species (hypothesis 2), and that these would exhibit greater change over time in assemblage structure than intensively fished areas (hypothesis 3).

6.2. Methods

6.2.1. GIS layers and site selection

The focused nature of the experimental work in terms of habitat type, sampling method and fishing pressure required appropriate site selection. This was undertaken in two stages. The first stage was the geographical mapping of individual selection criteria (depth, bottom substrate hardness, fishing pressure and habitats) which were combined in a GIS to enable querying and selection of areas with the desired attributes. The second stage, consisted of ground truthing the selected areas to finalise experimental areas using Scuba and expert knowledge to select habitat type and to avoid conflict with commercial fishing practices.

Bathymetry and hardness layers were created for the whole of the Northumberland inshore fisheries district using Olex data (2012 – 2014) from the NIFCA patrol vessel the *St Oswald* (Fig 6.1, C and D). The Olex system collects data from the ships' echosounder and GPS which it uses to provide an estimate of depth and seabed hardness for individual points. Over time, with the accumulation of data, these maps provide detail which far exceeds official paper charts (OLEX, 2014). Data were extracted from the Olex system and uploaded as a shapefile in Esri ArcGIS (v. 10.1). This point feature consisting of X, Y coordinates, depth (m) and substrate hardness (%) was used to produce a kernel density distribution which produces a continuous surface in a GIS. Kernel density distribution function was used to produce estimates of depth and substrate hardness (Fig 6.1, C and D). The smoothing factor determines the area around a given location within which data points contribute to the probability estimate (Wand and Jones, 1995; Van Der Veen and Logtmeijer, 2005). The distance (or search radius) of the smoothing factor was chosen as the mean distance between all points.

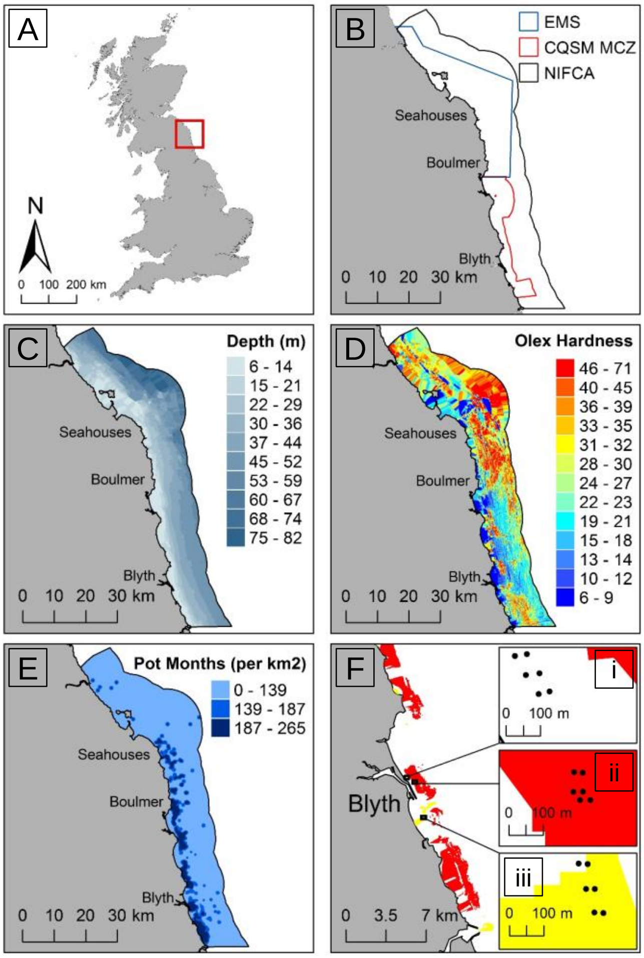


Fig 6.1. Location of Northumberland in Great Britain (A). Location of Berwickshire and North Northumberland Coast European Marine Site and the Coquet to St Mary's recommended MCZ within the NIFCA district (B). Bathymetry (depth below chart datum, m) (C). Olex Seafloor hardness (percent hardness) (D). Spatial distribution of mean summer fishing effort (2012 – 2013) (number of pots month^{-1} km $^{-2}$) (E). Spatial extent of areas matching broad selection criteria (red: < 16m, > 35% hardness, high fishing pressure; yellow: < 16m, > 35% hardness, low fishing pressure) (F). Final sites for experimental potting of Lhyp.Pk in intensively fished areas (insert i), FaAlCr in intensively fished areas (insert ii) and lightly fished areas (insert iii).

Summer fishing effort maps for 2012-2013 were produced for the NIFCA district following methods described in chapter 2. Fishing effort was divided into three categories for ease of use, using the natural jenks function (Esri ArcGIS v. 10.1): low (0 – 139 pots month^{-1} km $^{-2}$), medium (140 – 187 pots month^{-1} km $^{-2}$) and high (188 – 265 pots month^{-1} km $^{-2}$) fishing pressure (Fig 6.1, E).

Individual GIS layers were combined, subjected to selection rules, and highlighted areas assessed using expert knowledge. Due to the restrictions associated with scuba diving in terms of depth and bottom time available, sites were restricted to shallower than 16m depth. After discussion with NIFCA fisheries officers and skippers of the Newcastle University research vessel, a conservative value of hardness attributed to rocky reefs was determined as any area with percent hardness > 35%. Sites with medium fishing pressure (140 – 186 pots month^{-1} km $^{-2}$) were excluded. Areas matching the selection criteria are shown in Fig 6.1, F.

Finally, areas matching the broad scale selection criteria were further refined by selecting areas with appropriate habitats (FaAlCr and Lhyp.Pk) using previous ecological monitoring data (Edwards, 1983; Foster-Smith and Foster-Smith, 1987; Connor, 1989; Holt, 1994; Brazier *et al.*, 1998; Mercer *et al.*, 2003; Mercer, 2012) and more recent CEFAS habitat maps (CEFAS, 2012). A small selection of areas were ground truthed prior to experimental deployment of fishing gear. Final site selection is shown in Fig 6.1, F. Areas large enough for experimental potting were not found for Lhyp.Pk in lightly fished areas. It is thought that this is because kelp dominated habitats are all heavily fished during the summer months (NIFCA, pers comm.).

6.2.2. Design and layout of experimental fishing

Potting impacts were investigated by experimentally fishing and monitoring impacts over time using Scuba diving and photoquadrats. Three sites were selected for each fishing pressure (acting as replicates) (Fig 6.2) at similar depths and distances from each other (Van Rein *et al.*, 2012) . FaAlCr sites were sampled during July – September 2014 and Lhyp.Pk sites were sampled during July – August 2015.

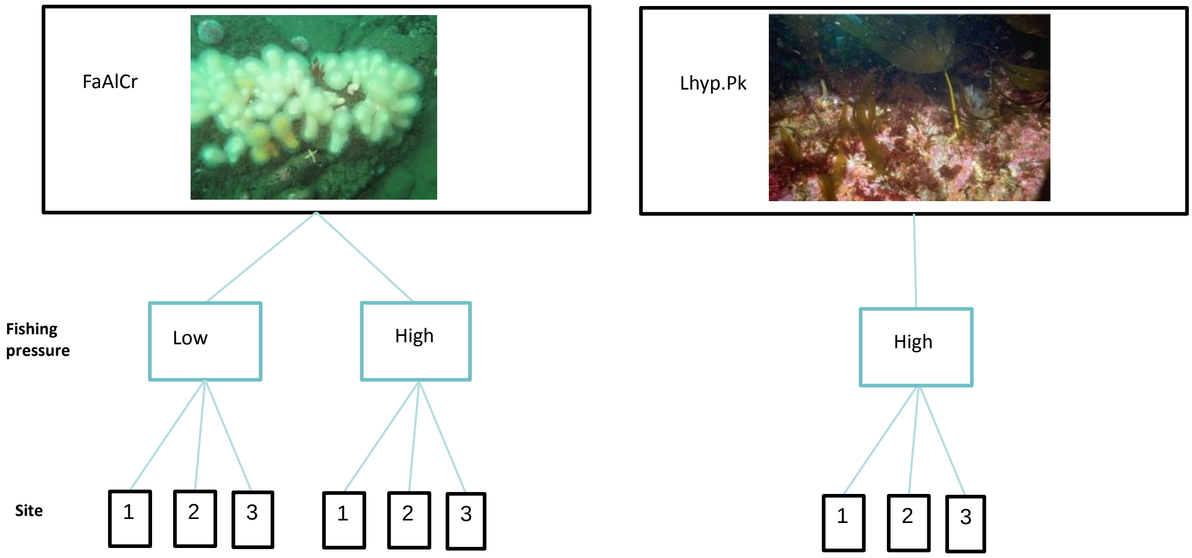


Fig 6.2. Diagram of the experimental design for biotopes FaAlCr and Lhyp.Pk. Summer fishing pressure; low = 0 –139 and high = 187 - 265 pots month^{-1} km^{2}. Sites within habitats had similar depths. Lhyp.Pk from Paul Brazier.

Experimental sites comprised two areas: a 25 x 10m impact area and a smaller 5 x 10m control area (Fig 6.3). These were delimited by anchor weights on the East and West sides of the site and located at the surface by marker buoys (Fig 6.3). Experimental sites were orientated in line with the tide which flows North to South.

A balanced BACI design (Before-After Control-Impact, Smith (2006)) was used. Scuba divers, using 625cm^2 photoquadrats (25 x 25cm) collected 120 samples at each site, sampling epibenthos and habitat type and providing a baseline of the area (Fig 6.3)(further detail on sampling methods in section 6.2.3). Following baseline sampling, a single standard steel framed parlour pot (0.66 x 0.46 x 0.38m with 10mm steel frame) attached on a 3m leg, tethered to a mainline maintained in place using two anchor weights, was left to soak in the impact areas of each site for a minimum of 24 hours (FaAlCr mean soak time : 71hrs 50mins (intensively fished areas) 64hrs 53mins (lightly fished areas) and Lhyp.Pk mean soak time: 168hrs 32mins (intensively fished areas)) (Fig 6.3) and then hauled against the tide (North ↔ South), following local commercial practice, by Newcastle University research vessel, *The Princess Royal,* the NIFCA patrol vessel, *St Oswald* or by hand from the dive RIB *Arktos* if water depth was < 10m. The experimental area was resampled, a further 120 photoquadrats being collected per site. Soaking, sampling and hauling were repeated a total of three times for each site.

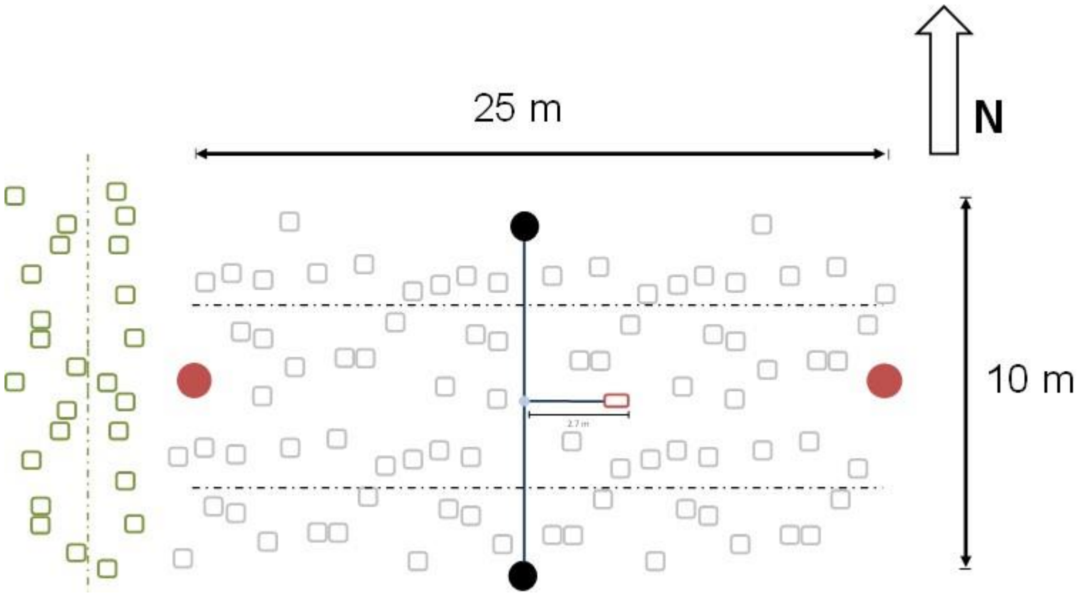


Fig 6.3. Diagram of example study site (Impact area (25 x 10m) and a control area (green line and squares, 5 x 10m) showing randomly distributed sample areas (grey squares, 0.625m^2 each), ≈ 20kg anchor weights (red circles), tape measures used for transects (dotted black line) and experimental potting fleet: 20kg weights (black circle), rope (blue lines) and parlour pot (red rectangle).

The fishing pressure of this experimental study was much higher than that of lightly and intensively fished areas in 2012-2013 (Table 6.1). The maximum number of pots at sea month^{-1} km^{-2} for lightly and intensively fished areas (Fig 6.1, E) were multiplied by the number of hauls per month to provide an estimate of fishing pressure (pot hauls month^{-1} km^{-2}). No data was available for the number of hauls in the region, so it was assumed that pots were hauled every day - a very conservative assumption because poor weather conditions will often prevent fishers hauling pots for at least some days in each month.

Table 6.1. The estimated maximum number of pots in the sea per month per km^{-2} (Fig 1), the number of hauls per month (estimated conservatively) and fishing pressure (pot hauls month^{-1} km^{-2}) in lightly and intensively fished areas in Northumberland compared to the experimental fishing undertaken.

	Maximum number of pots in the sea month^{-1} km^{-2}	Number of hauls month^{-1}	Fishing pressure (pot hauls month^{-1} km^{-2})
Lightly fished	139	30	4170
Intensively fished	265	30	7950
Experimental fishing	10,000	3	30,000

6.2.3. In-situ sampling

In-situ sampling was undertaken by Scuba divers using a 625cm^2 photoquadrat (25 x 25cm) (Canon Powershot s120, Ikelite underwater housing and a Sea&Sea YS-02 Underwater Strobe) (Fig 6.5, A) enabling sampling to be taken at the positional accuracy and selectivity needed for his type of in-depth study (Sayer, 2007). Diving was planned for slack water as recommended by Holt and Sanderson (2001, Section 3-3) and followed standard UK Diving at Work Regulations (1997). The use of the small photoquadrat (25 x 25cm) resulted in high resolution imagery of the 625cm^2 area and the short focal length (40cm) reduced the negative influence that water column turbidity would have on species identification (Leujak and Ormond, 2007). The simple deployment and ease of operation meant that *in-situ* task loading was minimised, thus increasing sampling replication for a given bottom time (Van Rein *et al.*, 2012).

Transects were deployed 2.5m South and North of the anchor weights respectively (dotted black line, Fig 6.3). Photoquadrat sampling was undertaken haphazardly on both sides of the tape measure up to 2.5m away once every metre resulting in a total of 50 photographs collected per transect. A further 20 quadrats were sampled in the control area in the same manner (Fig 6.3). Due to the spatial heterogeneity of temperate subtidal rocky reef habitats (Birkett *et al.*, 1998; Hartnoll, 1998; Van Rein *et al.*, 2012) a large number of small sized quadrats were deemed to provide a greater accuracy, precision and efficiency when describing communities compared to fewer large quadrats (Sayer and Poonian, 2007). As the small quadrat size did not allow for easy sampling of kelp abundance (*Laminaria hyperborea*, > 50cm), a crucial element of Lhyp.Pk community composition and structure, kelp abundance was recorded in a 1m^2 area on either side of transects every 5m.

6.2.4. Data extraction

In order to maintain efficient sampling, communities were treated as two-dimensional (Meese and Tomich, 1992; Van Rein *et al.*, 2011). Benthos from the images collected from FaAlCr habitats were identified and recorded to the lowest taxonomic level using up-to-date identification manuals (Cornelius, 1995; Hayward and Ryland, 1995; Bunker *et al.*, 2012; Wood, 2013b; Wood, 2013a) and expert knowledge. Epibenthos from the images collected from Lhyp.Pk habitats was identified to broad benthic categories of coarse taxonomic resolution to represent their structural role within the community. This was due to difficultly identifying individuals to species level (partially obstructed by shadows from, or overlapping algae) as well as the need for microscopic equipment for identification of many algal species. The groups selected (Van Rein *et al.*, 2012) were: red foliose algae > 1cm height above the substratum, non-red foliose algae > 1cm height above the substratum, red algal turf <1cm height above the substratum, non-red algal turf < 1cm height above the substratum, encrusting algae, mixed hydrozoan/bryozoan turf <1cm, feather hydroids and encrusting bryozoans.

Benthos which could not be identified because it was too far away (e.g. in a crevice) or too dark was excluded from the analysis. Mobile species were also excluded from the analysis. Percentage cover was recorded using a point method (Aronson *et al.*, 1994), which is time efficient, unbiased by observers, and is more sensitive to changes in community composition than visual estimation and frequency occurrence methods (Aronson *et al.*, 1994; Drummond and Connell, 2005; Van Rein *et al.*, 2012). Images were overlaid with 100 evenly placed points (Drummond and Connell, 2005), using photoQuad (v1.0) (Trygonis and Sini, 2012), the epibenthos under each point was identified and represented 1% of the image area (or 6.25cm^2) (Fig 6.5, b). Species abundances from a random sample of 20 images from each habitat were recorded using the point intercept method by a second experienced ecologist and species abundances were cross verified to the original abundances estimates. All cross verified images did not significantly differ in abundance or species composition between researchers.

6.2.5. Data analysis

The location of the pot or anchor weight along the transect was recorded during every dive. The data for the site were divided into 5 sections (north-south) along the transect (every 5m) (Fig 6.4). For each site, images from a 10 x 10m area, which had experienced maximum impact, were used for analysis. Data pre fishing, termed 'Baseline' data (B) were compared to the same areas after fishing, termed 'Impact' (I). To ensure any changes detected were potting impacts and not natural variation, data for control sites were analysed before and after (referred to as Control Baseline (C_B) and Control Impact (C_I) data, respectively). The spatial scale used for this experimental work was purposely small-scale in order to investigate any impacts with a high degree of accuracy. Experimental fishing studies provide useful insights into direct impacts, and relative severity of these as well as investigating habitat – fishing gear interactions (Kaiser *et al.*, 2006; Hinz *et al.*, 2009). However, results from small-scale experimental fishing impact studies are difficult to extrapolate to an ecosystem-wide scale which are ideally required for EBFM (Hiddink *et al.*, 2006; Hinz *et al.*, 2009). However, due to the limited research on potting impacts to date, robust and focused experimental evidence is first required (Eno *et al.*, 2001; Gray *et al.*, 2006).

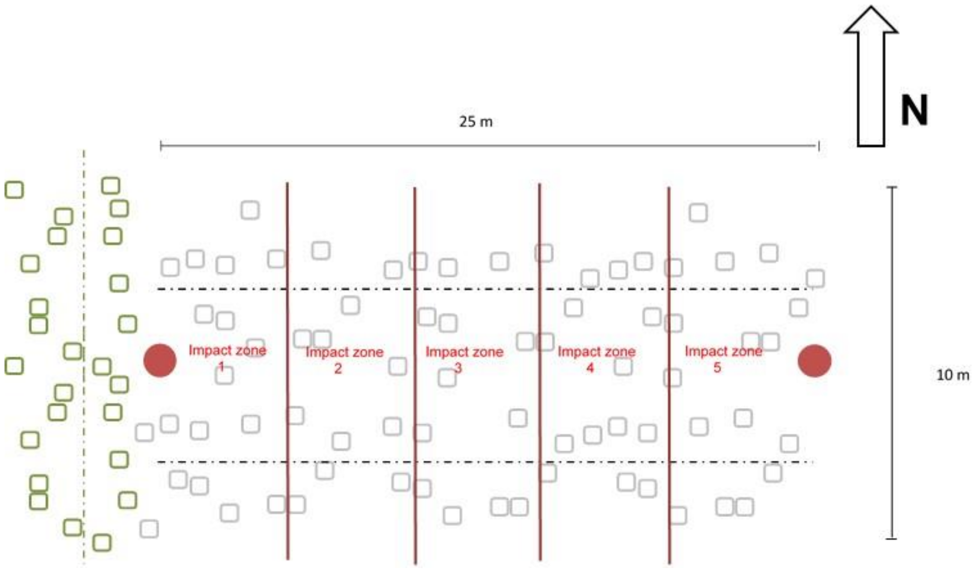


Fig 6.4. Diagram of example study site (Impact area (25 x 10m) and a control area (green line and squares, 5 x 10m) showing randomly distributed sample areas (grey squares, 0.25 x 0.25m^2 each), ≈ 20kg anchor weights (red circles), tape measures used for transects (dotted black line). Impact zones (red line, every 5m) are shown.

A total of 920 images were analysed (40 images for each B, I, C_B and C_I per site) for both high and low fished FaAlCr areas and 460 images were analysed (40 images for each B, I, C_B and C_I per site) for high fished Lhyp.Pk areas. Benthic community data were square root transformed to reduce dominance of common taxa (Martin *et al.*, 2012) and Bray Curtis similarity matrices produced. Exploratory statistics including multivariate analysis (cluster dendograms and multi-dimensional scaling plots) were conducted using PRIMER (v.6).

Differences in community data were explored between experimental treatments 'Baseline' – 'Control Baseline', 'Baseline – Impact', 'Control Baseline' – 'Control Impact' and 'Impact' – 'Control Impact' using mixed models in PERMANOVA (v. 1.0.5) (type III sum of squares, under a reduced model with 999 permutations) following a 2 factor design with interaction (site as a random factor with 3 levels: site 1, 2, 3, and treatment type as a fixed factor with 2 levels: B – C_B, B – I, C_B – C_I, I -C_I) for FaAlCr and Lhyp.Pk habitats. Tests of homogeneity of dispersion (PERMDISP routine in PERMANOVA) were used to test the null hypothesis of no difference in dispersion among *a priori* groups. Post-hoc analysis using the pairwise function in PERMANOVA (v. 1.0.5) investigated which factor levels were responsible for significant interactions (type III sum of squares, under a reduced model with 999 permutations).

The components of variation were estimated to provide a measure of the multivariate variability between factors within data sets (Anderson *et al.*, 2008; Martin *et al.*, 2012; Van Rein *et al.*, 2012). Due to the use of Bray-Curtis similarity matrices, the estimates of components of variation can be directly interpreted as percentage dissimilarity of conditions within experimental factors (Anderson *et al.*, 2008). Variability associated with factor 'site' indicates general spatial variability, that associated with factors B – C_B, B – I, C_B – C_I, I -C_I indicates temporal variability and that associated with 'Residuals' indicates more specifically the residual variability among replicate photoquad samples (Anderson *et al.*, 2008).

To visualize multivariate patterns, principal coordinate analysis (PCO) was used (PERMANOVA v. 1.0.5) because the variation explained by the axes of all plots was high, these were able to capture the high-dimensional structure adequately and thus provide a closer reflection of the resemblance values used in the partitioning methods for PERMANOVA than a non-metric MDS plot (Anderson *et al.*, 2008). There were too many samples to visually examine differences in assemblages between treatments B, I, C_B and C_I in a single ordination (n = 480) therefore centroids of each treatment and site were produced and plotted using PCO (Terlizzi *et al.*, 2005).

Similarity of percentage analysis (SIMPER, Clarke (1993)) was used to identify the percentage similarity that benthos contributed to the measure of Bray-Curtis Similarity for treatments (B – I and C_I – C_B). This analysis allowed identification of benthos that were most important in differentiating between treatments. Benthos were selected as important if they contributed to > 10% dissimilarity and if the dissimilarity/standard deviation ratio was > 1 (an indicator of consistency in contribution to dissimilarity across samples) (Terlizzi *et al*., 2005; Clarkre, 1993).

A total of 72 kelp abundance recordings were obtained from Lhyp.Pk habitats. Differences in kelp abundance between treatments were tested for significance in PERMANOVA v. 1.0.5 (type III sum of squares, under a reduced model with 999 permutations) following a 2 factor design with interaction (site as a random factor with 3 levels: site 1, 2, 3, and treatment type as a fixed factor with 2 levels: B – C_B, B – I, C_B – C_I, I –C_I).

6.3. Results

6.3.1. Site features

Intensively and lightly fished FaAlCr sites were on hard rocky substrates which had similar assemblages dominated by faunal (e.g. encrusting bryozoan, *Spirobranchus* spp and short hydroid turf) and algal (Corallinaceae) crusts (Fig 6.5, B and C). Sites tended to have a grazed appearance, potentially due to the abundance of *Echinus esculentus* (Connor *et al.*, 2004). Erect species such as *Alcyonium digitatum* and other feather hydroids were frequently encountered (Fig 6.7, A and B). There were some differences in assemblages between intensively and lightly fished areas. Generally, the intensively fished sites, which were slightly shallower (ca 12m chart datum compared to ca 13.5m chart datum for lightly fished sites), had low abundances of fleshy erect red algae (Fig 6.7, A and B). Intensively fished sites also had a greater diversity of Cnidaria (*Tubularia indivisa and Caryophyllia smithii*) and Chordata (*Ciona intestinalis* and *Clavellina lepadiformis*). All differences in assemblages were for species with low abundance (Fig 6.7, A and B).

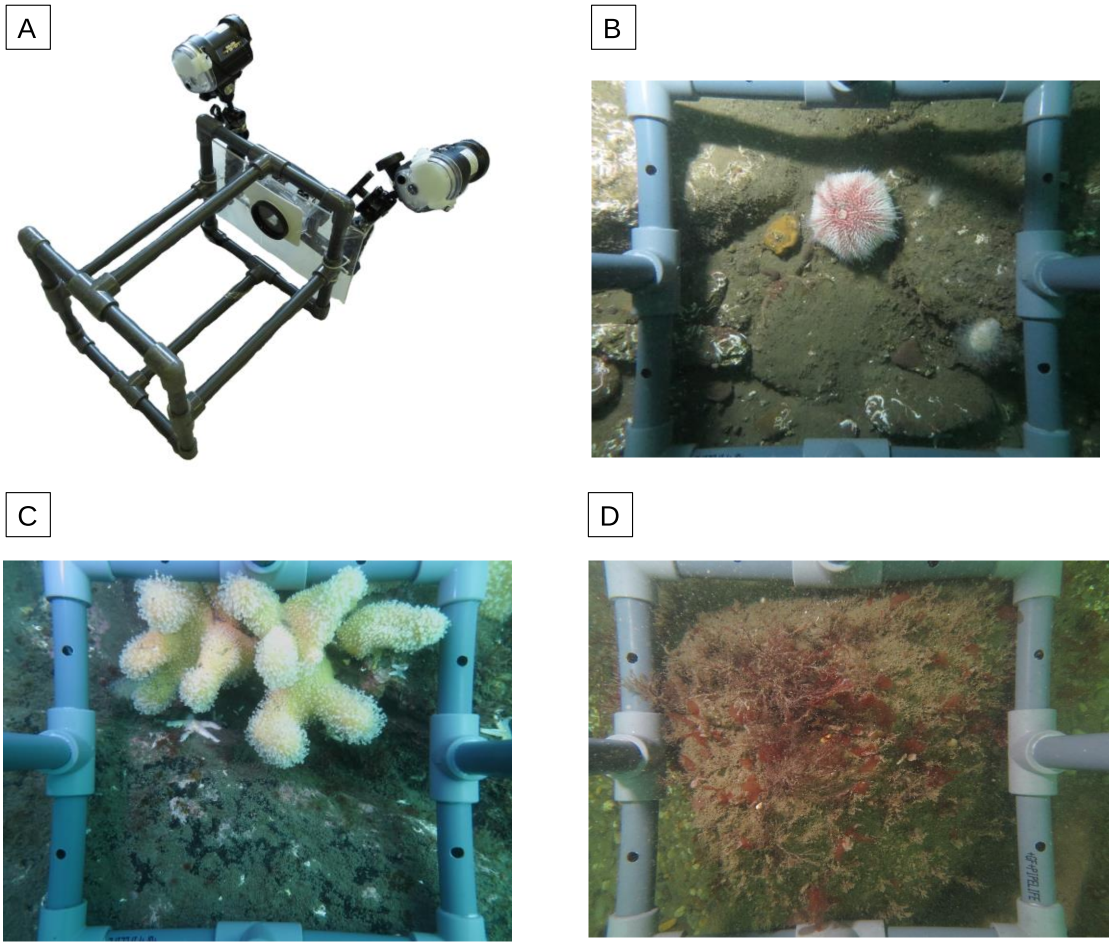


Fig 6.5. Photoquadrat with camera housing and strobes (A). Photoquadrats of typical species from lightly fished FaAlCr sites (B), intensively fished FaAlCr sites (C) and intensively fished Lhyp.Pk sites (D).

The kelp dominated Lhyp.Pk habitat (Connor *et al.*, 2004) was only sampled in intensively fished areas at depths of between 3.5 - 4m (chart datum). Sites were found on hard rocky substrates and were dominated by erect species such as sparse kelp *(Laminaria hyperborea* and *Saccharina latissimi*) with a dense turf of foliose red seaweeds (*Callophyllis laciniata, Plocamium cartilagineum, Delesseria sanguinea, Hypoglossum hypoglossoides* and *Phycodrys rubens*) as well as coralline crusts and faunal crusts (bryozoans and hydroids, *Spirobranchus* spp and *Balanus* spp) (Fig 6.7, C). Feather hydroids and sponges were also frequently encountered although in low abundances (Fig 6.7, C).

6.3.2. Multivariate analysis of potting impacts

The same overall changes were observed in percentage benthos cover between BACI treatments in different habitats and fishing pressures. The non-significant interaction Site x B - C_B indicated assemblages did not differ between Baseline versus Control treatments at all sites (post hoc test all $p > 0.179$) for all habitats and fishing pressures (all $p > 0.072$, Table 6.2). Thus any changes detected between B and I that were not mirrored in the controls (C_B - C_I) could be judged to be due experimental pot fishing.

There was a significant interaction between Site x B - I for all habitats and fishing pressures (all $p > 0.002$, Table 6.2). Post hoc testing confirmed assemblages differed between Baseline and Impact at all sites (all $p < 0.048$) none of which were affected by dispersion (all $p > 0.06$) for FaAlCr and Lhyp.Pk intensively fished habitats. Dispersion was detected for FaAlCr lightly fished habitats ($p < 0.014$). Assemblages also differed between C_B – C_I at all sites for all habitats and fishing pressures (all $p < 0.049$, Table 6.2, no dispersion effects, all $p > 0.094$). However, assemblages did not differ between I – C_I at any sites for all habitats and fishing pressures (all, $p > 0.145$, Table 6.2) indicating that temporal change in community composition between B - I is not attributed to potting impacts.

As dispersion effects were detected for FaAlCr low fished habitats it is unclear whether differences detected were due to location or dispersion difference (Anderson et al., 2008). In order to investigate this further, the PCO centroids of the multivariate data were used to examine the direction and magnitude of changes in assemblages between treatments (Fig 6.6).

Table 6.2. PERMANOVA based on the Bray-Curtis similarity (square root transformed) of multivariate data collected in intensively fished, lightly fished FaAlCr and intensively fished Lhyp.Pk areas (999 permutations per test). Highlighted grey cells show significant p-values. Results are shown for treatments B (baseline), I (impact), C_B (control baseline) and C_I (control impact).

Habitat	Treatment	Source of variation	df	SS	MS	Pseudo F	p	Estimates of components of variation
FaAlCr intensively fished	B - C_B	Site	2	19170	9585.1	17.061	**0.001**	11.3
		Site x B - C_B	2	2018.5	1009.3	1.797	0.072	3.5
		Residuals	206	1115730	561.8			23.7
	B - I	Site	2	13724	6862.1	14.121	**0.001**	9.2
		Site x B - I	2	6562.3	3281.2	6.752	**0.001**	8.6
		Residuals	221	107390	485.93			22.0
	C_B – C_I	Site	2	15200	7599.9	13.125	**0.001**	9.7
		Site x C_B - C_I	2	3837.7	1918.8	3.3139	**0.005**	6.0
		Residuals		125650	579.03			24.0
	I – C_I	Site	2	17899	8949.4	15.304	**0.001**	10.2
		Site x I - C_I	2	1812	906.01	1.5493	0.145	2.8
		Residuals		135670	584.79			24.1
FaAlCr lightly fished	B - C_B	Site	2	2832.1	1416	5.9586	**0.001**	3.8
		Site x B - C_B	2	585.88	292.94	1.2327	0.305	1.6
		Residuals	239	56797	237.65			15.4
	B - I	Site	2	5235.4	2617.7	11.272	**0.001**	5.4
		Site x B - I	2	2439.1	1219.5	5.2515	**0.001**	4.9
		Residuals	237	55038	232.23			15.2
	C_B – C_I	Site	2	4168.5	2084.3	9.6399	**0.001**	4.9
		Site x C_B - C_I	2	2005.2	1002.6	4.6371	**0.001**	4.5
		Residuals	230	49728	216.21			14.7
	I – C_I	Site	2	9065.2	4532.6	21.89	**0.001**	7.5
		Site x I - C_I	2	285.44	142.72	0.68926	0.7	1.3
		Residuals	228	47211	207.07			14.4
Lhyp.Pk intensively fished	B - C_B	Site	2	20 828	10 414	6.2	**0.001**	10.9
		Site x B - C_B	2	5 296	3 447	1.6	0.139	5.1
		Residuals	218	36 8260	1 689			41.1
	B - I	Site	2	10 875	5 437	3.2	**0.002**	7.6
		Site x B - I	2	10 220	5 109	3.0	**0.007**	10.3
		Residuals	195	32 7830	1 681			41.0
	C_B – C_I	Site	2	26 518	13 259	10.0	**0.001**	13.2
		Site x C_B - C_I	2	4 092	4 046	2.8	**0.045**	4.6
		Residuals	206	273 120	1 326			36.4
	I – C_I	Site	2	21 381	10 691	8.4	**0.001**	12.3
		Site x I - C_I	2	3 391	1 695	1.3	0.238	3.7

For all habitats and fishing pressures, changes in multivariate space between treatments were similar. B and C_B samples show clustering by site as do I and C_I (Fig 6.6, A,B and C). The direction and distance between B and I resembled those seen between C_B and C_I at individual sites (Fig 6.6, A,B and C) reinforcing PERMANOVA results (Table 6.2) that community assemblages changed in similar ways in the experimental and control areas suggesting experimental potting impacts were not responsible.

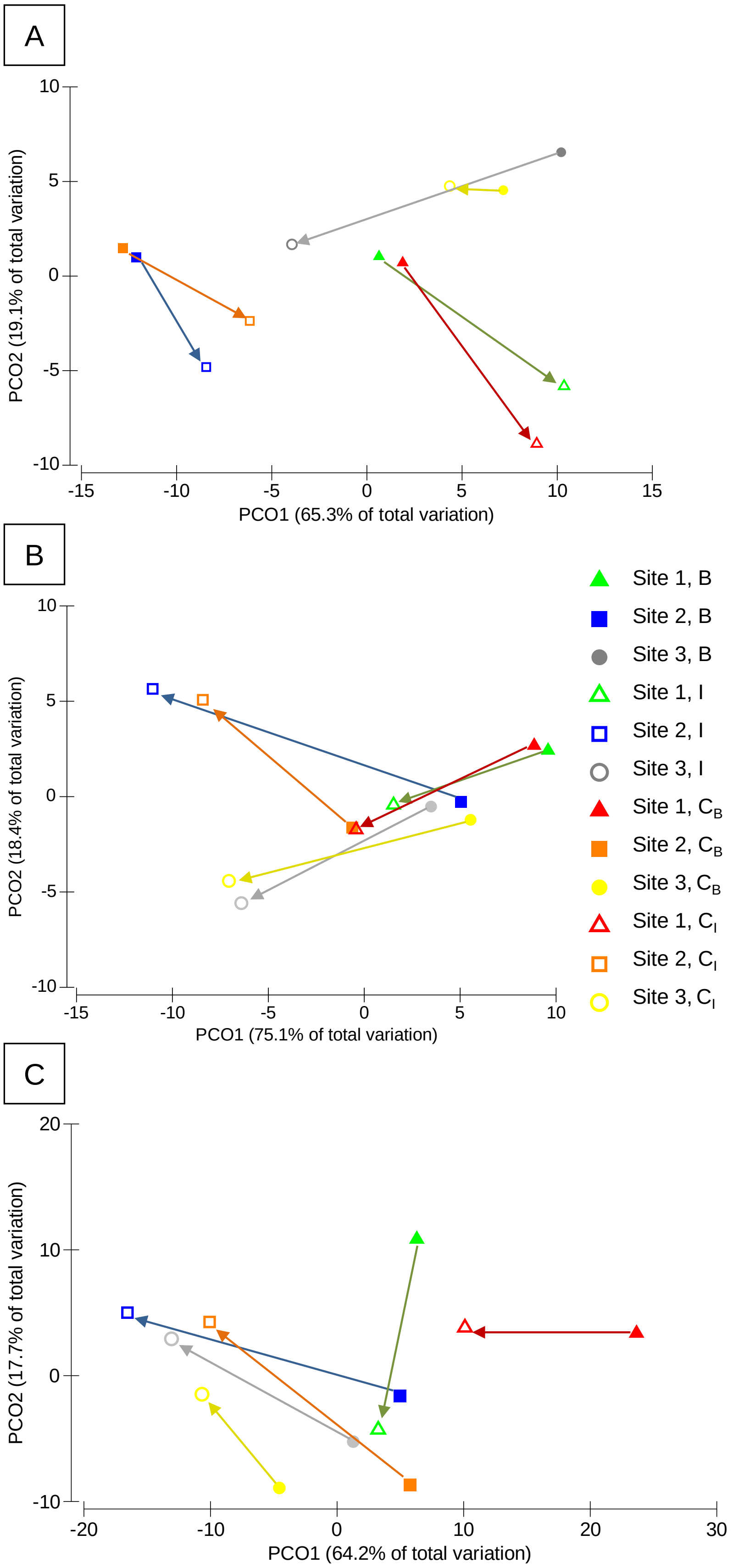


Fig 6.6. Principal coordinate analysis (PCO) of square root transformed percentage cover survey data grouped according to treatment and site for FaAlCr with high fishing pressure (A), low fishing pressure (B) and Lhyp.Pk with high fishing pressure (C). Lines show the direction of change in assemblage structure between B and I, and between C_B and C_I.

6.3.3. Species composition

There were only small differences in overall abundance of individual species between treatments in FaAlCr and Lhyp.Pk (Fig 6.7, A, B and C). Samples from intensively and lightly fished FaAlCr habitats had high cover of faunal and algal turf (often covered in silt), *Spirobranchus* spp, bare rock and gravel (Fig 6.7, A and B). Still frequently occurring but with lower percentage cover were encrusting algae and encrusting bryozoans (Fig 6.7, A and B). All other species were much less frequent.

Samples from intensively fished Lhyp.Pk had high cover of red algae turf (< 1cm), red algae (> 1cm), non-red algae (> 1cm) and short faunal (hydroid, bryozoan) turf often covered in silt (Fig 6.7, C). Frequently occurring but with lower percentage cover were encrusting algae, encrusting bryozoan, bare rock and *Spirobranchus* spp (Fig 6.7, C). All other species were much less frequent (Fig 6.7, C).

Species percentage cover data did not differ greatly pre- and post experimental fishing in either the experimental area or the control areas (Fig 6.7, A, B and C). Individual species' contributions to significant differences between treatments at site level using SIMPER showed the benthos components which were consistent indicators (i.e. contributed more than 10% of the dissimilarity for all sites and dissimilarity/standard deviation ratio > 1 (Terlizzi et al., 2005; Clarkre, 1993)) were the same for each site within habitats and contributed to the differences between treatments in similar ways (see appendix 4). No pattern in the benthos between treatments was consistent with those predicted by potting impacts. These results are consistent with the earlier broader analysis showing that B-I and C_B-C_I differ in similar ways due to either natural variation or habitat heterogeneity.

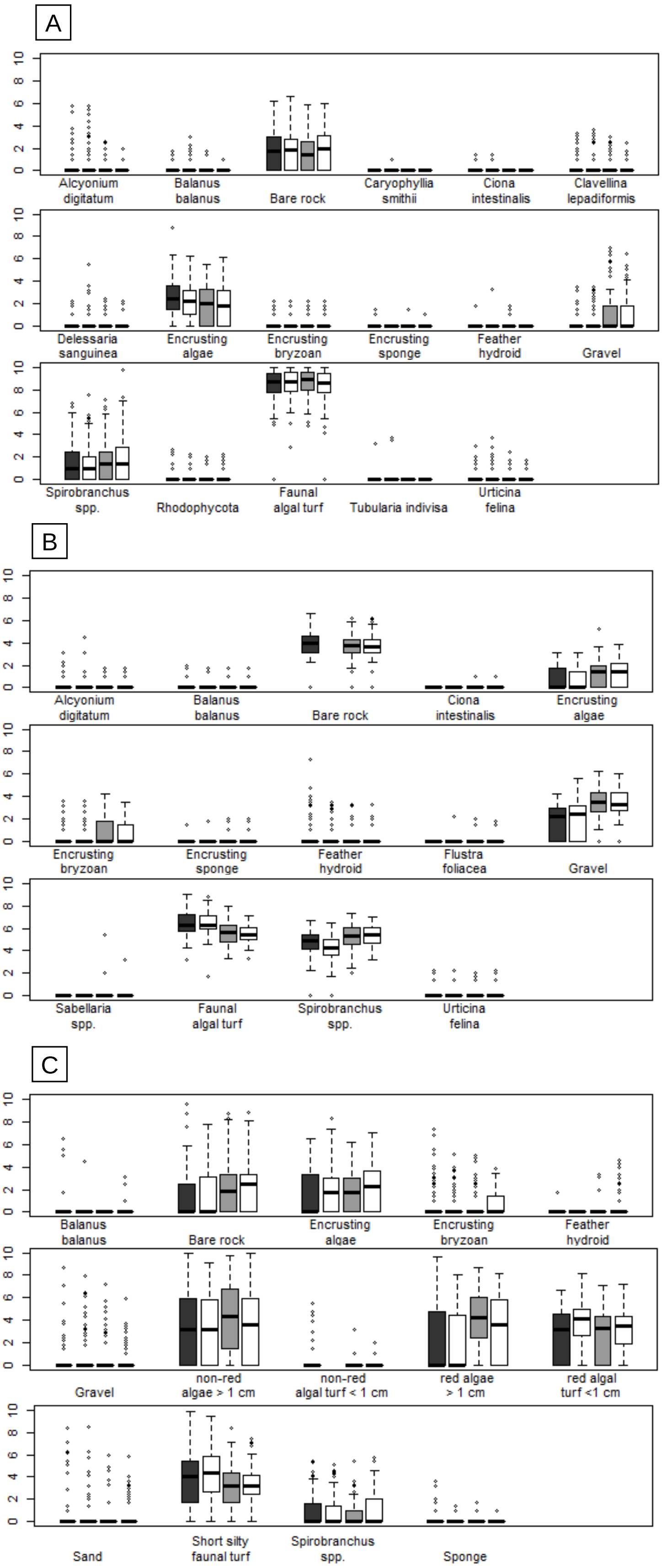


Fig 6.7. Square root transformed percentage cover of benthos from B (dark grey columns), C_B (white column), I (light grey) and C_I (white) experimental areas in intensively fished (A), lightly fished (B) FaAlCr and intensively fished Lhyp.Pk (C) habitats.

6.3.4. *Kelp abundance*

Kelp abundance (*Laminaria hyperborea* and *Saccharina latissimi* > 50cm) was highly variable (Fig 6.8) with no differences found between sites (all p > 0.08) or treatments (all p > 0.17).

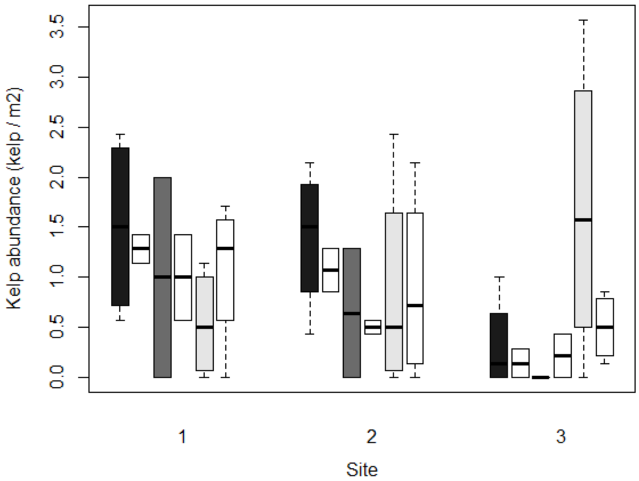


Fig 6.8. Mean in-situ kelp abundance (m^2) across three Lhyp.Pk sites at different experimental treatments. Grey boxes decreasing in shade represent baseline, Impact 1 and Impact 2, respectively. White boxes represent samples from controls for each previous treatment.

6.4. Discussion

Evidence from experimental fieldwork of direct impacts of potting on benthic communities in two different habitats with different fishing pressures will be discussed in the following sections. Robustness, sensitivity and explained variation of the analysis will be explored.

6.4.1. *Direct impacts of potting on epibenthos in Northumberland*

Similarly to previous research into potting impacts in the UK (Eno *et al.*, 2001; Coleman *et al.*, 2013), abundance of erect species did not decline with physical abrasion from pot fishing (rejection of hypothesis 1). This was the case in both intensively and lightly fished FaAlCr and intensively fished Lhyp.Pk habitats, although a shift in community composition was found but attributed to natural change (rejection of hypothesis 3), when considering controls. Repeated potting, over the course of four weeks was conducted in a small area (10 x 10m) at each site, equating to a fishing pressure of 30,000 pots hauled km $^{-2}$ year $^{-1}$, an unrealistically high fishing pressure, compared to even the highest recorded fishing pressure in the

NIFCA district of 1330 pots hauled km $^{-2}$ year $^{-1}$ in 2014 (chapter 2). The physical footprints of static gears such as shellfish pots are much smaller than mobile gears such as trawls and dredges (Nielsen *et al.*, 2013). Indeed, it is very unlikely, due to small size of the pot footprint on the seafloor, that pots fished commercially would land, soak and be hauled, in the same location in successive fishing trips (Eno *et al.*, 2001).

Although the fishing method used in this research was not typical of commercial fishing practices - a single pot, tethered to two weights, compared to a commercial gear setup of a fleet of 10 or more pots tethered to two weights - it was chosen due to the requirement to accurately place the pot in a small study area, allowing a large number of replicates to be sampled over a small distance, providing high statistical power. In addition, due to the use of two 20kg weights and pot in a small experimental area, this fishing method was deemed to be more likely to impact the seafloor than commercial fishing methods. Overall, with the intensity of fishing and potting gear layout, this experiment has far exceeded the limit of maximum likely impact from commercial potting over a short period of time (1 – 2 months), however previous work did not detect long-term impacts (Coleman *et al*., 2013).

Erect species found in intensively and lightly fished FaAlCr and intensively fished Lhyp.Pk habitats, such as *Laminaria hyperborea,* red and brown algae > 1 cm, *Delessaria sanguinea, Flustra foliacea*, *Alcyonium digitatum* and Feather hydroids (e.g. *Nemertesia ramosa*) are all expected to recover within several months (6 – 36) from the mechanical interference, crushing, physical blows against, or rubbing and erosion of a single passage of a standard scallop dredge landing on or being dragged across the organism (Jackson, 2004; Tyler-Walters, 2006; Tyler-Walters and Ballerstedt, 2007; Budd, 2008). Thus, it is likely that given the spaced out nature of repeat potting in any give location by commercial fishers, and the lower-level of impact from potting compared to scallop dredging (Thrush and Dayton, 2002; Coleman *et al.*, 2013; Eno *et al.*, 2013), these species are unlikely to be severely damaged even at the high potting pressures shown here or are expected to recover between fishing events. Encrusting and smaller species such as encrusting bryozoans, sponges, *Balanus* spp and *Spirobranchus* spp, are unlikely to be damaged by pots, their shape and size indicating these organisms can withstand physical disturbance and abrasion. Thus, it is unlikely, given the lack of evidence of short-term direct impacts that longer-term direct potting impacts occur in these habitats. However, rocky reef habitats in Northumberland may already be in a degraded state due to decades of fishing (Scheffer et al., 2001; Nyström et al., 2012; Caveen et al., 2014) and current potting impacts may not be detectable. The species composition of habitats in intensively fished FaAlCr areas, which were dominated by short mixed faunal and algal crust, with few large slower-growing erect fauna and flora, suggests harsh physical conditions (Hartnoll, 1998). Whether these physical conditions are due to long-term potting, environmental conditions, other impacts or a combination of these is unclear. However, local knowledge of the area indicates that abundant sediment, which could cause scour damage to species, and high energy environment from waves and currents (even in summer) are likely to, at least in part, contribute to determining the species composition of the area. Lightly fished FaAlCr habitats were also composed of hardy benthic species further suggesting that environmental conditions may be driving assemblage structure in these areas. Overall, intensively fished FaAlCr areas had a greater overall diversity and higher abundance of large erect species than low fished areas (Fig 6.7), in contrast to the earlier prediction from areas impacted by trawling (Collie *et al.*, 1997; Collie *et al.*, 2000b; Kaiser *et al.*, 2002; Lambert *et al.*, 2011), that sites with high fishing pressure would already be degraded compared to sites with low fishing pressure: lower species richness, lower abundance, smaller individuals or less abundant erect and/or slow growing species (rejection of hypothesis 2). In this case, there is no evidence that differences in community composition between intensively and lightly fished habitats is due to potting. However, as the biotope 'Faunal and algal crusts on exposed to moderately wave-exposed circalittoral rock' has somewhat variable community composition even between areas separated by only hundreds of metres (chapter 5), conclusions on the 'health' of these biotopes and historic impacts of potting would require much larger samples over a greater spatial extent. Investigation of the interplay between environmental variables and anthropogenic uses in the resultant community composition using predictive modelling (similar to Lambert *et al.* (2011)) would be useful in future.

Visual observations of changes in species distribution or impacted areas made during data collection are not always reflected in analyses, either because small impacted areas are not sampled (i.e. because of random sampling), or because small-scale changes may not be detected by analytical models. Throughout this fieldwork no observations were made during dive surveys that indicated potting was having any abrasive action on, or removal of, benthos. Algal species were observed as bending under the weight of the pot (similar to Eno *et al.* (2001) on seapens), rather than being abraded or removed. In addition, pots did not move throughout the duration of the SCUBA dives (approx. 1 hour) suggesting that pot movement does not occur during slack tides.

Laminaria hyperborea (> 50cm) provides habitat and substratum for many organisms, with high biodiversity recorded in kelp habitats (Birkett *et al.*, 1998). A loss of this species from the habitat could result in loss or degradation of the associated community (Tyler-Walters, 2007), as well as potentially changing the classification of the biotope. Here potting did not affect kelp abundance (Fig 6.8), however, recordings of kelp abundance were variable within transects and between treatments (Fig 6.8). This variability in kelp abundance, albeit expected due to the dynamic nature of these habitats where competition for light, space and food results in a species rich but patchy distribution patterns of flora and fauna (Jones *et al.*, 2000), may be masking small changes in abundance due to potting impacts. However, it is clear that potting impacts on epibenthic assemblages cannot be detected, even with high sample numbers, against the background of natural variability, suggesting that these impacts are not pushing the ecosystem outside natural limits (De Backer *et al.*, 2010).

6.4.2. Limitations of the experimental design

Potting impacts were investigated on the two most frequently occurring biotopes in Northumberland (chapter 4). FaAlCr and Lhyp biotopes were recorded a mean 60% and 17% frequency occurrence using a stratified random sampling methodology across the Northumberland fishing district (Mercer, 2012). In addition, FaAlCr habitats are actively targeted by fishers over other habitats (chapter 3). These habitats are deemed good experimental habitats and to be representative of local fishing practices, as well as being present across many parts of Europe (MESH, 2008). However, rarer habitats with smaller spatial extents may be more sensitive to potting impacts, for example *Sabellaria spinulosa* habitats which are present in Northumberland in small areas, are thought to be sensitive to fishing (Holt *et al.*, 1995; Vorberg, 2000; Roberts *et al.*, 2010; Eno *et al.*, 2013). In the region, these habitats are also much more difficult to locate and may not be large enough to allow experimental potting.

This research sought to examine whether potting impacts were greater (or more detectable) in shallower habitats, which may be subject to higher wave action (Lewis et al., 2009). There was no evidence that shallower habitats in Northumberland were more sensitive to potting impacts even though all habitats at all fishing pressures were subject to sustained periods of bad weather (swell > 1.5m over > 7 days). In addition, pots left in shallow water during bad weather did not follow commercial practice as fishers move fleets of pots to deeper water areas in order to avoid damage to their fishing gear (Turner *et al.*, 2012; Coleman *et al.*, 2013), and as such habitats sampled in this research are rarely fished in winter and spring (chapter 2). This study, with high levels of potting effort, maximising possible damage demonstrated that there was no detectable impact.

6.4.3. Accuracy of epibenthic community sampling

Community data extracted from images must be accurate and robust for results and conclusions from this research to be valid. Two of the most commonly used methods for collecting quantitative epibenthic data (spatially and for abundance estimates) for monitoring and impact assessment are *in-situ* quadrat recording and photoquadrats which are processed at a later date (Davies *et al.*, 2001; Sayer, 2007; Van Rein *et al.*, 2011; Moore *et al.*, 2015). The advantages and limitations photoquadrats used for this research will be compared to *in-situ* quadrat recording in the following section.

In circalittoral habitats approximately half as many species were identified from photoquadrat samples compared to *in*-situ records of the same quadrat (Moore *et al.*, 2015) (average taxa recorded per quadrat 15.4 *in-situ*, 9.3 photoquadrat). Although large distinctive species (*Delessaria sanguinea, Flustra foliacea, Clavellina lepadiformis, Alcyonium digitatum*) were consistently recorded between *in-situ* and photoquadrat methods, smaller inconspicuous species were consistently under-recorded in photoquadrat samples (Moore *et al.*, 2015). This is presumably because of the static nature of the image resulting in smaller species being obscured by larger species or due to the quality of photographs (i.e. limited resolution, inadequate illuminations and/or blur) (Moore *et al.*, 2015). Under-recorded species tended to be cryptic (situated in crevices/cracks or undersides of boulders) (Moore *et al.*, 2015). Encrusting bryozoans, sponges, algae and barnacles were frequently recorded in the current study although species level identification was often not possible due to the resolution of the imagery. Photoquadrat images from this research were consistently good quality, with image enhancement in Digital Photo Professional (v 3.14) allowing species to be more easily identified, although it is likely that smaller cryptic species were still under-recorded. However, a consistently low number of identified small inconspicuous species was unlikely to affect the results of this research as these species were unlikely to be affected by abrasive potting impacts if larger more exposed individuals were not.

Although fewer species are recorded in photoquadrats, estimates of species abundance and species number are more consistent between observers than those from *in-situ* sampling (higher Bray-Curtis similarity by approximately 10%) (Moore *et al.*, 2015). This is because more time is available for the identification and accurate abundance estimate (i.e. availability of identification manuals and time to examine the image) in turn lowering the number of misidentifications. In addition, a point intercept method for image analysis was chosen for this research as it is time efficient (taking half the time of other frequency based methods (Van Rein et al., 2012)), unbiased by observers (Aronson et al., 1994; Drummond and Connell, 2005) and is more sensitive to changes in community composition than visual estimation and frequency occurrence methods (Van Rein et al., 2012) further increasing accuracy of recorded species abundance. In order to check the quality and accuracy of species identification in this research, species abundances recorded from 20 images using the point intercept method were compared to those estimated by a second experienced ecologist. No significant differences were found between abundance estimates or the number of species identified.

The primary advantage of photoquadrat sampling is the increased number of samples that can be taken at each site compared to *in-situ* recording. In circalittoral surveys at 20m depth, community data from 2 - 3 quadrats were obtained per surveyor per dive compared to 30 photoquadrat images (Moore *et al.*, 2015). In this research, which was at shallower depth (ca 6 – 14m), 120 images were obtained per dive. This high sample number resulted in a comprehensive coverage of sites as well as increased power for the statistical analysis.

6.4.4. Adequacy of PERMANOVA routine

Samples between treatments were very similar (high Bray-Curtis similarities) for data from FaAlCr intensively and lightly fished areas (Table 6.2). This is also reflected in the PERMANOVA model in the form of very low estimates of components of variation. These suggest there was very little variation in assemblages within the experimental period (Table 6.2). Samples between treatments for intensively fished Lhyp.Pk habitats were less similar than FaAlCr samples, reflected in higher estimates of components of variation suggesting there was variation in assemblages within the experimental period (Table 6.2).

The greatest variation in assemblages in both habitats in intensively and lightly fished areas was the residual variability among replicate photoquadrats (estimates of components of variation for Residuals, Table 6.2). Samples were still very similar for FaAlCr intensively and lightly fished areas, sharing a high number of species with one another (76 – 78%, for FaAlCr intensively fished areas, 84.6 – 85.6% for FaAlCr lightly fished areas) but were more variable for Lhyp.Pk intensively fished areas (40.9 – 60% similarity between samples). Over and above this variation, significant interaction terms Site x B-I and Site x C_B-C_I had low estimates of components of variation of 8.7 and 6.0 respectively for intensively fished FaAlCr areas, even lower estimates of components of variation of 4.9 and 4.5 for lightly fished FaAlCr areas and 7.6 and 13.2 estimates of components of variation for intensively fished Lhyp.Pk areas, suggesting temporal differences (B-I and C_B-C_I) only contributed a small amount towards dissimilarity and variation in the model.

In summary, the robust (balanced) BACI design, analysed using the PERMANOVA routine, resulted in small changes in community composition over time being detected for all sites. This sensitive model allowed conclusions on experimental potting impacts on faunal and algal crust habitats, during summer months to be made with a high degree of confidence.

6.5. Concluding remarks

There was no evidence that experimental potting produced short-term direct impacts on the epibenthos of intensively and lightly fished rocky FaAlCr or Lhyp.Pk habitats over a range of weather conditions (swell: 0 – 2.5 m; tidal current: 0 – 1.3 km.h-^{1}; wind: 0 – 53 km.h-^{1}). Given, the lack of evidence of direct potting impacts, the quick recovery of dominant and characteristic benthic species in both FaAlCr and Lhyp.Pk habitats and the intensity of fishing in the NIFCA district (i.e. pots are unlikely to interact with the same individuals of the benthic community repeatedly over short periods of time), it is also unlikely that long-term (chronic) direct potting impacts occur in these habitats in Northumberland. However, these habitats may already be degraded due to decades of fishing and therefore conclusions only apply to the current state of habitats. No conclusions are made with regards to the sustainability of the current exploitation levels of target species stock or any in-direct effects of potting on habitats and ecosystems in Northumberland. Using results from this study for other temperate locations with similar habitats, assessing fishing effort may be sufficient to determine whether potting impacts are likely to occur. Results from this work are widely applicable to other areas of the UK and Europe due to the focus on potential impacts on abundant and commonly distributed European benthos.

Chapter 7 . Northumberland Inshore Potting Impacts: Synthesis and Recommendations

7.1. Introduction

Habitat quality and health has been recognised as an integral part to management of sustainable fish stocks (Pikitch *et al.*, 2004; Armstrong and Falk-Petersen, 2008; Salomidi *et al.*, 2012). Epifauna which live on rocky reefs are key ecosystem components (Hiscock and Tyler-Walters, 2006) but may be impacted by fishing (Lambert *et al.*, 2011). There is a large body of knowledge for mobile fisheries but little information for the impacts of static gears, including the nationally and locally important trap fishery (Coleman *et al.*, 2013). Using designated marine protected areas in Northumberland as a case study, this research has shown spatial and temporal trends in fishing effort and habitat preferences of potting in Northumberland (2004 – 2014) (chapters 2 and 3). These data are key components of effective EBFM (Crowder and Norse, 2008), and allowed potential potting impacts in areas of interest or of particular vulnerability to be investigated. Chapters 4 and 5 discussed the need for robust data for impact assessments of fishing gears by using previously collected monitoring data. Although results from these chapters add to the evidence base, for the purposes of management these data are incomplete, due to: a lack of abundance data; few temporal data; the high variability of species composition used to classify biotopes (UK Biotope classification, Connor et al., 2004) potentially masking changes between years; and a lack of in-depth, localised, baseline knowledge of the area. These issues were addressed in chapter 6 where direct impacts of experimental potting were observed *in situ*, following a carefully designed methodology. The high fishing intensity in small experimental areas coupled with high levels of sampling and replication, provided robust evidence that current levels of potting are unlikely to have a direct physical impact on epibenthos in faunal and algal crust, and *Laminaria* spp dominated habitats in Northumberland coastal waters. No differences were found in benthic community assemblage structure between areas which were previously fished at different intensities, although conclusions on current levels of stock or indirect effects of fishing were not within the scope of this research. This final chapter aims to summarise key findings, outlining the management implications of these and providing recommendations for future studies investigating similarly fine-scale impacts.

7.2. Implications for management of changes in fishing effort distribution and habitat use over time

Changes in fishing effort distribution between years were found across large sections of the study area between years (chapter 2), highlighting the high inter-annual variability of fishing effort over time at a regional scale (Lynch, 2014). This further highlights the usefulness of monitoring fisheries over long-time periods because ‘snap shots’ of the fishery may lead to under or over estimates of chronic fishing effort (Lynch, 2014). Increases were concentrated close to the shore, particularly around the larger ports of the district, presumably to target high value lobster.

Results from this research reinforce the usefulness of combining landings data with fishing spatial distribution. Cumulative changes in fishing effort distribution provided greater detail on temporal trends than either analysis of fishing effort data (chapter 2, section 3.1) or vessel distributions independently (chapter 2, section 3.2). For example, these data demonstrated that fishing effort did not increase uniformly across the district but has become highly concentrated inshore, especially from 2010 – 2013. These maps highlight areas which may be damaged due to repeated high fishing efforts or may be sensitive to impacts (Eno *et al.*, 2013). For example, the inshore of the study area were prioritised for research of physical impacts of fishing on the seafloor (chapter 6).

The high resolution (1km^2) maps allowed temporal trends of inshore potting to be investigated in detail. Although VMS provides more detailed information on vessel movement (i.e. VMS units are required to report 99% of all vessel GPS locations with an accuracy of at least 500m as well as data on heading and speed (Defra, 2005c; Defra, 2005b)), the tendency for inshore shellfish fleets to be composed of small vessels (< 10m, which don’t gather VMS data) operating close to shore, means fishing activity of such vessels is not indicated (Breen *et al.*, 2014). This is particularly the case in Northumberland, where a mean of 70 (± 9) % < 10m fishing vessels operated annually in the NIFCA district between 2004 – 2013. The methods used to map fishing effort could be readily applied to other parts of the UK, where the sightings and effort data used are also routinely collected by fisheries enforcement agencies.

Further long-term monitoring of inshore fishing effort is recommended because fishing effort trends were shown to change between years (chapter 2). Little information on the drivers for changes in fishing pressure are available for the UK or specifically for Northumberland. Further social research conducted through key informant interviews would allow drivers to be explored in greater depth, and is recommended.

The increase in potting effort, specifically inshore, shows the importance of understanding potential impacts on epibenthos. Results from this research provided estimates of fishing effort, allowing both estimates for district wide differences in species assemblages to be investigated (chapter 4 and 5) as well as focused assessments of experimental potting impacts (chapter 6).

Assessments of full impacts of a fishery require basic data that describes the health, quality and status of individual habitats and species, as well as data for fine-scale distributions of ongoing fishing activities (Pedersen *et al.*, 2009; Eno *et al.*, 2013). Building on results from chapter 2, which provided information on potting density, chapter 3 elucidated temporal changes in fishers' habitat selection at different classification scales (EUNIS level 3 – 6), with data resolution of $1m^2$. Space-time clustering was found for fishers operating in the CQSM MCZ, suggesting that these were actively targeting habitats of interest. Compositional analysis of habitat showed that fishers preferred rocky habitats over sediment habitats when using both EUNIS level 3 and 6 habitat maps, although fishing effort (pots fished year $^{-1}$ km $^{-2}$) was similar between rocky habitats at EUNIS level 5-6.

The combination of fishing effort and habitat-use data over time is necessary in order to investigate fine-scale potting impacts (Chapter 4, 5 and 6), as well as providing data which can underpin evidence based EBFM. Results from this research suggest that fishers distribute themselves within their territories (Turner *et al.*, 2012) according to habitat type. This has implications for fisher redistribution due to fisheries closures – an aspect of marine spatial planning that has not been investigated for inshore fisheries in the UK to date. This may in part be due to the lack of available in-depth cross-disciplinary data i.e. reliable habitat and temporal fishing distributions. The effects of fishery closures due to offshore development or marine protected areas will be gear specific (Campbell *et al.*, 2014). The impact of displacement of pot fishers due to areal closures, may be greater than those attributed to mobile gear users because potting is so highly localised (Campbell *et al.*, 2014; Guenther *et al.*, 2015; Turner *et al.*, 2015), further demonstrated by this research through persistent use of inshore areas over the course of a decade (chapter 2). When fishery closures are enforced, spillover is frequently assumed to occur (McClanahan and Mangi, 2000; Howarth *et al.*, 2011). This assumes that the increase in target species abundance within the MPA will cause migration of target individuals outside the closed area (Hoskin *et al.*, 2011; Howarth *et al.*, 2015) and that the displaced adjacent fisheries will focus fishing effort along the edges of the MPA as catches will be highest there, a phenomenon called “fishing the line” (Guenther *et al.*, 2015). However, such models do not incorporate habitat specific fishing type and effort, habitat type, heterogeneity of the underlying habitat, or the limits imposed by fishing methods/gear. For example, pot fishers target rocky habitat over softer sediment habitats (chapter 3) presumably to target lobster (Turner *et al.,* 2009). Thus habitat outside theoretical closed areas may not be suitable for the displaced fishers if rocky reefs are preferentially selected for closure. In addition, fixed-gear fisheries where pots are set, soaked, pulled and reset in the same location results in areas that are “marked” or occupied, and stop other fisherman fishing in the same location. This has resulted in high levels of territoriality amongst pot fishers (Acheson, 1975; Turner *et al.*, 2012). Displacement of potting effort may increase disputes, the effects and severity of which, will largely be determined by *a priori* territorial distributions and habitat quality (Guenther *et al.*, 2015). After fishing closures in inshore areas in California (USA), lobster fishermen targeted areas based on competition with other fishers and habitat quality (i.e. catch), rather than targeting areas close to MPA borders (Guenther *et al.*, 2015). Potting is currently permitted in the two large protected areas in the NIFCA district: CSQM MCZ and BNNC EMS. These areas are the most heavily fished inshore areas (chapter 2) and is therefore crucial to understand how any future management could affect fishers’ displacement in Northumberland. Multicriteria decision analysis tools such as Marxan are increasingly used to combine a variety of different spatially explicit selection criteria in a GIS for marine reserve planning and renewable energy developments (Baban and Parry, 2001; Villa *et al.*, 2002; Lieberknecht *et al.*, 2004; Bruce and Eliot, 2006; Prest *et al.*, 2007). These tools primarily focus on the distribution of the natural resources important for development or conservation purposes, often ignoring fishing distribution data which can minimise environmental and economic costs of area closures by displacement (Dinmore *et al.*, 2003; Lieberknecht *et al.*, 2004; Richardson *et al.*, 2006; Campbell *et al.*, 2014), resulting in wider acceptance, cost-effective and politically feasible protected areas (Richardson *et al.*, 2006; Campbell *et al.*, 2014). Although results of this research provide a crucial step towards a better understanding of long-term fishing effort and spatial distribution in Northumberland, potentially enabling a predictive capacity for Marxan type modelling, a lack of access to cumulative uses, socio-economic information and habitat data outside the CQSM MCZ raises scientific and socio-economic concerns about the underpinning of marine spatial management decisions (Campbell *et al.*, 2014). These data would be particularly important for further investigation of closure effects in the NIFCA district as potting is the most important inshore fishery in Northumberland; highest potting vessel sightings per unit effort in England and Wales (Vanstaen and Breen, 2014).

7.3. Pragmatically detecting ecological change over time to support monitoring.

7.3.1. Biotope analysis

The number and range of biotopes observed in this study was maintained between years through the persistence of few, heavily dominating biotopes, namely infralittoral kelp (Lhyp) and circalittoral faunal and algal crust (FaAlCr) biotopes with only non-significant differences of rare (low frequency counts) biotopes. Although the investigation of biotope richness and persistence of dominant biotopes between years suggests that ecosystem health was maintained throughout the BNNC EMS, it is a simple analysis both in terms of the data required and the statistical analysis used (Davies *et al.*, 2001; Saunders *et al.*, 2011). In this particular case, conclusions that can be drawn from this data are limited due to the broad nature of biotope analysis and the low number of sampling years (2002/03 and 2011).

This type of data however, may be better suited for use in mapping geographic extent of biotopes (chapter 3). This allows the limited biological information (temporally and spatially) available for this research to be combined with acoustic data (AGDS, SSS or MBES) and mapped (Lucieer, 2008). Specifically this would allow further information to be gathered on specific biotopes that may be increasing / decreasing spatially as well as investigate potential causality of any changes (Crowder and Norse, 2008). Use of habitat maps combined with information on their associated human uses and political and legal arrangements are recommended as a more holistic approach to monitoring spatial change in marine ecosystems (Salomidi *et al.*, 2012) specifically for anthropogenic activities that vary in spatial scale such as fishing (Diesing *et al.*, 2009) (chapters 2 and 3).

7.3.2. Sample number and scale of biotope analysis

Biotope analysis can be undertaken at different scales ranging from very broad habitat description (level 2) to sub-biotope (level 6). Conclusions in this report are similar when using either biotopes (level 5) or sub-biotopes (level 6) with no change detected between years. However, the use of biotope data for low-level fishing impact studies remains problematic and a null result in this case may be due to methodological issues rather than a lack of biotope change between years. The use of a broader biological classification level (e.g. level 5), although only likely to detect spatially and ecologically extensive impacts, results indicating change may be more robust as small scale fluctuations of natural variability are already largely taken into account. In contrast, the use of a more detailed biological classification level (level 6) may result in detection of smaller scale impacts, however, in order to discount the effects of natural variability in biotopes, much larger data sets are required both spatially and temporally and little information exists for UK habitats (Hartnoll, 1998).

Although conclusions on biotope change were not possible due to the limited temporal data, sampling effort was sufficient to allow robust, albeit simple, statistical analysis. In addition, biotope richness curves showed that biotope sampling for the Boulmer transect was more than adequate (chapter 4). In contrast, sampling the Farnes Island transect may not fully represent biotope richness. Future sampling effort applied could be adjusted depending on biotope richness in a similar way to that already routinely done for assessments of species richness (Gotelli and Colwell, 2001). Sampling effort curves produced after preliminary sampling would enable researchers to estimate the number of samples required in different areas, saving money and providing appropriate results during a monitoring campaign (Gotelli and Colwell, 2001). The use of biotope richness as a method of detecting change, although relatively broad, may still be an important metric for investigation of long-term ecosystem changes because these may be the only data that can be extracted from older records, thus providing adequate temporal sampling. However, results from this research suggest that for investigation of small-scale impacts such as those expected from potting, changes in biotope richness would not be precise enough to detect impacts.

Overall, for investigation into small-scale fishing impacts, such as potting, the use of a biotope classification system alone is not recommended. This broad scale does not allow changes in abundance, species diversity or species composition for each biotope to be taken into account. Ecological quality of biotopes within the BNNC EMS has not been assessed using this method. The maintenance in quality of features of interest is likely to be a critical aspect of future monitoring programmes as part of DEFRA’s revised approach to fisheries management in EMSs. However, previous methods used in EMS condition monitoring do not allow this to be investigated. However, the use of biotopes may provide a useful overview that can help focus on particular habitats or species affected as part of a wider program of research. Detection of change within biotopes was further explored in chapter 5 and conclusions are discussed below.

7.3.3. Change in taxonomic composition, species richness and indicator species between years.

Overall, there was very little evidence to suggest that taxonomic composition of assemblages, species richness or sensitive species of rocky reefs in the BNNC EMS changed between 2002/03 -2011, in spite of increasing fishing pressure. However, across all analyses the level of uncertainty was high, with increased spatial and temporal sampling required (see chapter 5). Specifically, the high number of samples required due to the complexity of the model (3-5 factors) meant that only the most commonly found biotopes had adequate sample sizes for robust testing. There is some evidence that species composition and richness of FaAlCr, the most commonly identified biotope in the study, may have changed between 2002/03 – 2011, but it was unclear, whether this was natural variation or change due to anthropogenic impacts. Further work is required to establish levels of natural annual and seasonal variation within this biotope. Understanding the magnitude and direction of natural variation, which can be cyclical and linear, is crucial for effective monitoring of the marine environment (Davies *et al.*, 2001) and increased research in this field has been encouraged previously (Hartnoll, 1998; Nordheim and Boedeker, 1998), although evidence remains sparse.

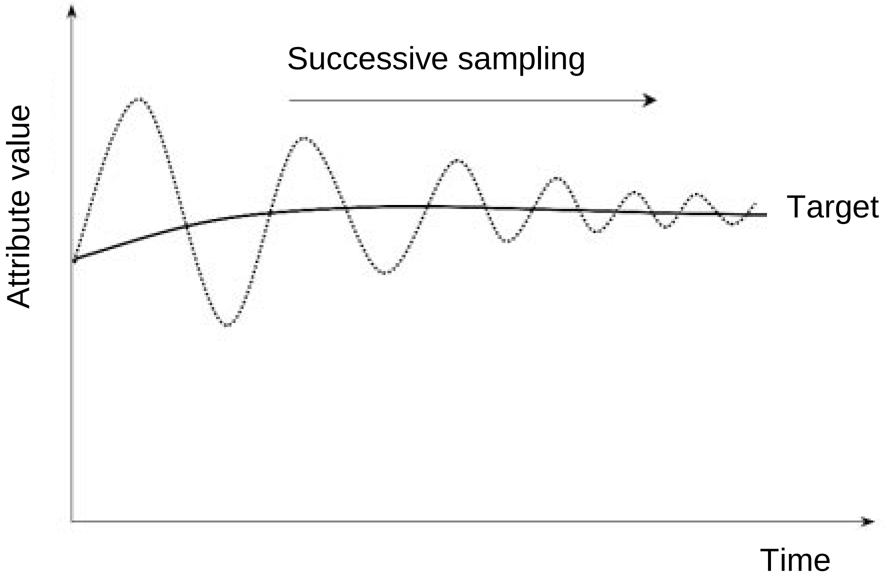


Fig 7.1. Attribute target value over time (Diagram modified from Davies, 2001). Hypothetical example of estimates of target value (dashed line) and target value (solid line) based on the number of available samples.

Fig 7.1 shows a hypothetical example of how an attribute's target value, defined as an aspect of the feature of interest, e.g kelp abundance, which is used to infer condition of the target feature, may be refined over time (Davies *et al.*, 2001). If the target value fluctuates (i.e. has high variability), successive surveillance cycles enable a more accurate determination of the confidence limits (dashed line) and hence the target value (solid line) (Fig 7.1). The target value may initially be located at the upper confidence limit and therefore be inaccurate, with subsequent data collection resulting in a substantial decrease in its final value (Fig 7.1). This example taken from monitoring guidelines highlights the importance of adequate temporal sampling and requirement of an understanding of yearly variability. It should be noted that in this example, attribute target value is a univariate measure and multivariate measurements (i.e. taxonomic community assemblages) will be more complex as the direction and magnitude of change for each target value may change in different ways. A recent approach to monitoring whole ecosystems uses ‘trajectories in ecosystem state space’ with change visualized as Euclidean distance (Tett *et al.*, 2013) (Fig 7.2). However, this novel approach is underpinned by having long term data with appropriate sampling. At present monitoring data for the BNNC EMS do not allow this type of analysis.

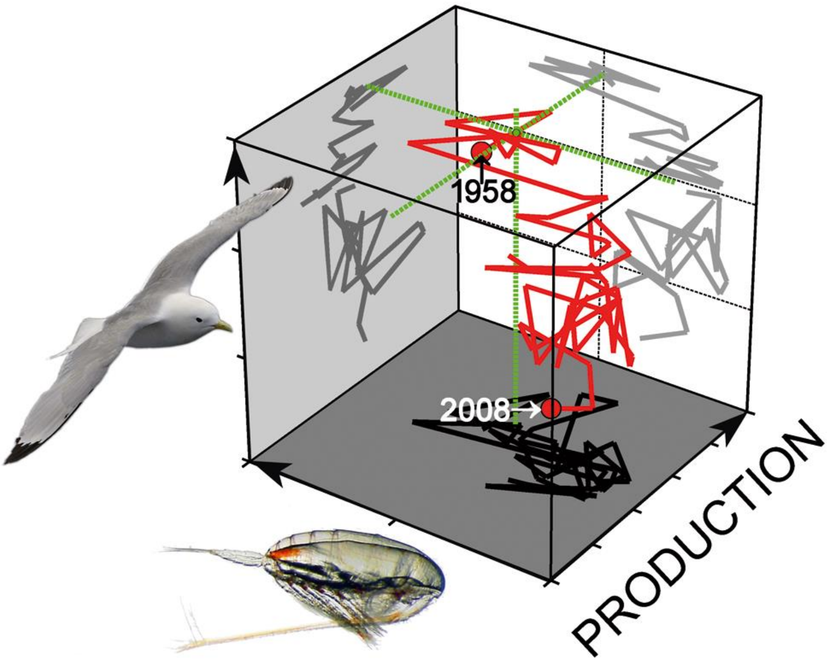


Fig 7.2. Changes in the northern North Sea, 1958–2008, plotted in a state space defined by the breeding success of kittiwakes (y-axis), abundance of copepods *Calanus* spp (x-axis), and simulated annual primary production (z-axis) (Tett *et al.,* 2013).

The same differences were found between analyses using taxonomic composition of assemblages and sensitive species, thus a reduced species list using indicator species may be advantageous due to the lowered cost of sampling and effort in identification. Devising a list of sensitive species for each biotope specifically for pot fishing would allow quicker more focused surveys to be conducted. This may allow more samples to be acquired and processed during surveys, increasing their statistical robustness (Dayton et al., 1998; Hartnoll, 1998). However, unexpected changes to other species may occur which may be overlooked if only a reduced species list is being monitored (Coleman et al., 2013) (chapter 6).

Further, the scale at which changes in taxonomic composition of assemblages of biotopes were investigated here, i.e. at transect or area level, altered results, suggesting biotope composition variability between transects. The BNNC EMS area-wide model did not have as many significant differences as individual transect models suggesting area-wide models may be masking impacts or change. Therefore, although greater sampling effort and increased analytical time is required, using individual transect data to investigate even large changes is recommended as area-wide data may mask potential impacts or change. For investigation into fine-scale potting impacts, transect level (~ 25km^2) may still be too broad. Indeed, rigorous investigation into ecosystem impacts of potting (Coleman *et al.,* 2013) may not have found any evidence of impact due to the similarly large scale at which these were undertaken. Investigation over smaller spatial scales, such as permanent transects, similar to those used by Eno *et al*. (2001), may be more adequate and is explored in chapter 6.

7.3.4. Attributing change to fishing pressure

Fishing pressure is not continuous throughout the BNNC EMS (chapter 2). Finding the optimum levels of potting that crab and lobster populations and the habitat can support is a key requirement for management purposes (Eno *et al.*, 2001; CEFAS, 2014). In this research, using historic monitoring data, there was no evidence that different potting intensities had any effect on floral and faunal community composition of rocky reefs in the BNNC EMS between 2002/03 – 2011. Some evidence of differences in species richness (*S)* in biotopes FaAlCr and FaAlCr.Bri at different fishing pressures was gained but no obvious pattern was apparent and results were inconclusive due to low temporal replication. In order to explore this question fully, robust evidence of change is needed requiring increased spatial and temporal sampling (chapter 5). In addition, in order to incorporate fishing pressures as a factor in the model, additional replicates in each biotope at different fishing pressures are required (Fig 7.3).

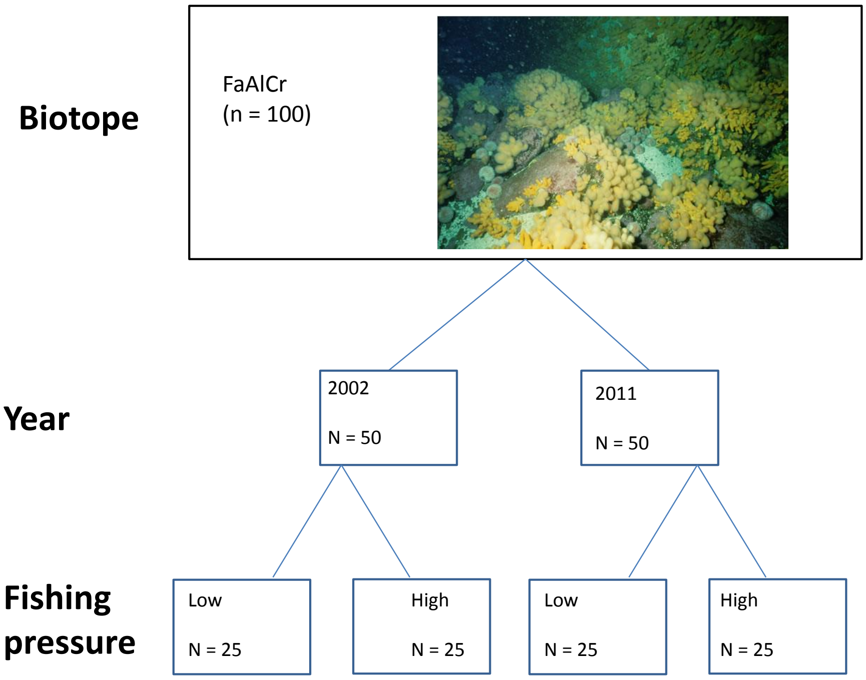


Fig 7.3. Schematic diagram of example sample number for each biotope with year and fishing pressure as factors.

The addition of factors and the number of levels these would have requires an increased number of samples to have the same explanatory power (Fig 7.3). For example, 3 levels of fishing pressure as opposed to 2 shown would result in n = 16 for each level. Using historic monitoring data, this was not a problem for the most abundant biotopes (i.e FaAlCr (n > 250) and Lhyp.Ft (n > 100)) but many biotopes sampled less frequently (Chapter 4) did not have enough samples to run robust tests even at a BNNC EMS area-wide level i.e. combining all transect data. This illustrates the need for careful planning and knowledge of the data requirements for statistical tests needed to investigate the effects of fishing pressure on habitat and species composition. The simple metric of species richness suggested that some changes were detected between years and fishing pressures. This merits further investigation which could easily be investigated as part of a more focused investigation into direct impacts.

As illustrated in previous sections, large datasets are needed to investigate long-term changes in biotopes and attribute these to fishing pressure (Lambert *et al.*, 2011). Alternatively potting effects on biotopes could be investigated by examining direct impacts of pots *in situ* (chapter 6).

7.4. Adequacy of monitoring data for scientific investigation

Results from chapter 4 and 5 add to the evidence base, but for the purposes of management these historic monitoring data are incomplete due to the lack of abundance data, inconsistent seasonal sampling and poor visibility, which resulted in a less detailed analysis and low confidence in results preventing robust conclusions.

Abundance data are a prerequisite for detailed ecological analyses of this kind (Carbines and Cole, 2009). It is therefore recommended that the use of monitoring data for such work is inadequate unless abundance estimates are possible. Encouragingly, development in this methodology and a reduction in price of the hardware has resulted in more recent monitoring frequently including methods to estimate abundance (CEFAS, 2012).

Due to issues associated with samples taken from different seasons (summer 2002 and autumn 2011) and the low amount of annual sampling, this data cannot be used to determine conclusively if anthropogenic change has occurred in the BNNC EMS. However, monitoring data may be useful in order to investigate areas which warrant particular interest or further work. This type of exploratory work is cost efficient because the data already exist and could result in more focused studies. This can also include learning valuable lessons on videographic methods and avoiding pitfalls in data analysis. For example the present work highlighted differences in FaAlCr between years which warranted further investigation (chapter 6).

The effect of visibility on species and biotope identification requires a revised method to produce more robust results. This would include devising a visibility scale based on the measurement of visibility for the biotope as a whole, for example, by recording percentage of time in minutes / seconds that substrate is identifiable. In addition, other factors such as light levels and turbidity would ideally also be taken into account. It should be noted that camera pitch, roll and speed of movement is related to the camera setup. A larger and heavier camera, such as that used by CEFAS (2012), would be much more stable and travel more slowly than the smaller camera system used for this research. However, logistical issues arise from using larger, heavier, camera systems such as the need for winches and larger boats to operate these systems which often cannot operate in shallower water (< 10m deep). The selection of a drop-down video system will rely on cost, vessel equipment and size.

Although costly at approximately USD 100,000 the underwater camera system described by Rosenkranz *et al.* (2008) is an example of best available equipment.

7.5. Observation of direct potting impacts on common reef habitats

Direct potting impacts on epibenthos were investigated through observation of experimental fishing in faunal and algal crust (FaAlCr) and *Laminaria hyperborea* (Lhyp.Pk) dominated habitats which had been subjected to known fishing effort over the course of 10 years (chapter 2), allowing effects of historical fishing pressure to be taken into account in the impact analysis. The lack of statistical power for detecting impacts of potting gear which has a small physical footprint (≈ $1m^2$) in comparison to the large deployment area (≈ $200m^2$), were overcome through fishing of a single pot tethered to two weights in a small experimental area (10 x 10m), allowing high sampling of photoquadrats (120 images per transect) and replication.

Similarly to previous research into potting impacts, no effects were detected in this research (Eno *et al.*, 2001; Coleman *et al.*, 2013). There was no evidence of abundance of erect and presumably fragile species (Roberts *et al*., 2010) decreasing due to physical abrasion from pot fishing. This was the case in both high and low fished FaAlCr and high fished Lhyp.Pk habitats; a shift in community composition was found but this was attributed to natural change. Overall, the high fished FaAlCr habitat had a greater overall diversity and higher abundance of large erect species compared to low fished sites, contrary to the earlier prediction that, similarly to areas impacted by trawling (Collie et al., 1997; Collie et al., 2000; Kaiser et al., 2002; Lambert et al., 2011), sites with high fishing pressure would already be degraded compared to sites with low fishing pressure (i.e. lower species diversity, lower abundance, smaller individuals or less abundant erect and/or slow growing species).

As with all ecological studies not all possible conditions and habitats were investigated. Limitations of the experimental design included: short experimental period (pots were only fished over the course of a summer month), although unseasonal weather conditions were experienced throughout fieldwork (swell: 0 – 2.5m; tidal current: 0 – $1.3km\ h^{-1}$; wind: 0 – $53km\ h^{-1}$) thus increasing the likelihood of pot movement and damage (Lewis *et al.*, 2009); potting impacts were only investigated on two common habitats, other habitats may be more sensitive (e.g. *Sabellaria spinulosa* biogenic reefs) although these may be difficult to locate (chapter 3); depth restrictions due to the use of scuba for sampling and the need to collect high sample numbers resulted in only shallow sites selected for study, deeper habitats may react differently to potting impacts, although fragile habitats in deeper water have already been investigated by Eno et al. (2001) in different locations in the UK with no damage detected; only impacts in summer were monitored, impacts may be greater in winter due to more adverse weather (Lewis *et al.*, 2009), however, poor visibility in the water associated with adverse weather makes this difficult to assess (Van Rein *et al.*, 2011).

In summary, given the lack of evidence of direct potting impacts, the quick recovery of dominant and characteristic benthic species in both FaAlCr and Lhyp.Pk habitats and the intensity of fishing in the NIFCA district (i.e. pots are unlikely to interact with the same individuals of the benthic community repeatedly over short periods of time), it is unlikely that short-term direct potting impacts occur in habitats with similar community composition.

7.6. Power analysis of potting impacts and future research

Determining ecologically significant impacts, in the context of legislation, is often a complex question, and little guidance on how this can be done is available (IEEM, 2006). Many protected areas in the UK allow activities within their limits provided that these activities do not adversely affect site integrity or undermine the achievement of conservation objectives (MMO, 2012). Site integrity is defined as “the coherence of its ecological structure and function, across its whole area, that enables it to sustain the habitat, complex of habitats and/or the levels of populations of the species for which it was classified” (Department for Communities and Local Government, 2005). Integrity can be measured in several ways, one of which is the average level of populations that would be considered 'acceptably characteristic of the site or ecosystem' (Department for Communities and Local Government, 2005). However, the exact level of potting impacts at which the reduction of abundance of populations is deemed as affecting site integrity is not known. Power analysis of epibenthic abundance data from experimental fishing on common habitats in the BNNC EMS (chapter 6) allows exploration into the number of samples needed to detect a given potting effect size on abundance for aid in developing future monitoring programs.

Power analysis, a useful prospective planning tool for designing experimental studies, can provide appropriate sample size to be determined: large enough to allow robust conclusions to be made whilst small enough to avoid potentially wasting resources with excess sampling (Ryan, 2013). Prospective power analysis is not available for multivariate data. However, univariate power analysis on a reduced species (i.e. sensitive species or species of interest) list can provide evidence on detection of a given effect size of potting impacts. Comparison of univariate and multivariate data for the use in detection of fine scale impacts is here discussed as well as the usefulness of a reduced species (i.e. indicator species) for detection of fine scale impacts.

7.6.1. Power analysis methods

Abundance of individual characterising (Faunal and algal crust, encrusting algae, *Spirobranchus* spp) and sensitive species (*Clavellina lepadiformis, Alcyonium digitatum*, Feather hydroid and *Sabellaria* spp) were extracted from the combined B and C_B data for all high and low fished FaAlCr sites (chapter 6). Characterising species are defined as a "species characteristic of the biotope (dominant, highly faithful and frequent) which are important for the classification of that biotope" (Hiscock and Tyler-Walters, 2006). These species are important as a loss or degradation could result in community shift and loss of biotope (Tyler-Walters, 2002). Species sensitive to potting were chosen due to being erect, and/or easily snapped or shattered upon impact and long-lived (Eno *et al.*, 2001; Roberts *et al.*, 2010). Sensitive species are important in maintaining healthy ecosystems that remain resilient to environmental pressures (Tyler-Walters, 2002).

Mean, standard deviation and effect size (i.e. 20, 10, 5% difference in mean abundance) were calculated for each species' abundance. Data for all benthos were arcsine transformed (Crawley, 2005) but was not normally distributed (with the exception of the very abundant faunal and algal crust). The lack of normality was not deemed to be due to low sample size (n = 102 – 125). This meant any analysis of univariate data (single species) would be undertaken using non-parametric methods, in this case a Mann–Whitney U test (Ryan, 2013). Non-parametric tests are less powerful than the equivalent parametric test, with the Asymptotic relative efficiency (ARE) ratio describing the difference in power (Serfling, 2011; Ryan, 2013). The ARE of the Mann–Whitney U test relative to the t-test cannot be less than 0.864 regardless of the distribution (Conover, 1980). This equates to Mann–Whitney U test requiring an increase in sample size of 15.7% (1/0.864 = 1.157) to ensure adequate power. All estimate sample size calculations were undertaken in RStudio (v.

0.97.237) using standard deviation and effect size calculated from collected data and power (1 – beta) set to 0.8 (Cohen, 1988). Resulting sample number was then increased by 15.7% in order to account for the difference in power between parametric and non-parametric models.

7.6.2. Power analysis results

Univariate two-sample tests of individual benthic cover did not differ between baseline (B and B_C) and impact (I and B_I) (all species $p > 0.05$, Mann-Whitney). Only the most common characterising benthos (mean cover > 8%), namely, faunal and algal crust (in high and low fished areas), encrusting algae (in high fished areas), bare rock (low fished areas) and *Spirobranchus* spp (low fished areas), showed that any changes in abundance of 10% or more would likely be detected with the amount of sampling undertaken for this experimental work (n = 102 – 125). Less abundant sensitive species required much larger sample sizes to detect any changes in abundance (Table 7.1 and Table 7.2); very rare species such as *Sabellaria* spp required a predicted 5555 samples to detect a 20% change in percentage cover (Table 7.2).

Table 7.1. Sample number required to detect a change of 20%, 10% and 5% in mean abundance (N = 102) of characterising species and sensitive species (highlighted in grey) for sites with high fishing pressure.

Benthos	**Mean (% cover)**	**Standard deviation**	**Abundance change**	**Sample Number**
Faunal and algal crust	74.08	19.73	20%	16
			10%	57
			5%	220
Encrusting algae	8.78	10.35	20%	119
			10%	470
			5%	1873
Bare rock	5.75	7.48	20%	180
			10%	714
			5%	2851
Spirobranchus spp	5.77	10.15	20%	356
			10%	1416
			5%	5649
Clavellina lepadiformis	0.66	2.05	20%	1075
			10%	4294
			5%	17268
Feather hydroid	1.19	4.71	20%	1900
			10%	7595
			5%	29654
Alcyonium digitatum	0.06	0.42	20%	1737
			10%	6943
			5%	27627

Table 7.2. Sample number required to detect a change of 20%, 10% and 5% in mean abundance (N = 124) of characterising species and sensitive species (highlighted in grey) for sites with low fishing pressure.

Benthos	**Mean (% cover)**	**Standard deviation**	**Abundance change**	**Sample Number**
Faunal and algal crust	43.62	14.25	20%	3
			10%	41
			5%	159
Encrusting algae	1.65	2.41	20%	326
			10%	1298
			5%	5189
Bare rock	15.98	7.40	20%	20
			10%	71
			5%	278
Spirobranchus spp	23.05	9.69	20%	19
			10%	68
			5%	264
Gravel	5.48	5.06	20%	129
			10%	512
			5%	2022
Alcyonium digitatum	0.24	1.10	20%	1577
			10%	6680
			5%	23828
Sabellaria spp.	0.28	2.70	20%	5555
			10%	22212
			5%	74653
Feather hydroid	1.39	5.77	20%	1776
			10%	7098
			5%	28384

7.6.3. Usefulness of reduced species lists

Univariate testing of species data raises important considerations with regards to sample size and statistical methods for monitoring and impact assessment. Firstly, the univariate Mann-Whitney test was less sensitive for detecting change than the multivariate approach used in chapter 6 as no differences were found between treatments for any species when using the univariate model. In addition, it is estimated that even for the most abundant species in this study, the number of samples collected would only allow changes in abundance of > 10% to be detected. Whether changes in abundance of 10% for these species would be deemed as affecting site integrity is not known. For future work, it is recommended that the more powerful permutation multivariate tests are used for investigation into small scale changes (Terlizzi *et al.*, 2005).

A reduced species list using indicator species may be advantageous due to the lowered cost of sampling and effort in identification; devising a list of sensitive species for each biotope specifically for pot fishing would allow quicker surveys, more samples to be acquired and processed during surveys and increased statistical robustness (Dayton et al., 1998; Hartnoll, 1998). However, unexpected changes to other species may occur which may be overlooked if only a reduced species list is being monitored (Coleman et al., 2013). In addition, when using univariate tests, selecting species which are deemed important (i.e. characterising or sensitive) must be decided beforehand. As there is no evidence in this study (or others, Eno *et al*., 2001; Coleman *et al.,* 2013) that any of the sampled species are sensitive to potting impacts, univariate testing will be less useful than multivariate testing for change detection. Creating a sensitive species list for monitoring potting impacts was attempted in prior research (Eno *et al*., 2001; Coleman *et al.,* 2013), however, these species will only be valid for local areas surrounding the experimental site as species composition and abundance can vary greatly within the UK. For example, infrequently encountered species in one location may be abundant in others. In addition, further information on the level of impacts which would cause the loss of site integrity is needed if any small scale impacts are found in future; for the present work this aspect was not investigated as no impacts were detected.

7.7. Concluding remarks

Previous research has focused on large-scale mobile gear seabed impacts and has resulted in significant knowledge gaps with regards to smaller-scale static gear impacts such as those possible from potting. The gap in evidence is particularly problematic as it is a prerequisite for monitoring that ensures the sustainability of inshore fisheries. Further, with the increasing protection afforded to important habitats such as reefs, it is important to understand any impacts that fishing may have on these crucial habitats. This research has provided evidence required for appropriate assessment of potting in Northumberland with methods that could be adapted for other locations in Europe and the UK with similar habitats. Following this, pre-existing monitoring data could be used for a secondary purpose: the investigation of long-term potting impacts, and as such produce evidence as to whether potting impacts habitat and epibenthos, therefore avoiding costly new data collection. However, results from this research suggest that for investigation into small-scale potting impacts, condition monitoring data have limitations. Although little evidence of potting impacts were detected on biotopes (chapter 4), taxonomic community assemblages, sensitive species or species richness (chapter 5), these results are not robust due to the data used having a lack of abundance estimates, inconsistent seasonal sampling and poor visibility. Furthermore, this research has highlighted that natural variability of biotopes and communities varies significantly spatially and temporally within the BNNC EMS. As such, increased sampling would be required in order to adequately investigate the complicated interactions between species change in different habitats and under different fishing pressures.

Recommendations derived from this work and from scientific literature enabled a more focused approach to fishing impacts on commonly found habitats in Northumberland to be taken. The influence of historical fishing pressure was assessed, as well as cumulative impacts of experimental fishing using a robust BACI design with adequate sample size and replication. In line with previous studies (Eno *et al.* 2001; Coleman *et al.,* 2013), these results suggest that potting is unlikely to directly impact epibenthos; no conclusions are made with regards to the sustainability of the stock or indirect impacts of potting.

Overall, this research has highlighted the need for further monitoring of spatial fishing effort and habitat use in order to inform on-going changes to management – although it is thought that at current potting levels, impacts are very unlikely to affect “site integrity”. In addition, previous monitoring methods are likely to be ineffective in all but the largest impact detection – data collection that allows a multivariate approach is recommended for future monitoring work.

Appendices

Appendix 1

Code used in Rstudio:

Changes in fishing effort over time

Code of the negative binomial regression model using estimates of 14.3% extra fishing effort attributable to vessels >10m between 2006 – 2009

```
library(AER)
library(MASS)

R_GLM <- read.csv("~/PhD/GIS/FISHING_PRESSURE/Landings/R_GLM.csv")

pot <- R_GLM$pots
ContMonth <-R_GLM$month
active <-R_GLM$active.vessels
n <- length(ContMonth)

Model<- glm.nb(pot~ ContMonth + active, data=R_GLM)
summary(Model2)
plot(Model2)

#plot data with line of best fit from coefficients of the model
pred.month1 <- seq(1, 168, 1)
pred.pot1 <- predict(Model, list(wt = pred.month1), type = "response", se.fit=TRUE)
LL1 <- pred.pot1$fit - 1.96 * pred.pot1$se.fit
UL1 <- pred.pot1$fit + 1.96 * pred.pot1$se.fit

plot(pot ~ ContMonth, xlab = "Months", ylab = "Number of Pots Fished", ylim=c(0,
40000))
lines (pred.month, LL1)
lines (pred.month, UL1)
polygon(c(ContMonth, rev(ContMonth)),c(UL1, rev(LL1)), col = rgb(0,0,0,0.25), border
= NA)
lines (pred.month1, pred.pot1$fit)
```

Code and results of the negative binomial regression model using lower estimate of 8.5% for 2006 – 2009

```
R_GLM_lower <-
read.csv("~/PhD/GIS/FISHING_PRESSURE/Landings/R_GLM_lower.csv")
```

```
pot.l <- R_GLM_lower$Pots
ContMonth <-R_GLM$month
active <-R_GLM$active.vessels
n <- length(ContMonth)

Model<- glm.nb(pot.l~ ContMonth + active, data=R_GLM)
summary(Model)
```

Regression coefficients, standard errors, t - value, and p-values significance for each variable using a negative binomial regression model with lower estimates for 2006 – 2009 pots fished..

	Estimate	**Std Error**	**z - value**	**p- value**
Intercept	8.5	0.07	124.4	< 0.0001
Months	0.005	0.0002	17.91	< 0.0001
Active vessels	0.012	0.0006	19.62	< 0.0001

Code and results of the negative binomial regression model using upper estimate of 20.1% for 2006 – 2009

```
R_GLM_upper <-
read.csv("~/PhD/GIS/FISHING_PRESSURE/Landings/R_GLM_upper.csv")

pot.u <- R_GLM_upper$Pots
ContMonth <-R_GLM$month
active <-R_GLM$active.vessels
n <- length(ContMonth)

Model<- glm.nb(pot.u~ ContMonth + active, data=R_GLM)
summary(Model)
```

Regression coefficients, standard errors, t - value, and p-values significance for each variable using a negative binomial regression model with lower estimates for 2006 – 2009 pots fished..

	Estimate	**Std Error**	**z - value**	**p- value**
Intercept	8.6	0.07	124.6	< 0.0001
Months	0.005	0.0003	16.15	< 0.0001
Active vessels	0.010	0.0007	16.1	< 0.0001

Changes in fishing effort distribution over time.

```
library(raster)
library(rgdal)
```

```
r1 <- raster("raster_fishing_pressure_map_year1")
r2 <- raster("raster_fishing_pressure_map_year2")
g1 <- sampleRegular(r1, 5e5, cells=TRUE, sp=TRUE)
g2 <- sampleRegular(r2, 5e5, cells=TRUE, sp=TRUE)

g1.data <-na.omit(g1)
g2.data <-na.omit(g2)

totreps <- 50000
samplesize <- 5000

out <- vector(length=totreps)
for (rep in 1:totreps) {
 sample.rows <- sample(1:NROW(g1.data), size=samplesize, replace=FALSE)
 data1.tmp <- g1.data[sample.rows,]
 data2.tmp <- g2.data[sample.rows,]
 ttest <- t.test(data1.tmp[,2], data2.tmp[,2], paired=TRUE)
 out[rep] <- ttest$statistic
}
hist(out, xlim=c(-5,25), xlab="t - value", main=NULL)
abline(v=qt(0.05, samplesize-1), col="red")
print(length(out[out>qt(0.05, samplesize-1)]))
```

Appendix 2

R code:

```
# Fishers' sightings spatial clustering and habitat use
# Author Fabrice Stephenson
# Modified from: EFB AFB spatial Clustering by Aileen Mill
# Date: 23th Jun 2015

library(splancs)
library(sp)
library(rgdal)

#read in data
p <- readOGR(“~Habitat", layer="CQSM_MCZ")
plot(p)
poly.gon<- getpoly()
plot(poly.gon)
```

```
pts.all <- read.csv(“~Sightings.csv", header=T, na.strings="*")
pts.1 <- read.csv("~Sightings.1.csv", header=T, na.strings="*")
pts.2 <- read.csv("~Sightings.2.csv", header=T, na.strings="*")
pts.3 <- read.csv("~Sightings.3.csv", header=T, na.strings="*")
pts.4 <- read.csv("~Sightings.4.csv", header=T, na.strings="*")
pts.5 <- read.csv("~Sightings.5.csv", header=T, na.strings="*")
scale.spat <- seq(0, 11000, by=250)
#year 1
pts.sp1 <- cbind(pts.1$X, pts.1$Y)
khat.obs1 <- khat(pts=pts.sp1, poly=as.matrix(poly.gon), s=scale.spat)
#simulation envelope year 1
V.khat <- Kenv.csr(nptg=500, poly=as.matrix(poly.gon), nsim=500, s=scale.spat)
khat.sp1 <- data.frame(khat.obs1, lower=V.khat$lower, upper=V.khat$upper)
rand.L1 <- sqrt(khat.sp1/pi) - scale.spat
#plot output of year 1
plot(scale.spat, rand.L1$khat.obs1, xlim=range(0,10000), ylim=range(rand.L5),
xlab="Distance (m)", ylab="L value", type="l", col="Grey", lwd=2)
lines(scale.spat, rand.L1$upper, lty=2)
lines(scale.spat, rand.L1$lower, lty=2)
peak <- scale.spat[which.max(rand.L1$khat.obs1)]
print(peak)
#year 2
pts.sp2 <- cbind(pts.2$X, pts.2$Y)
khat.obs2 <- khat(pts=pts.sp2, poly=as.matrix(poly.gon), s=scale.spat)
#simulation envelope year 2
khat.sp2 <- data.frame(khat.obs2, lower=V.khat$lower, upper=V.khat$upper)
rand.L2 <- sqrt(khat.sp2/pi) - scale.spat
#plot output
plot(scale.spat, rand.L2$khat.obs2, xlim=range(scale.spat), ylim=range(rand.L2),
type="l", col="green2", lwd=2, ann=FALSE)
lines(scale.spat, rand.L2$upper, lty=2)
lines(scale.spat, rand.L2$lower, lty=2)
peak <- scale.spat[which.max(rand.L2$khat.obs2)]
print(peak)
#year 3
pts.sp3 <- cbind(pts.3$X, pts.3$Y)
khat.obs3 <- khat(pts=pts.sp3, poly=as.matrix(poly.gon), s=scale.spat)
```

```
#simulation envelope year 3
khat.sp3 <- data.frame(khat.obs3, lower=V.khat$lower, upper=V.khat$upper)
rand.L3 <- sqrt(khat.sp3/pi) - scale.spat
#plot output
plot(scale.spat, rand.L3$khat.obs3, xlim=range(scale.spat), ylim=range(rand.L3),
type="l", col="green2", lwd=2, ann=FALSE)
lines(scale.spat, rand.L3$upper, lty=2)
lines(scale.spat, rand.L3$lower, lty=2)
peak <- scale.spat[which.max(rand.L3$khat.obs3)]
print(peak)
#Year 4
pts.sp4 <- cbind(pts.4$X, pts.4$Y)
khat.obs4 <- khat(pts=pts.sp4, poly=as.matrix(poly.gon), s=scale.spat)
#simulation envelope year 4
khat.sp4 <- data.frame(khat.obs4, lower=V.khat$lower, upper=V.khat$upper)
rand.L4 <- sqrt(khat.sp4/pi) - scale.spat

#plot output
plot(scale.spat, rand.L4$khat.obs4, xlim=range(scale.spat), ylim=range(rand.L4),
type="l", col="grey2", lwd=2)
lines(scale.spat, rand.L4$upper, lty=2)
lines(scale.spat, rand.L4$lower, lty=2)
peak <- scale.spat[which.max(rand.L4$khat.obs4)]
print(peak)
#year 5
pts.sp5 <- cbind(pts.5$X, pts.5$Y)
khat.obs5 <- khat(pts=pts.sp5, poly=as.matrix(poly.gon), s=scale.spat)
#simulation envelope year 5
khat.sp5 <- data.frame(khat.obs5, lower=V.khat$lower, upper=V.khat$upper)
rand.L5 <- sqrt(khat.sp5/pi) - scale.spat
#plot output
plot(scale.spat, rand.L5$khat.obs5, xlim=range(scale.spat), ylim=range(rand.L5),
type="l", col="green2", lwd=2, ann=FALSE)
lines(scale.spat, rand.L5$upper, lty=2)
lines(scale.spat, rand.L5$lower, lty=2)
peak <- scale.spat[which.max(rand.L5$khat.obs5)]
print(peak)
```

```
#plot output of all years on single graph
plot(scale.spat, rand.L1$khat.obs1, xlim=range(0,9000), ylim=range(rand.L5),
xlab="Distance (m)", ylab="Measure of Spatial Aggregation, L", type="l", col="Grey5",
lwd=2)
lines(scale.spat, rand.L5$khat.obs5, col="green3", lwd=2)
lines(scale.spat, rand.L2$khat.obs2, col="red3", lwd=2)
lines(scale.spat, rand.L3$khat.obs3, col="Blue3", lwd=2)
lines(scale.spat, rand.L4$khat.obs4, col="Yellow3", lwd=2)
lines(scale.spat, rand.L1$upper, lty=2)
lines(scale.spat, rand.L1$lower, lty=2)

#temporal clustering
pts.all <- read.csv("~Sightings.csv", header=T, na.strings="*")
pts.sp <- cbind(pts.all$X, pts.all$Y)
pts.tm <- pts.all$Year

scale.spat <- seq(0, 10000, by=250)
scale.time <- seq(1,5, by=5)

rand.kh <- stkhat (pts=pts.sp, times=pts.tm, poly=as.matrix(poly.gon),
c(1,5),seq(0,10000,250),seq(1,5,1))
rand.se <- stsecal(pts=pts.sp, times=pts.tm, poly=as.matrix(poly.gon),
c(1,5),seq(0,10000,250),seq(1,5,1))
rand.mc <- stmctest(pts=pts.sp, times=pts.tm, poly=as.matrix(poly.gon),
c(1,5),seq(0,10000,250),seq(1,5,1),nsim =500 ,quiet=FALSE)
stdiagn<- stdiagn(pts=pts.sp, rand.kh, rand.se, rand.mc)
```

Appendix 3

Information on species important to communities within different biotopes was obtained from the Marine Life Information Network (MarLIN). Change or loss of these species would indicate a deterioration of biotope health. Information on sensitivity of biotopes, including species selected and rationale are shown below.

Biotope: CR.MCR.EcCr.FaAlCr.Pom

Species name	Community importance	Description of community importance
Echinus esculentus	Key functional	Biotope has intense urchin grazing. Loss of species would result in major changes in the community. i.e. shift to *Alcyonium* or *Metridium* dominated communities.
Alcyonium digitatum	Important characterizing	Important characterizing species since it occurs in most records of the biotope
Lithophyllum incrustans	Important other	Included to represent the encrusting coralline algae characteristic of the biotope
Pomatoceros triqueter	Important other	Characteristic faunal crusts

Biotope: CR.MCR.EcCr.FaAlCr

Species name	Community importance	Description of community importance
Echinus esculentus	Key functional	Urchin grazing (although less than FaAlCr.Pom). Responsible for keeping species richness relatively low.
Alcyonium digitatum	Important characterizing	Important characterizing and structural species and is included to represent the sensitivity of large epifauna.
Lithophyllum incrustans	Important other	Included to represent the encrusting coralline algae characteristic of the biotope
Nemertesia spp, *Thuiaria thuja*	Important other	Characteristic robust hydroids present in biotope
Flustra foliacea	Important other	Characteristic faunal species
Pomatoceros triqueter	Important other	Characteristic faunal crusts

Biotope: CR.MCR.EcCr.FaAlCr.Adig

Species name	**Community importance**	**Description of community importance**
Echinus esculentus	Key functional	Urchin grazing (although less than FaAlCr.Pom). Responsible for keeping species richness relatively low.
Alcyonium digitatum	Important characterizing	Important characterizing and structural species and is included to represent the sensitivity of large epifauna.
Lithophyllum incrustans	Important other	Included to represent the encrusting coralline algae characteristic of the biotope
Pomatoceros triqueter	Important other	Characteristic faunal crusts

Biotope: CR.MCR.EcCr.FaAlCr.Bri

Species name	**Community importance**	**Description of community importance**
Ophiothrix fragilis, *Ophiocomina nigra,* *Ophiura albida*	Key structural	May form dense beds providing habitat for other species and a food source for commonly found starfish
Asterias rubens	Important Functional	Predation upon Brittlestar beds may control abundance although *Asterias rubens* is commonly found in this biotope.
Alcyonium digitatum	Important characterizing	Important characterizing species since it occurs in most records of the biotope
Abietinaria abietina, *Thuiaria thuja*	Important characterizing	Only robust hydroids able to tolerate the significant 'smothering' effect from the dense `mat' of brittlestars.

Biotope: CR.MCR.EcCr.FaAlCr.Flu

Species name	**Community importance**	**Description of community importance**
Echinus esculentus	Key functional	Urchin grazing (although less than FaAlCr.Pom). Responsible for keeping species richness relatively low.

Flustra foliacea	Key structural	Provides a habitat for species and is characteristic of this biotope (if lost would result in loss of the biotope as described).
Alcyonium digitatum	Important characterizing	Important characterizing and structural species and is included to represent the sensitivity of large epifauna.
Nemertesia sp., *Thuiaria thuja*	Important characterizing	Included to represent clumps of hydroids that occur in this biotope. Although, loss of a single species may not be detrimental, loss of the bryozoan/ hydroid turf would result in degradation of the community, and potentially loss of the biotope as described.
Pomatoceros triqueter	Important other	Characteristic faunal crusts

Biotope: IR.MIR.KR.Lhyp.Ft

Species name	**Community importance**	**Description of community importance**
Laminaria hyperborea	Key structural	Provides the major biological structure to the biotope. Provides substratum for numerous species and is the major source of primary production in this community, either directly or in the form of drift (broken off) algae. If lost, the biotope would cease to be Lhyp.Ft.
Echinus esculentus	Key functional	High densities of urchin grazing can result in loss of kelp ('urchin barrens'). The presence of this biotope is partly reliant on low or no populations of sea urchins. Although their grazing may prevent dominance by any one species of understorey algae, resulting in a more species rich epifauna/flora.
Delesseria sanguinea, *Palmaria palmata,* *Plocamium sp.*	Important characterizing	Representative of the red foliose algae that survive in the biotope

Biotope: IR.MIR.KR.Lhyp.Pk

Species name	**Community importance**	**Description of community importance**
Laminaria hyperborea	Key structural	Provides the major biological structure to the biotope. Although is less dense in Lhyp.Pk. Provides substratum

		for numerous species and is the major source of primary production in this community, either directly or in the form of drift (broken off) algae. If lost, the biotope would cease to be Lhyp.Pk.
Echinus esculentus	Key functional	High densities of urchin grazing can result in loss of kelp and red algae ('urchin barrens'). The presence of this biotope is partly reliant on low populations of sea urchins. Although their grazing may prevent dominance by any one species of understorey algae, resulting in a more species rich epifauna/flora.
Alcyonium digitatum	Important other	Important characterizing and structural species.
Delesseria sanguinea, *Palmaria palmata,* *Plocamium sp.*	Important characterizing	Representative of the red foliose algae that survive in the biotope
Lithophyllum incrustans	Important characterizing	Is visually dominant on the rock

Biotope: IR.MIR.KR.Lhyp.GzPk

Species name	**Community importance**	**Description of community importance**
Laminaria hyperborea	Key structural	Although less dense than Lhyp.Pk the kelp in this biotope still provides the major biological structure to the biotope. The kelp stipes may or may not be grazed. In the most extremely grazed areas, the stipes are also devoid of seaweeds. More usually, however, the stipes offers a refuge from grazing, and are characterised by dense turfs of red seaweeds,
Echinus esculentus	Key functional	This biotope often has (or has recently had) high densities of urchins, which, through grazing, have substantially altered the community structure by removing most of the seaweeds and leaving only coralline crusts on the rock.
Alcyonium digitatum	Important other	Important characterizing and structural species and is included to represent the sensitivity of large epifauna.
Delesseria sanguinea, *Palmaria palmata,*	Important characterizing	Representative of the red foliose algae that survive in the biotope (although at lower densities than Lhyp.Pk)

Plocamium sp.		
Lithophyllum incrustans	Important characterizing	Is visually dominant on the rock
Pomatoceros triqueter	Important characterizing	Is visually dominant on the rock and represents the barren nature of the substrate in this biotope.

Biotope: IR.MIR.KR.LhypT.Ft

Species name	**Community importance**	**Description of community importance**
Laminaria hyperborea	Key structural	Provides the major biological structure to the biotope. Provides substratum for numerous species and is the major source of primary production in this community, either directly or in the form of drift (broken off) algae. If lost, the biotope would cease to be LhypT.Ft.
Echinus esculentus	Key functional	High densities of urchin grazing can result in loss of kelp ('urchin barrens'). The presence of this biotope is partly reliant on low or no populations of sea urchins. Although their grazing may prevent dominance by any one species of understorey algae, resulting in a more species rich epifauna/flora.
Delesseria sanguinea, *Palmaria palmata,* *Plocamium sp.*	Important characterizing	Representative of the red foliose algae that survive in the biotope
Membranoptera *Alata, Obelia* *Geniculate, Membranipora membranacea*	Important characterizing	Holdfast and stipe fauna is a particularly species rich part of the biotope. These species have been suggested as specifically associated with holdfasts and stipes and therefore critical to the identity of this biotope.

Biotope: IR.MIR.KR.LhypT.Pk

Species name	**Community importance**	**Description of community importance**
Laminaria hyperborea	Key structural	Provides the major biological structure to the biotope. Although is less dense than LhypT.Ft. Provides

		substratum for numerous species and is the major source of primary production in this community, either directly or in the form of drift (broken off) algae. If lost, the biotope would cease to be LhypT.Pk.
Echinus esculentus	Key functional	High densities of urchin grazing can result in loss of kelp and red algae ('urchin barrens'). The presence of this biotope is partly reliant on low populations of sea urchins. Although their grazing may prevent dominance by any one species of understorey algae, resulting in a more species rich epifauna/flora.
Alcyonium digitatum	Important other	Important characterizing and structural species and is included to represent the sensitivity of large epifauna.
Delesseria sanguinea, *Palmaria palmata,* *Plocamium sp.*	Important characterizing	Representative of the red foliose algae that survive in the biotope
Membranoptera *Alata, Obelia* *Geniculate, Membranipora membranacea*	Important characterizing	Holdfast and stipe fauna is a particularly species rich part of the biotope. These species have been suggested as specifically associated with holdfasts and stipes and therefore critical to the identity of this biotope.
Lithophyllum incrustans	Important characterizing	Is visually dominant on the rock

Appendix 4

Intensively fished FaAlCr

Contributions of individual species to significant differences detected between treatments on a site level, showed similarities between sites for all treatments (SIMPER, Table A.1).

Table A.1. Mean benthic cover contributing more than 10% to Bray-Curtis dissimilarity between treatments (SIMPER), in order of contribution for high fished FaAlCr habitats. Only consistent indicator benthic cover was used. Difference in abundance between treatments are shown. Mean dissimilarity in samples between B-I and CB-CI are shown for sites. Grey cells indicate difference in direction of mean abundance.

Site	Species	B	I		C_B	C_I	
		Av.Abund	Av.Abund	± Av.Abund	Av.Abund	Av.Abund	± Av.Abund
Site 1	*Spirobranchus* spp	2.28	6.35	4.07	2.86	8.29	5.44
	Bare rock	5.11	5.95	0.85	3.39	4.49	1.11
	Encrusting algae	4.28	5.81	1.52	3.13	3.31	0.18
	Faunal algal turf	73.10	63.84	-9.26	66.10	63.52	-2.58
		Dissimilarity (%)		31.2	Dissimilarity (%)		36
Site 2	Encrusting algae	2.13	2.76	0.62	1.93	2.96	1.03
	Bare rock	1.39	0.81	-0.58	1.49	2.13	0.64
	Spirobranchus spp	0.35	1.00	0.65	0.20	1.10	0.90
	Faunal algal turf	86.12	82.63	-3.49	86.86	82.45	-4.42
		Dissimilarity (%)		27.8	Dissimilarity (%)		27.5
Site 3	*Spirobranchus* spp	5.15	1.96	-3.19	3.28	3.46	0.18
	Encrusting algae	12.60	4.84	-7.76	8.76	5.29	-3.47
	Bare rock	3.84	1.54	-2.30	4.93	6.50	1.57
	Faunal algal turf	60.37	76.56	16.19	64.80	64.96	0.16
		Dissimilarity (%)		32.3	Dissimilarity (%)		36.2

Benthos which were consistent indicators and contributed more than 10% of the dissimilarity for all sites were similar between treatments although varied in order of contribution between sites (Encrusting algae (red, brown, pink), *Spirobranchus* spp, Bare rock and Faunal algal turf) (Table A.1). Faunal algal turf displayed the largest differences in mean benthos percentage cover between B-I and C_B-C_I although these were not consistent across sites, ranging from increases of 16.19 % (B-I, site 3, Table A.1) to decreases of 9.26% (B-I, site 1, Table A.1). Other benthos differed by lower percentage cover between B-I and C_B-C_I (< 7.8%, Table A.1).

A comparison of changes in mean percentage cover of benthos between B-I and C_B-C_I showed that these were also similar between treatments with only three mean abundances differing in direction between treatments (show in grey in Table A.1, Bare rock, site 2; *Spirobranchus* spp, site 3; Bare rock, site 3). There is no pattern in increase/decrease of taxon between treatments that are consistent throughout sites with those predicted by potting impacts. These results are consistent with the earlier broader analysis showing that B-I and C_B-C_I differ in similar ways.

Lightly fished FaAlCr

SIMPER analysis, undertaken to investigate in more detail contributions of individual benthos to significant differences detected between treatments on a site level, showed similarities between sites for all treatments (Table A.2.).

Table A.2. Mean percentage benthic cover contributing more than 10% to Bray-Curtis dissimilarity between treatments (SIMPER) for low fished FaAlCr habitats. Only consistent indicator benthic cover was used. Difference in abundance between treatments are shown. Mean dissimilarity in samples between B-I and CB-CI are shown by sites. Grey cells indicate difference in direction of mean abundance.

Site	**Species**	**B**	**I**		**C_B**	**C_I**	
		Av.Abund	Av.Abund	± Av.Abund	Av.Abund	Av.Abund	± Av.Abund
Site 1	Gravel (stone or shell)	2.72	7.78	5.06	2.07	7.84	5.7
	Spirobranchus spp	18.83	25.2	6.36	20.34	27.14	6.8
	Faunal algal turf	49.98	38.93	-11.04	47.33	33.40	-13.92
	Bare rock	13.24	13.98	0.73	13.76	12.60	-1.16
	Encrusting algae	0.7	0.79	0.08	0.72	0.65	-0.06
	Feather hydroid	0.44	0.03	-0.41	N/a	N/a	N/a
		Disimilarity (%)		21.7	Disimilarity (%)		22.89
Site 2	Gravel (stone or shell)	3.64	11.35	7.7	6.96	10.04	3.07
	Encrusting bryozoan	0.01	3.42	3.41	0.06	2.13	2.06
	Encrusting algae	0.43	4.57	4.1	0.81	3.24	2.43
	Faunal algal turf	42.51	29.16	-13.34	31.58	31.24	-0.33
	Bare rock	14.21	9.48	-4.72	13.32	9.73	-3.58
	Spirobranchus spp	26.62	28.19	1.5	24.50	27.98	3.48
		Disimilarity (%)		25.3	Disimilarity (%)		20.24
Site 3	Gravel (stone or shell)	4.08	15.92	11.83	3.45	15.36	11.90
	Spirobranchus spp	21.06	30.14	9.07	20.16	30.03	9.87
	Faunal algal turf	36.12	26.31	-9.8	37.33	25.6	-11.72
	Bare rock	17.30	17.22	-0.08	14.89	16.16	1.26
	Encrusting algae	0.79	0.73	-0.05	0.33	1.14	0.80
	Encrusting bryozoan	N/a	N/a	N/a	0.05	0.14	0.08
		Disimilarity (%)		22.4	Disimilarity (%)		21.96

Benthos that contributed more than 10% of the dissimilarity (consistently) between treatments, at all sites included: Gravel (stone or shell), *Spirobranchus* spp, Encrusting algae, Faunal and algal turf and Bare rock (Table A.2.). The largest increase in mean benthos percentage cover was gravel between B-I at site 3 (11.9%, Table A.2.) with the largest decrease found in short silty faunal algal turf between C_B-C_I at site 1 (13.9%, Table A.2.). Other benthos (*Spirobranchus* spp Encrusting algae, Feather hydroid, Encrusting bryozoan) differed by lower percentage cover between B-I and C_B-C_I (< 9.8%, Table A.2.). In general, benthos increased or decreased consistently across sites (Table A.2.), however, differences were not in line with those predicted by potting impacts (i.e. although Faunal and algal turf decreased, the increase of *Spirobranchus* spp and decrease Bare rock are not consistent).

A comparison of changes in mean percentage cover of benthos between B-I and C_B-C_I showed that these were also similar between treatments with only small scale differences in mean abundances on six occasions (highlighted grey in Table A.2.). This equates to differences in mean percentage cover between B-I and C_B-C_I ranging from 0.23 to -0.21%. There were minimal differences in direction between experimental and control sites consistent with the earlier broader analysis showing that B-I and C_B-C_I differ in similar ways.

Intensively fished Lhyp.Pk

Contributions of individual species to significant differences detected between treatments on a site level, showed similarities between sites for all treatments (SIMPER, Table A.3.).

Table A.3. Mean benthic cover contributing more than 10% to Bray-Curtis dissimilarity between treatments (SIMPER), in order of contribution. Only consistent indicator benthic cover was used. Difference in abundance between treatments are shown. Mean dissimilarity in samples between B-I and C_B-C_I are shown sites. Grey cells indicate difference in direction of mean abundance.

Site	Species	B	I		C_B	C_I	
		Av.Abund	Av.Abund	± Av.Abund	Av.Abund	Av.Abund	± Av.Abund
Site 1	Non-red algae (> 1 cm)	7.29	7.29	0	4	7.84	3.84
	Red algae (> 1 cm)	5.76	14.44	8.68	0.25	4.84	4.59
	Short silty faunal turf	10.89	16.81	5.92	18.49	25	6.51
	Bare rock	4.41	4.84	0.43	4.41	7.84	3.43
	Red algae turf (< 1 cm)	3.61	11.56	7.95	7.84	7.84	0
	Encrusting algae	2.56	1.96	-0.6	2.89	2.56	-0.33
		Dissimilarity (%)		60.1	Dissimilarity (%)		58.6
Site 2	Non-red algae (> 1 cm)	12.96	33.64	20.68	10.89	17.64	6.75
	Short silty faunal turf	17.64	3.61	-14.03	19.36	5.29	-14.07
	Red algae (> 1 cm)	5.76	12.96	7.2	5.76	13.69	7.93
	Red algae turf (< 1 cm)	7.84	4.84	-3	14.44	9.61	-4.83
	Encrusting algae	4.41	3.61	-0.8	6.25	6.25	0
		Dissimilarity (%)		57.2	Dissimilarity (%)		51.6
Site 3	Non-red algae (> 1 cm)	14.44	24.01	9.57	19.36	15.21	-4.15
	Red algae (> 1 cm)	7.84	20.25	12.41	12.96	24.01	11.05
	Red algae turf (< 1 cm)	12.25	5.76	-6.49	17.64	10.89	-6.75
	Bare rock	1.69	4	2.31	2.25	3.61	1.36
	Short silty faunal turf	14.44	6.76	-7.68	13.69	7.29	-6.4
		Dissimilarity (%)		50.5	Dissimilarity (%)		40.0

Benthos which were consistent indicators and contributed more than 10% of the dissimilarity for all sites were similar between treatments (Non-red algae (> 1cm), Red algae (> 1cm), Short silty faunal turf, Bare rock Red algae turf (< 1cm) and Encrusting algae)(Table A.3.). The largest increase in mean benthos percentage cover was non red algae (> 1cm) between B-I at site 2 (20.68%, Table A.3.) with the largest decrease found in short silty faunal algal turf between C_B-C_I at site 2 (14.07%,

Table A.3.). Non- red algae (> 1cm) and Red algae (> 1cm) differed consistently in mean benthos percentage cover between B-I and C_B-C_I across all sites with mean increases of 0 - 9.57% (Table A.3.). Other benthos differed less consistently between sites but similarly between treatments B-I and C_B-C_I in individual sites. Comparison of changes in mean percentage cover of benthos between B-I and C_B-C_I showed that these were also similar between treatments with only Non-red algae (> 1cm), site 3 differing in direction of change. There is no pattern in increase/decrease of taxon between treatments that are consistent throughout sites with those predicted by potting impacts. The increase in Red algae (> 1 cm) and Non- red algae (> 1cm) between B – I and C_B-C_I is consistent with either natural variation or habitat heterogeneity. These results are consistent with the earlier broader analysis showing that B-I and C_B-C_I differ in similar ways.

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