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Population characteristics and environmental interactions of the king scallop fishery in the English Channel

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Population Characteristics and Environmental Interactions of the King Scallop Fishery in the English Channel

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ABSTRACT

Scallops are currently the 3rd most valuable species in UK fisheries, with a first sale value of over £60 million per annum. Scallops are a non-quota species and landings have more than doubled in the last eight years. Forty percent of king scallop (*Pecten maximus*) landings into the UK originate from the English Channel. Despite the economic importance of this fishery there has never been a full stock assessment of scallops in English waters and there is a general paucity of scientific data for stocks in the English Channel. Existing management measures are not aligned with the biological structure of the stock, or based on robust science.

The aims of the study were to provide data to assess the sustainability of the king scallop fishery through identifying the maximum spatial extent and distribution of fishing effort in the English Channel and defining scallop population structure by the degree of larval connectivity between spatially segregated scallop beds. The environmental impacts of the dredge fishery were investigated through quantifying bycatch and the impact of the dredge fishery on the habitats and communities present across scallop fishing grounds.

The activity and spatial extent of the inshore and offshore scallop fleets indicated that although scallop fishing has occurred across a large proportion of the English Channel, fishing behaviour is sporadic and is concentrated in areas that are characterised by consistent scallop abundance. Economic and legislative drivers have altered historical fishing patterns in recent years. The patchiness of fishing behaviour, coupled with the nomadic nature of the fleet suggest that closure of areas that are infrequently fished could provide ecosystem benefits and potential benefits for the wider fishery while having minimal impact on fleet behaviour.

Despite the well documented environmental impacts of scallop dredges, it is important to understand the environmental context in which fishing occurs as well as the predicted recovery timescales for benthic communities. Within the context of the English Channel king scallop fishery, species diversity and benthic community composition are constrained primarily by natural physical disturbance. It was not possible to detect community level response to a gradient of scallop fishing intensity against a background of environmental forcing. This could be due to historical fishing activity (40+ years) that may have changed the

community to a stable altered state that continues to reflect the background environmental gradient across the English Channel.

Bycatch in the English Channel was low compared to other towed mobile fishing gears and compared to other scallop dredge fisheries in the UK. Bycatch composition varied over local and broad spatial scales. The fishery affected a limited number of bycatch species of ecological importance and low biomass of such species were present, indicating that the population impacts of the dredge fishery on these species are likely to be minimal. Discards of commercial species also varied significantly with location and were higher in the eastern English Channel.

Three reproductively distinct populations of scallops have been identified in the English Channel, indicating the largest appropriate management units. Large scale oceanographic currents maintain larval connectivity across much of the English side of the Channel; however complex hydrodynamic processes within Falmouth Bay suggest that larval dispersal is prevented at localised spatial scales in this location. The population in the Baie de Seine is reproductively isolated from the eastern English Channel, however larvae may disperse west, to the Baie St Brieuc and southern Cornwall, via residual currents.

Improving the management, sustainability and public perception of the English Channel king scallop dredge fishery is a priority for the UK scallop industry. This thesis addresses fundamental gaps in the scientific data to inform future management and the sustainable exploitation of the fishery.

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ACRONYMS

ANCOVA	Analysis of Covariance
ANOVA	Analysis of Variance
BSS	Bed Shear Stress
BTA	Biological Trait Analysis
CEFAS	Centre for Environment, Fisheries & Aquaculture Science
CI	Confidence Interval
CITES	Convention on International Trade in Endangered Species of Wild Fauna and Flora
DEFRA	Department for Environment, Food and Rural Affairs
EAF	Ecosystem Approach to Fisheries
EBFM	Ecosystem-based Fisheries Management
ETP	Endangered, Threatened or Protected
EU	European Union
FCA	Fuzzy Correspondence Analysis
FDR	False Discovery Rate
FI	Fishing Intensity
FIP	Fisheries Improvement Project
FTE	Full Time Employment
GES	Good Environmental Status
GPS	Global Positioning System
HWE	Hardy-Weinberg Equilibrium
ICES	International Council for the Exploration of the Sea
ICUN	International Union for Conservation of Nature
IFCA	Inshore Fisheries and Conservation Authority
JNCC	Joint Nature Conservation Committee
km	Kilometres
LD	Linkage Disequilibrium
LK	Local Knowledge

LOA	Length Overall
m	Metres
M	Molar concentration
MCS	Marine Conservation Society
MCZ	Marine Conservation Zone
MDS	Multi-Dimensional Scaling
MFPO	Manx Fish Producers Organisation
mins	Minutes
MLS	Minimum Landing Size
mm	Millimetres
MMO	Marine Management Organisation
MSC	Marine Stewardship Council
MSFD	Marine Strategy Framework Directive
MSY	Maximum Sustainable Yield
NEODAAS	Natural Environment Research Council Earth Observation Data Acquisition and Analysis Service
NGO	Non-governmental organisations
NM	Nautical Miles
PCA	Principle Component Analysis
PCoA	Principal Coordinate Analysis
PCR	Polymerase Chain Reaction
PDO	Protected Designation of Origin
PLD	Pelagic Larval Dispersal
S.D.	Standard Deviation
SAC	Special Areas of Conservation
SAGB	Shellfish Association of Great Britain
SFP	Sustainable Fisheries Partnership
SPUE	Sightings-Per-Unit-Effort
SSC	Sustainable Seafood Coalition

SWFPO	South-West Fish Producers Organisation
TAC	Total Allowable Catch
VMS	Vessel Monitoring System
WWF	World Wildlife Fund

CHAPTER 1: INTRODUCTION

Based on current understanding, at least a third of the world's fish populations are over-fished (Worm *et al*., 2009; FAO, 2014); a decade ago the estimate was below 20 % (FAO, 2002). This has serious implications for global food security as one-sixth of our expanding population is reliant on seafood as the primary source of protein (Tidwell & Allan, 2001). There are also financial implications to over-fishing (Srinivasan *et al*., 2010) with businesses and millions of livelihoods worldwide dependant on the industry; the annual rate of increase in employment in fisheries (3.2 %) is greater than that of population growth (FAO, 2014). Over-fishing not only depletes a stock, but can also lead to inefficient harvesting and can alter the habitat on which the target species and other species depend, with serious consequences for the diversity and stability of marine ecosystems (Hauge *et al*., 2009). Around 60 % of global fish stocks are fully exploited (the most desirable situation in order to optimise and maintain food resources) and 10 % could be exploited further (FAO, 2014). To reduce over-fishing, maximise sustainable harvests and protect fishing livelihoods there is a global drive towards promoting responsible and sustainable fisheries governance (Jaquet & Pauly, 2007). This has manifested in the strategic development of governmental policies that set out objectives for improving the management of fish stocks and the wider marine environment (MSFD, 2008/56/EC; Borja *et al*., 2010).

In the EU, the European Marine Strategy Framework Directive (MSFD) incorporates eleven descriptors of 'good environmental status', against which objectives of fisheries management can be set, with the aim of managing fisheries and conserving fish stocks. There are many factors that can influence the decline of fish stocks and marine ecosystems, such as habitat destruction, climate change, pollution, extreme weather and natural fluctuations in environmental conditions; however this thesis focuses on the anthropogenic impacts of fishing. In the MSFD, descriptor 6 relates to impacts on benthic ecosystems and requires that "*Sea-floor integrity is at a level that ensures that the structure and functions of the ecosystems are safeguarded and benthic ecosystems, in particular, are not adversely affected*". EU member states therefore have a statutory obligation to consider the impacts of any fishing activity which has an impact upon the sea-floor, as part of overall management objectives. Single-species based fisheries management is based on the concept of harvesting a species at the maximum sustainable yield (MSY). However, this approach does not account for the ecological interactions of a species population, such as trophic interactions and ecological processes (Hauge *et al*., 2009). The concept of sustainable management leads us toward an ecosystem approach to fisheries (EAF), which accounts not just for the species in question, but also for fishery-ecosystem interactions (Ecosystem Principles Advisory Panel 1999; Browman *et al*., 2004). This approach is also referred to as Ecosystem-based fisheries management (EBFM) (Link *et al*., 2011). Healthy and resilient marine ecosystems, including sustainable fish populations are promoted (Garcia *et al*., 2003) and such an approach is essential to achieve sustainable development of the full range of the ocean's ecosystem goods and services (Pauly *et al.*, 2002).

Eco-certification

'Sustainability' is a popular contemporary term used in many contexts across environmental, political, social marketing (Kotler & Zaltman, 1997) and media communications. In an era where digital communications enable the rapid dissemination of information and propaganda across the globe, fishers and the wider industry (e.g. wholesalers, retailers and restaurateurs) are increasingly aware of global public concern for the need to exploit our natural resources in a sustainable manner (Jaquet & Pauly, 2007). In fisheries management, consideration must be given to socio-economic, habitat and food resource sustainability such that resources can be "*maintained at a certain rate or level*". Unravelling the complexities of which species, from which fisheries or locations, and by which harvesting methods are the most sustainable for human consumption is no simple task. To enable consumers to understand the impacts of harvesting from our oceans and satisfy political agendas, eco-certification bodies that promote sustainable fisheries and aquaculture practices have become more pervasive in recent years and act to educate consumers and provide a baseline on which the global community can make decisions on sustainable purchasing (Jacquet & Pauly, 2007; Gutierrez *et al*., 2012). Such organisations include the Marine Stewardship Council (MSC), Friend of the Sea and the Aquaculture Stewardship Council. Many environmental non-governmental organisations (NGOs) also provide guidance and standards by which to evaluate the sustainability of a species or fishery (e.g. Marine Conservation Society (MCS); World Wildlife Fund (WWF); Greenpeace; Sea Web; Monterey Bay Aquarium). Each organisation has defined standards and objectives that aim to improve the sustainability of fisheries, geared towards promoting sustainable seafood.

Marine Stewardship Council

The Marine Stewardship Council (MSC) was created in 1996 by WWF and Unilever (who at the time were the world's largest frozen fish purchaser) and became an independent non-profit organisation in 1999. The number of MSC certified fisheries worldwide is currently 224, with a further 98 at various stages of the assessment process (MSC, 2014). This represents 9 % of global wild-capture tonnage (MSC, 2014) and certification signifies that these fisheries can be considered responsible and sustainable. The MSC certification process is underpinned by three key principles (see box 1), which in turn contain 31 indicators of sustainability. An independent assessment body evaluates the scientific evidence available for each indicator and scores a fishery based specific criteria. Where a fishery is data deficient a precautionary approach can be used, enabling a certain amount of flexibility in the assessment process. Fisheries that receive an intermediate score are incentivised to make improvements in order to achieve a higher score within specified timescales (Agnew *et al*., 2013). Fisheries scientists can provide evidence to inform P1 and P2, however P3 relates to management of the fishery, which must be designed and implemented by industry and policy makers. These three principles provide a structured framework for the sustainable management of a fishery.

Box 1: MSC Principles

PRINCIPLE 1 (P1): Fishing activity must occur at a sustainable level

PRINCIPLE 2 (P2): Fishing operations should maintain the productivity, structure and function of the ecosystem on which the fishery depends

PRINCIPLE 3 (P3): The fishery must meet all the local, national and international legislation and the management system must be enforceable and able to respond to changes in the fishery

There are numerous benefits to eco-certification schemes. Fisheries that are certified are more likely to have biomass levels at MSY (94 % of certified fisheries are maintained at or above MSY (MSC, 2014)) and stock biomass is likely to have increased at a faster rate than uncertified fisheries over the past decade (Gutierrez *et al*., 2012). Achievement of certification can improve sales security and aid marketing of a product. In the UK, the top six food retailers (http://www.retaileconomics.co.uk/) all stock MSC certified products (MSC, 2015). The most MSC certified seafood sold by a single retailer in 2013 was worth £149 million. UK supermarkets do not exclusively promote MSC certified seafood. Other partners through which sustainable sourcing is championed include the Marine Conservation Society, the Sustainable Fisheries Partnership (SFP), the WWF and the Sustainable Seafood Coalition (SSC), which was established by the organisation ClientEarth. However, these schemes attract some criticism. Aspects that challenge the efficacy of eco-certification schemes such as the MSC are a lack of tangible economic benefits to industry, the high cost and lengthy process involved, consumer apathy or lack of awareness of the certification body and issues over property rights within a fishery that impede the requirements of certification (Kaiser & Edwards-Jones, 2006; Gulbrandsen, 2009). The MSC has also received criticism for failing to accurately identify sustainable stocks (Sutton, 2003), achieve ecological improvements (Kaiser & Edwards-Jones, 2006; Jacquet & Pauly, 2007) and inconsistencies in the interpretation of assessment criteria (Ward, 2008). Despite criticisms of the MSC, the assessment process encourages the gathering and assimilation of data for a fishery, can enhance market access and sales value (Roheim *et al*., 2011), provide social (Perez-Ramırez, 2012) and environmental benefits (Martin *et al*., 2012), enhance transparency, dialogue and cooperation between stakeholders (Agnew *et al*., 2013) and improve the overall management of a fishery.

The king scallop fishery in the UK

King scallops, *Pecten maximus*, are broadcast spawners (Cragg, 2006) and mature in 2-4 years (Mason, 1958). They are also resistant to physical damage with the majority of damage following contact with dredge gear occurring as minor shell chips (Jenkins *et al*., 2001). These factors mean they are able to sustain high levels of exploitation if managed appropriately. The UK scallop fishing industry supports approximately 600 FTE fishing jobs and 750 FTE processing jobs (Scottish Fishermen's Federation, 2013). It is currently the 3rd most valuable fishery in the UK with a first sale value of £62.5 million in 2013, behind *Nephrops* (£85.9 million) and mackerel (£70.1 million) (MMO, 2014). King scallops are found in a range of habitats across northern Europe from shallow, sheltered inshore grounds to areas of seabed subject to high levels of natural disturbance, such as wave stress or tidally induced currents, although the UK is the largest producer in Europe (Seafish, 2013). In the UK, the main areas where scallop fishing occurs include western Scotland, the Irish Sea, Cardigan Bay and the English Channel.

There are no Total Allowable Catch (TAC) limits for scallops and recent exploitation is at the highest historical level; annual landings into the UK, from UK and foreign vessels increased by over 50 % between 2009 and 2012, from 35,100 to 54,200 tonnes. There was a slight decrease in 2013 to 49,400 tonnes which resulted from the restriction of fishing activity under EU law (see later section on ‘Current management’, p. 41). ICES area VII (which encompasses the English Channel and the Irish Sea, Figure 1.1) has a much greater economic importance to the fleet than area VI (west of Scotland). This is reflected in fleet behaviour in recent years. Between 2002 and 2013 the number of vessels targeting scallops in ICES area VI decreased by 21 % (resulting in an overall effort reduction of 46 %) while the number of scallop vessels operating in ICES area VII increased by 20 % (resulting in an effort increase of 39 %) (MMO, 2014). The increase in effort in area VII is due to both the displacement of activity from other sea areas and increased activity by vessels already fishing in that area (MMO, 2014). The importance of scallop fishing grounds in the English Channel (sub-areas VIId and VIIe) is highlighted by the fact that landings from the English Channel contributed between 37 and 56 % to total UK landings between 2009 and 2013 (Figure 1.2).

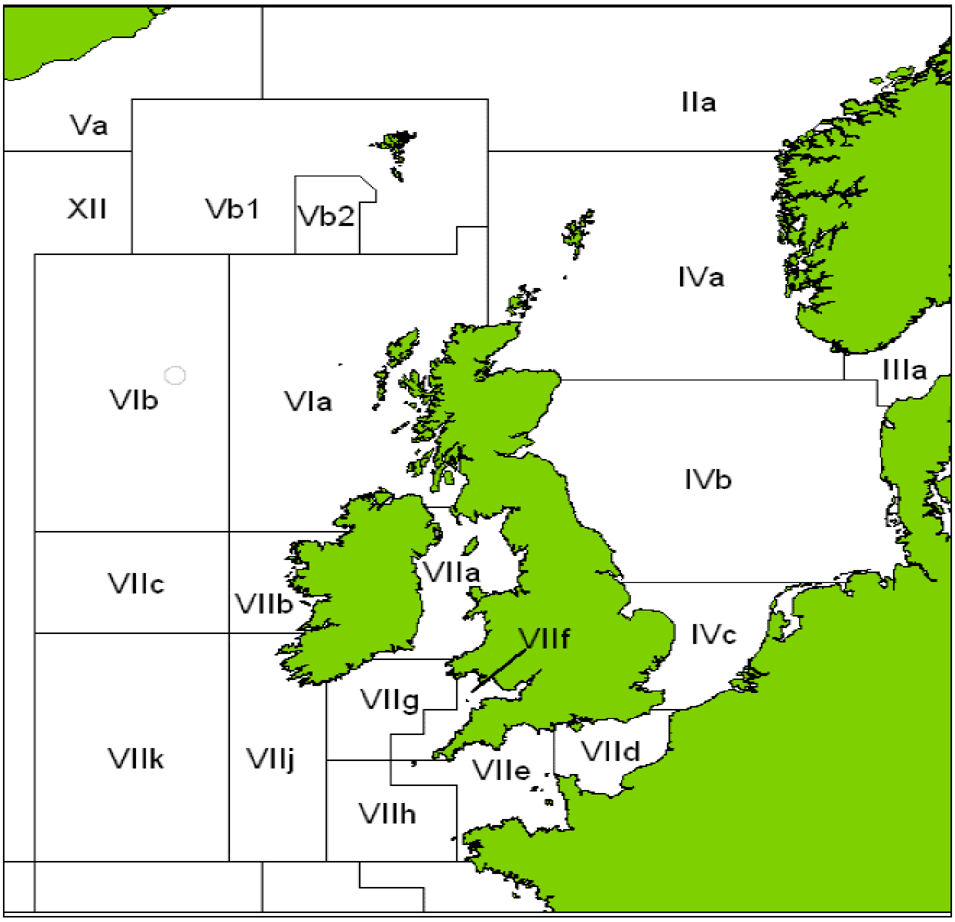

Figure 1.1: A map showing ICES statistical sub-rectangles covering UK waters. EU Western Waters regulations relate to all of area VII (sub-rectangles a-k).

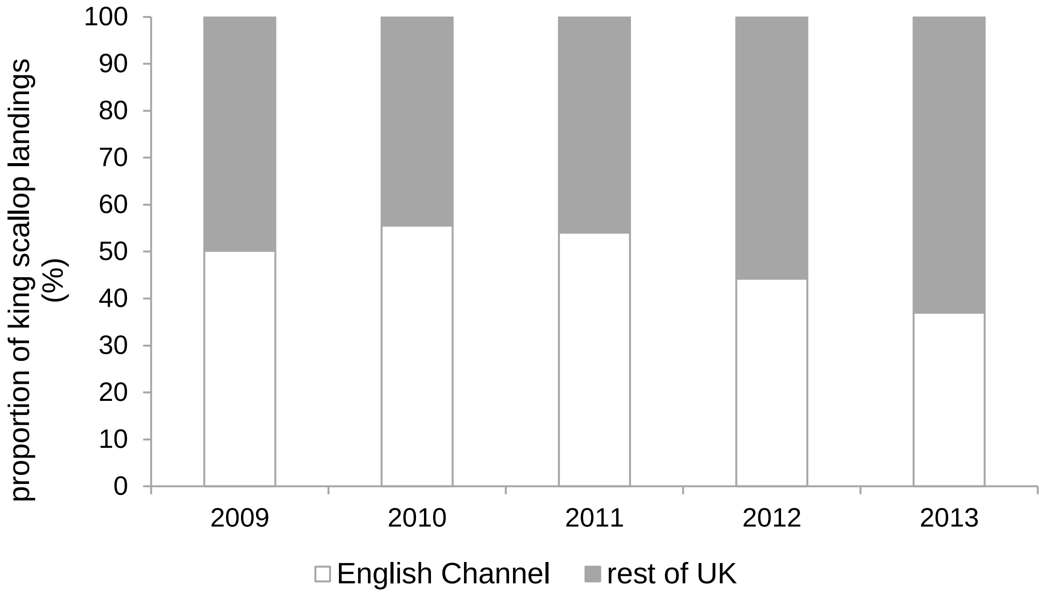

Figure 1.2: Proportion of scallop landings (tonnes) between 2009 and 2013 from the fishery in the English Channel (ICES sub-areas VIId and VIIe) (white bars) and the rest of the UK (grey bars). Data provided by the MMO.

Despite the economic and livelihood importance of the king scallop fishery, there is a general paucity of data regarding scallop stocks in the English Channel compared to other parts of the UK. Comprehensive assessment of scallop stocks has occurred in Scotland where a market sampling programme has been conducted by Marine Scotland since the 1970s. This has generated sufficient data to calculate catch-at-age data since 1982. Ireland completed stock surveys between 2001 and 2005, while annual stock assessment surveys conducted by Bangor University commenced in Welsh and Manx (Isle of Man) waters in 2013 and 2006, respectively. However, data for scallops in English waters are limited both spatially and temporally with sporadic sampling undertaken by The Centre for Environment, Fisheries & Aquaculture Science (CEFAS). A comprehensive stock assessment in English waters has never been completed, largely due to the fact that scallops are not subject to TAC and have not been prioritised for research funding. A recent attempt to gather data in the form of an industry self-sampling scheme was unsuccessful in obtaining enough data for a robust stock assessment as the limited number of samples obtained did not cover a large enough spatial or temporal scale (Bell *et al*., 2014).

Scallop fishing gear

Traditional Newhaven design scallop dredges used by the UK scallop fishing fleet comprise heavy steel tow bars with spring-loaded toothed dredges (that rake the seabed to a depth of 10 cm) attached to a steel ring collector bag that is dragged along the seabed behind the teeth (Figure 1.3). As the bag fills the weight increases and the impact on fauna it comes into contact with may increase (Jenkins *et al*., 2001). Although the king scallop dredge is reported as the most environmentally damaging type of fishing gear to benthic habitats including gravel, sand and mud (Collie *et al*. 2000, Kaiser *et al*. 2006), impacts vary widely according to the environmental context in which the fishing occurs. Due to the potential negative environmental impacts and the scale of the UK scallop dredge fishery, the industry has been portrayed negatively in the media in recent years (Guardian, 2013). In recognition of the need to design management that meets the biological requirements of the stock, ensures the safe-guarding of the resource and improves public perception of the fishery, the 'English Channel Scallop group' was formed in 2011 by scallop fishermen and processors, with the aim of developing a strategy for the sustainable and profitable management of the fishery. The group was formed in response to the National Scallop strategy, which was initiated by The Shellfish Association of Great Britain (SAGB). The strategy highlighted increased global competition and the need to adopt best practice throughout the industry. The group discussed putting the English Channel king scallop fishery forward for Marine Stewardship Council (MSC) certification. The MSC assessment process would provide a structured framework for identifying and addressing gaps in scientific knowledge for the scallop fishery with the aim of informing better management and enhancing sales security, value and public perception of the industry. The data requirements of MSC assessment form the basis of this thesis.

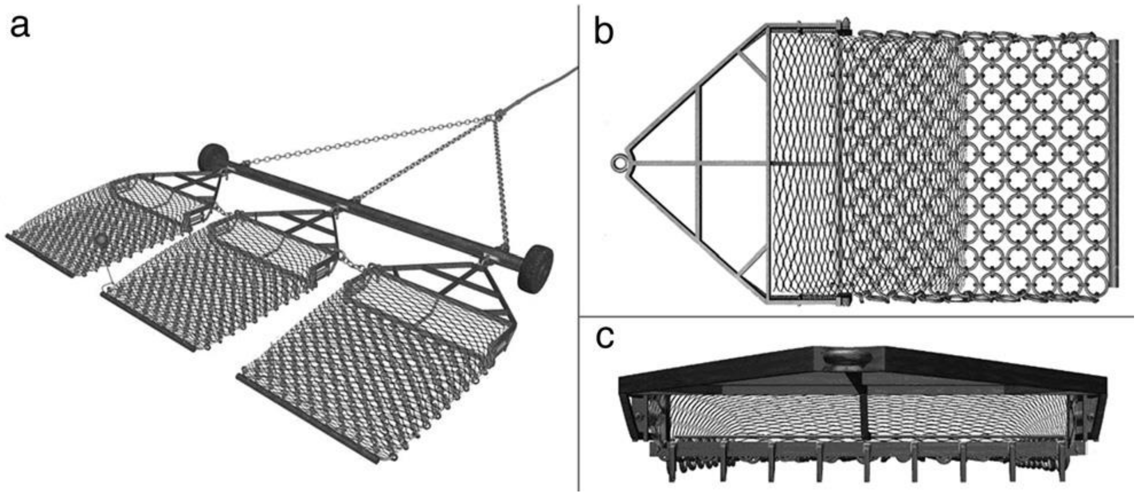

Figure 1.3: Newhaven king scallop dredge design: (a) aerial view of a gang of 3 dredges attached to the tow bar; (b) top elevation of the steel rings of the belly bag; (c) forward elevation of the tooth bar. Source: Boulcott & Howell (2011).

Environmental impacts of scallop dredging

Numerous studies have quantified dredge impacts in a wide range of benthic habitats with examples given in Table 1.1, although impacts vary widely depending on the environmental context. Areas of high natural disturbance are less likely to experience significant impacts from dredging than areas of low natural disturbance. Sciberras *et al*. (2013) found no differences in species diversity and scallop abundance between closed and seasonally fished areas of traditional scallop fishing grounds in Cardigan Bay, Wales. This could be attributed to fishing activity prior to the closure (areas had been closed for 13 to 23 months when sampled) or, that effects of fishing were masked by high levels of natural disturbance (such as currents and wave-induced bed-stress) between sampling events in the shallow, dynamic gravel/sand habitat. In contrast, Hall-Spencer and Moore (2000) found that scallop dredging had severe and long-lasting impacts on beds of slow-growing coralline algae that had previously not been dredged. Time taken for a habitat to recover post-fishing disturbance is often used to indicate the sensitivity of the seabed and biota. In the UK, recovery to the state of a pristine, undisturbed environment is unlikely to occur due to the extent of fishing activity; around 50 % of the English and Welsh seabed is fished every year (Foden *et al.,* 2010). Therefore, a measure of acceptable recovery can be used as an alternative target. For example, recovery to at least 90 % of previous benthic biomass, abundance or production (Hiddink *et al.,* 2006), or 80 % of previous total biomass or stable population levels (Gilkinson *et al.,* 2005) can be used as an alternative target.

Physical damage to non-target organisms can be caused by scallop dredges: on contact with the heavy steel tow bar or mesh collector bag; or from being towed inside the dredge along with the catch, rocks and stones. Selective mortality occurs between taxa (Hinz *et al*., 2011), with implications for short-term (Kaiser *et al.*, 1996) and long-term community changes (Veale *et al.,* 2001). Organisms with traits such as early maturity, broadcast spawning, rapid colonisation, high fecundity, extended larval stages, moderate to high mobility and robustness are more likely to flourish in areas of high disturbance (Bradshaw *et al*., 2002) and may therefore be less susceptible to mortality from fishing gear. Dredging can also have physiological impacts on benthic organisms. Juvenile scallops suffer a decrease in adenylic energetic charge (a measure of acute stress) for three days after exposure to dredging (defined as movement of the individual inside the collector bag). They also suffer from anaerobic respiration during emersion, leading to a decrease in the number of shell adductions, which is the behaviour used to escape predators (Jenkins & Brand, 2001). Consequences of sub-lethal impacts to scallops include slower righting and recessing speeds which can lead to reduced survival (Maguire *et al*., 2002) and increased vulnerability to predators (Jenkins *et al*., 2004).

Consideration of environmental impacts must also include the interaction of the fishery with other species. The EU Common Fisheries Policy (CFP) requires that all species should be managed within MSY. For each species it is necessary not only to consider mortality from the gear that specifically targets that species, but all bycatch from other fisheries or gears, so that MSY can be achieved in all fisheries simultaneously. By-catch assemblage varies depending on the substrate, and gear type and the degree of damage sustained is related to the morphological (Eleftheriou & Robertson, 1992) and behavioural (Bergman *et al.*, 1990) characteristics of the organism as well as the volume of stones in the dredge and the total catch volume (Veale *et al.,* 2001). Commercially valuable species caught in scallop fisheries around the UK include flatfish, skates, rays, monkfish, edible crabs, octopus and cuttlefish (Hill *et al.,* 1996, Enever *et al.,* 2007). Damage can be equal, or greater (e.g. for *Cancer pagurus*), in large, benthic invertebrates when not captured, than when occurring in dredge bycatch (Jenkins *et al.,* 2001). As dredge capture efficiency of such organisms is on average <25 % this implies most damage to bycatch occurs to individuals that remain on the seabed. Other organisms including the starfish, *Asterias rubens,* and the whelk, *Neptunea antiqua,* obtain greater damage inside the dredge. Damage to un-caught individuals is directly related to gear efficiency, with less efficient gears being more harmful (Gaspar *et al.,* 2003).

Table 1.1: Studies on the direct and indirect ecosystem impacts of dredging.

Type	Impact	Author(s)	Survey methods
Direct	Reduced benthic biomass and emergent epifauna	[a] Hiddink *et al.*, 2006b; [b] Lambert *et al.*, 2011	[a] Modelled using state pressure indicators, [b] Comparison of natural versus fishing pressure on distribution of biomass
Direct	Physical damage to organisms and seabed, reduction in physical diversity of seabed	[a] Eleftheriou & Robertson, 1992; [b] Minchin, 1992; [c] Black & Parry, 1994; [d] Collie *et al.*, 1997; [e] Bradshaw *et al.*, 2002	[a] Grab & core sampling, diver observations; [b] diver observations; [c] Turbidity sensor and video camera; [d] video/still photos; [e] seabed core samples
Indirect	Recovery times	Kaiser *et al.*, 2006	Meta-analysis of 101 fishing impact manipulations
Indirect	Reduced biodiversity	Cooper, 2007	Grab-sampling of benthic macro-invertebrate communities
Indirect	Benthic community shift, long-term impacts on epifaunal assemblages and habitats	[a] Kaiser & Spencer, 1996; [b] Collie *et al.*, 2000; [c] Veale *et al.*, 2000; [d] Bradshaw *et al.*, 2002	[a] Side-scan sonar and beam-trawling; [b] Meta-analysis; [c] abundance, biomass and production from log-book data; [d] Multi- and univariate analysis of benthic community data
Indirect	Change in functional composition or production of benthic communities	[a] Tillin *et al.*, 2006, [b] Strain *et al.*, 2012	[a] Functional community composition from beam-trawl samples; [b] Time series of dive surveys

High mortality of bycatch organisms can have negative implications for trophic relationships and community interactions (Bascompte *et al*., 2005). It is therefore important to quantify levels of bycatch for the fishery in order to manage it within the remit of EU legislation and prevent serious harm to the species and habitats with which scallops co-occur.

Stock connectivity

In order for management of a fishery to be effective it is important to understand the connectivity between two or more spatially segregated fish stocks. Wide geographic variability in scallop life-history characteristics such as time of spawning, recruitment, growth rate and size at maturity highlights the need for biologically distinct stocks to be identified so that management can be designed to match the scale of population processes. For example, it is important to assess at what scale such sub-populations should be managed in order to maintain sufficient spawning stock in each sub-population, based on the degree of larval input from other sources. This is particularly relevant for sedentary species such as scallops that exist as meta-populations, consisting of a number of smaller sub-populations separated by areas of unsuitable habitat that may be connected to varying degrees.

Scallop larvae spend up to 40 days in the water column before settling on a suitable substrate (Minchin, 1992) during which time they can disperse between locations. This process serves to connect or isolate spatially segregated beds of scallops, depending on the direction and strength of water currents and larval behaviour. Tian *et al.* (2009) used water current models to demonstrate that scallop larvae in the George's Bank area of the north-west Atlantic could potentially drift thousands of kilometres from spawning grounds. Important larval exchanges between scallop sub-populations were identified and it was found that none of the major beds were self-sustaining. This is important to understand in the context of management, as over-exploitation of one sub-population may reduce the resilience of connected sub-populations, if reliant on larval input to maintain population levels. Other forces such as cold-water upwellings, wind or other physical oceanographic processes can influence the broad-scale dispersal of larvae which may travel large distances, or be retained within localised patches. A scallop bed in one location could act as a source of larvae to another area, or could be a larval sink (an area in which the majority of larvae are retained and contribute very few larvae to the meta-population (Liu *et al*., 2011). Larval sources can be identified as a priority for conservation while larval sinks may be regarded as fully exploitable.

Climate change has warmed UK waters by around 1.0°C in the last 40 years (Perry *et al*., 2005). In the context of using closed areas to conserve a species stock, understanding connectivity between areas will be critical if climate change causes species ranges to shift out of areas established for their protection (Cheung *et al.,* 2009). Some fish stocks are able to respond to changes in climate via distributional modifications (Perry *et al*., 2005). Species that have a reduced capacity to move may experience increased mortality due to changes in local environmental conditions. Particle tracking models have been used to indicate the movement of scallop larvae on tidal currents around the Isle of Man, with the findings used to inform stock enhancement programmes (Neill & Kaiser, 2008). The model incorporated information on hydrodynamic conditions including tidal flows, currents, gyres, meteorological forcing (e.g. wind) and larval biology. The authors of the study suggested that future models could incorporate further scenarios of wind and temperature variations forecast under climate change predictions. The importance of connectivity is demonstrated through being one of the seven design principles specified for the creation of the UK network of Marine Conservation Zones (MCZ's) under the Marine and Coastal Access Act 2009 (Hill *et al.* 2010) and is a crucial factor to be considered when designing marine reserves to ensure that population growth can spill-over from the protected to exploited areas. Increased adult biomass in protected scallop beds generates larvae for replenishment of connected populations (Howarth *et al*., 2011).

Population genetics can be used as an indirect means of tracing migration between marine populations. Patterns of genetic variation within populations are used to predict the degree to which marine populations deviate from Hardy-Weinberg equilibrium (Hardy, 1908; Weinberg, 1908). A population is said to be in Hardy-Weinberg equilibrium if there is no mutation, migration or natural selection occurring, population growth is unrestricted and any individual can mate with any other individual (Beebee & Rowe, 2004). This situation does not occur in reality (Schaal, 1975) but is used as a benchmark to describe to what degree a population deviates from equilibrium. Exchange of genes between populations results in homogenisation of allele frequencies, therefore differences in the frequency of alleles between two populations can be used to estimate the genetic connectivity between them. Migration between populations prevents genetic differentiation and decreases the effect of genetic drift by increasing the effective population size (the number of successful breeding individuals in a population). In fisheries management, effective population size is a key factor to consider when predicting future recruitment, setting sustainable catch levels and maintaining the genetic fitness of a population. Genetic variation is highly sensitive to migration therefore genetic markers can provide strong evidence of a closed population (Hellberg *et al.,* 2002) and are used to reveal genetic differences or similarities between sub-populations. If connectivity among populations is low and gene flow is restricted, genetic drift may lead to divergence in allele frequencies within a population. This divergence can be detected using molecular markers such as microsatellites and used to infer population connectivity. Microsatellites (tandem repeats of 2-10 base pair nucleotide motifs) are effective molecular markers for studies of population ecology due to their fast mutation rates (Zhang & Hewitt, 2003). The major limitation of this approach is that homogenisation of allele frequencies can occur with very low levels of migration: slightly more than one per generation (Crow and Kimura, 1970). Whilst this level of connectivity is significant over evolutionary timescales to connect and homogenise subpopulations, it is negligible for fisheries management that operates over shorter timescales. Genetically distinct stocks therefore represent the largest scale at which populations should be managed.

Current management of the English Channel scallop fishery

The current management regime for scallops in English waters is not based on biological knowledge of the stocks or a comprehensive assessment of habitat impacts and is therefore at risk of being inadequate to meet political, environmental and socio-economic objectives. A scallop permit (also known as scallop entitlement) is required to fish for scallops in the UK. In 2014, there were 103 active scallop licences for >15 m LOA (Length Overall) vessels, and 66 for <15 m LOA vessels in the English Channel. In the whole of the UK, there were 384 scallop permits, but 156 were considered latent (S. Pilgrim-Morrison, MMO, pers. comm.). This indicates the potential for a significant increase in effort in the fishery, should the owners of the licences decide to use them. The >15 m LOA scallop fleet operating in ICES areas VI and VII are managed on the basis of effort restriction under the Western Waters Effort Regime (Council Regulation (EC) No 1415/2004). This legislation prevents total effort in the fishery exceeding a baseline level. Fishing effort is defined as fishing capacity (gross tonnage or engine power (kW)) x activity (Communication from the Commission to the Council and the European Parliament of 5 February 2007 on improving fishing capacity and effort indicators under the common fisheries policy – COM(2007) 39 final). Under the Western Waters regime, each scallop vessel is allowed to fish for a set number of days each year in area VII. These days are allocated on a quarterly basis, issued as a licence variation by the Marine Management Organisation (MMO) with each vessel receiving the same number of days regardless of vessel size. Effort days that are not used during the quarter in which they are allocated become available for redistribution among the entire fleet in the next quarter but cannot be ‘rolled-over’ to the following year. The UK annual effort allocation in area VII is 3,315,619 kW days (kW relates to vessel engine size, with larger vessels accounting for a greater proportion of the effort uptake). In 2010 and 2011 the fleet exceeded this limit (by 26 and 21 % respectively, K. Williamson, MMO, pers. comm.), which resulted in a month long closure of the fishery in October 2011 to avoid further overuse and related penalties. Following this, effort uptake has been closely monitored and enforced by the devolved Administrations. In 2013, the uptake of effort in area VI was 54 % (out of an allowance of 1,974,425 kW days) while in area VII, effort uptake was 92 % (out of an allowance of 4,267,619 kW days (including additional effort sourced from swaps, see below)) (MMO 2014). Had enforcement not been in place, total effort in area VII is likely to have been higher, as seen in 2011 and 2012. In recent years the UK devolved Administrations have obtained additional effort days for the fleet by exchanging fish quota with other nations (including Ireland and Holland). The French scallop fleet have more than double the annual Western Waters effort allocation of the UK, but only utilise around half of it each year. Although there are a greater number of French scallop vessels, there are no vessels over 18 m LOA and the French scallop fishery is closed for half of the year. Unused French effort days have been obtained by the UK fleet by agreeing to seasonal closure of fishing grounds in the eastern English Channel, aligning with French fishery restrictions. If all of the French effort days were utilised this would significantly increase pressure on the scallop stocks in the English Channel, with unknown consequences.

Each devolved UK Administration has the power to implement additional fisheries management and the measures used vary between regions (see Appendix 1.1 for a full list of management measures across the UK, Channel Islands and the Isle of Man). Technical gear specifications are in place across all devolved administrations and applicable across the EEZ (Exclusive Economic Zone), with further technical and spatial measures enforced in inshore areas (<6 NM from the coast). Scotland restricts the total number of dredges that can be towed in both inshore and offshore (>12 NM) waters. In England, ten regional IFCAs (Inshore Fisheries and Conservation Authorities) have the power to impose byelaws to restrict fishing activity within the 6 NM limit. Scotland, Wales and the Isle of Man all have more stringent restrictions than England in the 6-12 NM zone (such as limits on engine size and the number of dredges per side) which has resulted in some displacement of larger, nomadic scallop boats from Scottish to English waters. All management measures serve to restrict overall effort, or reduce the catch of undersized scallops, in various ways.

Despite current management measures, due to the increasing number of vessels entering the scallop fishery, latent capacity (the number of scallop licences that are not currently in use), historically high landings, no stock assessment and no TAC for the king scallop fishery, there is a risk of over-exploitation occurring. Scallops are a sedentary species and fishing patterns are irregular; fishers continually shift effort between spatially distinct grounds to maximise catch rates. Therefore, a near-constant or increasing catch rate can mask stock depletion. EU effort limitations originated from a political regime rather than being based on the biology of the stock. Therefore, despite the benefits gained from limiting total fishing effort of >15 m LOA vessels in the English Channel, there is no available data to ascertain whether existing levels of fishing are above or below that which would achieve MSY, which is a requirement under the CFP. The UK Western Waters effort allowance is not sufficient to enable the free movement of the fleet at current capacity and if more vessels enter the fishery this allowance will be further spread between vessels. This causes displacement of effort from area VII (part of the Western Waters management area) back to area VI, but it is unknown whether the reproductive capability of stocks in area VI will sustain the additional exploitation. Vessels have to travel between areas VI and VII more frequently as a result of displacement, leading to increased fuel costs and vessel owners are reporting financial losses (personal observation). A major challenge for fisheries managers is to achieve a number of potentially conflicting objectives: adhering to national and EU legislation; meeting conservation objectives; and meeting the diverse (and often differing) requirements of small, medium and large vessels in the fleet. The economic and behavioural impacts (Lee *et al*., 2010) to the fishing fleet caused by management measures must be carefully considered. Robust, up to date knowledge regarding the fishery, along with industry engagement is necessary for this to occur.

Examples of successful scallop fishery management

The sea scallop (*Placopecten magellanicus*) fishery on Georges Bank (north-western Atlantic) is the most valuable fishery in the US (worth around $600 million p.a.) and was MSC certified in December 2013. The fishery uses New Bedford dredge gear, involves 348 vessels and is the largest wild-caught scallop fishery in the world. The sea scallop has similar life-history and reproductive characteristics to the king scallop. In the 1990's the fishery was being exploited at unsustainable levels but was rescued from the brink of collapse via closure of three large areas of ground to fishing in 1994, totalling 17,000 km^2 (Murwaski *et al.*, 2000). Within 4 years of closure, scallop biomass was 14 times greater in closed, compared to open, areas. Other management measures, including restrictions on crew size, the number of days spent at sea and daily landings, helped to reduced pressure on the stocks (Valderrama & Anderson, 2006). Management is reactive and is revised each year following an annual stock survey. Grounds are opened on a rotational basis and a proportion of the annual profits go towards further research into the stocks and habitats. Evidence from the fishery demonstrates that greater benefits (both ecological and economic) are obtained from longer closures (6-8 years). The Georges Bank scallop fishery is a success story and demonstrates what can be achieved with appropriate, effective management and engagement with the industry. Although the background, scale and context of scallop fisheries around the world varies, recurring themes throughout examples of effective management include a high level of industry involvement in developing management, effort limitations and spatial or temporal closures.

In Shetland, effective management of the scallop fishery is attributed to a high level of industry and community involvement, consensus and support. The Shetland Shellfish Management Organisation is a community based group, formed to implement fisheries management initiatives. It represents not only industry but also local government, community councils, Scottish Natural Heritage and a local fisheries college. Local involvement is a key aspect of the management as 20 % of the population are employed in the fishing industry (Goodlad, 2005). A regulating order for the scallop fishery, proposed and implemented by the industry was introduced in 2000 to enforce limited, non-transferable entry permits, a curfew and limit on total dredges, closed areas and seasonal closures. The fishery achieved MSC certification in 2005. Since then catch levels have increased consistently and spawning stock biomass has remained stable at above average levels (Barreto & Bailey, 2014) indicating that current harvest levels are sustainable.

The Bay of Saint-Brieuc scallop fishery, fished exclusively by French fishermen (255 vessels), contributes between 20-50 % of total French scallop landings each year (Lesueur *et al*., 2008). A cooperative was formed in the 1970's and governance has been based on co-management between the industry and fisheries administration, combined with scientific support. Fishermen are represented at the regional level, by the Regional Committee of

Maritime Fisheries and Marine Fish Farms (Comité Régional des Pêches Maritimes et des Elevages Marins - CRPMEM). Although regulatory decision-making falls to the national, or regional Administrations, fishermen are fully involved in the process and suggest opening and closing dates for the fishing season and quotas. Entry to the fishery is controlled by licences and strictly limited. Vessels may only operate two dredges at a time, for 45 minutes per day and the fishery is closed between April and October. Yet, the fishery remains productive and profitable with landings worth 10.9 million Euros in 2012-2013 (Anon, undated).

Contrary to the examples above, informed management does not always result in positive outcomes. The Isle of Man is a unique study system in that 100 % coverage of fishing effort data are available in addition to long-term community and habitat data. As a self-governing British Crown dependency that is not a member of the EU, the Isle of Man government has full discretion over management of fisheries within the 3 NM zone. In 2009 a large area of Ramsey Bay in the northeast of the island was designated the nation's first Marine Nature Reserve, for the purpose of biodiversity conservation and fisheries management. The area was closed to fishing, in agreement with local fishermen, providing a valuable breeding stock of scallops. The implementation of the reserve was facilitated by extensive stakeholder consultation, involving user groups and the wider public. This generated local support and enabled local knowledge to inform decisions about management. Subsequent scientific assessment of the reserve showed that scallop density had increased and scallops were larger and faster-growing compared to other areas around the island (Dignan *et al*., 2014). Other benefits to populations in protected areas include higher reproductive biomass and potential, increased survival of juveniles, and larval export to nearby fishing grounds (Beukers-Stewart *et al*., 2005; Howarth *et al*., 2011). In 2013, 20 % of the closed area was reopened and a healthy TAC of 23t was harvested by just 3 vessels, with the profits shared as dividends among all members of the Manx Fish Producers Organisation (MFPO) (Dignan *et al*., 2014). This targeted method of exploitation resulted in a 9-fold increase in energy efficiency compared to normal fishing activity in open areas, when considering meat yield versus the energy expended to catch the scallops (Dignan *et al*., 2014).

In contrast, developments in the Manx queen scallop fishery have resulted in a different outcome. Queen scallops have been fished around the Isle of Man since the 1950's. In 2011, the queen scallop trawl fishery successfully achieved MSC certification and the following year the fishery was awarded European Union Protected Designation of Origin (PDO) status.

Between 1983 and 2009 landings of queen scallops in to the UK and the Isle of Man from Manx waters averaged 3875t per year. However by 2010 landings had increased to 10,717t compared to the recommended TAC of 5,000 t (Dignan *et al*., 2014b). This led to MSC certification being suspended in June 2014 as stock levels were found to be beyond critical thresholds of sustainability (Dignan *et al*., 2014b). This demonstrates that even with the best data available to inform management, social and political pressures (such as the desire of the government to maintain income from the fishery) can over-ride scientific advice and impede effective management (Daw & Gray, 2005).

Rationale for the study

The UK scallop Association initiated an MSC pre-assessment of the English Channel king scallop fishery in 2010. This highlighted key areas where data required for the full assessment process was lacking. Following this, the present study was initiated, funded by members of the Scallop Association with the research aims driven directly by the data requirements. Hence, throughout the thesis references are made as to which MSC criteria the research is addressing.

A full stock assessment required for P1 (sustainable stocks) was considered beyond the scope of the present study. However, data required to fulfil the other criteria under P1 (stock structure) and P2 (regarding environmental impacts) were included in the study. Under P1, in order to assess the sustainability of the stock it is necessary to ascertain the spatial extent of the fishery and the biological structure of the population. Under P2, there were two aims. Firstly to quantify the impact of the fishery on the species and habitats present within the fishery, while accounting for natural disturbance. Secondly, bycatch of the fishery needed to be quantified, to assess whether the fishery has significant impacts on other species.

Specific objectives for each chapter were to:

1) Define the spatial extent and relative intensity of the inshore and offshore scallop dredge fisheries
2) Assess how the spatial and temporal scale of aggregation of vessel monitoring system (VMS) data influences estimates of fishing intensity, as such data are used to assess the degree of impact the fishery has on the seabed environment

3) Assess the impacts of the dredge fishery on the habitats and communities present across scallop fishing grounds in the English Channel, accounting for variation in environmental conditions
4) Identify and quantify bycatch species and compare to other dredge fisheries around the UK
5) Quantify the degree of larval connectivity between scallop beds in the English Channel to inform appropriate management units.

Management measures should ensure the long-term viability of the fishery, be effective and enforceable and have the capacity to adapt to changes in the fishery, as specified under P3. Although the science alone cannot address the latter points, once comprehensive data about the fishery has been collected, the industry will be in a position to work towards these goals and inform sustainable management to meet the requirements of P3. It is expected that further benefits from enhanced management will include a more profitable, sustainable fishery, improved public perception, increased sales price and markets and reduced environmental impacts.

CHAPTER 2: MAPPING THE SPATIAL FOOTPRINT OF THE INSHORE AND OFFSHORE SCALLOP DREDGE FISHERIES

MSC data requirements addressed:

P1	Data requirements
Stock status	In order to assess the stock status, the spatial distribution of the fishery needs to be determined.
P2	**Data requirements**
Habitats	Management strategies should be appropriate to the scale and intensity of the fishery. There should be adequate knowledge of the impacts of the fishery on the ecosystem; this involves knowledge of the total area impacted by the fishery.

Abstract

At the forefront of fisheries management plans is the requirement for knowledge of the area impacted by the fishery. Fishers' local knowledge (LK) data have been shown to be a useful alternative to quantitative sources of fishing effort, such as that derived from Vessel Monitoring System (VMS) data. Skippers of both inshore (<15 m LOA (Length overall)) and offshore (15-40 m LOA) scallop vessels were asked to indicate on a map where they fished to determine spatial patterns of scallop fishing activity in the English Channel. During semi-structured interviews information was also gathered regarding changes in fishing habits over the past 10 years and opinion on current and future management of the fishery. For offshore vessels, fishing effort from LK significantly correlated with VMS data and the correlation increased with increasing grid cell resolution. As grid cell size increased, so did the estimation of the total area impacted by the fishery. LK data enabled a map of relative inshore (<6 NM) scallop fishing activity to be produced, covering the full extent of the scallop fishery in the English Channel for the first time. The LK data provided a reasonable representation of the spatial extent of the inshore fishing activity, whereas representation of the offshore fishery was more conservative. The LK data also highlighted frequently fished areas that are of particular importance to the inshore fleet. This information gives insight into the potential socio-economic impacts of ground closures that could occur through the future designation of Marine Conservation Zones (MCZs). Factors that have influenced changes in fishing behaviour over recent years were identified and differences in opinion regarding management based on individual requirements (largely characterised by an inshore vs. offshore fleet divide) were ascertained, with fishers' expressing varying levels of support for current management measures. Such knowledge can inform the development of future marine planning in relation to both sustainable fishing and conservation objectives.

2.1 Introduction

Mapping temporal and spatial patterns of fishing activity is an integral part of marine spatial planning. It enables the extent of the environmental impact to be determined (Jennings *et al.,* 2012) as well as the potential economic impacts of proposed management measures (Pederson *et al.,* 2009). Where empirical evidence for a fishery is scarce or absent, information can be gathered directly from fishers to inform marine and fisheries management (Bergmann *et al.,* 2004; Drew 2005; Hall & Close 2007; Shepperson *et al.*, 2014). Using the Local Knowledge (LK) of resource users or indigenous communities to complement (or in some cases, as a substitute for) scientific knowledge is not a novel concept (Johannes 1978; Freeman 1992). Scientists can utilise LK from fishers to ascertain where fishing occurs; the seasonality of fishing; identify locations of potential gear conflict; place economic or perceived value on fishing grounds; aid the design and planning of Marine Protected Areas (MPAs) and attain estimates of fishing intensity (Close & Hall, 2006; Lieberknecht *et al.,* 2011; Yates & Schoeman, 2013; Leite *et al.,* 2013). Fishers can have a greater ability to detect short-term trends in fisheries than available scientific data and are able to provide information on year-to-year variability of fish stocks (Rochet *et al.,* 2008). In contrast, scientific surveys are often limited in temporal and spatial scales, whereas experienced fishers have years of knowledge and interact with the fishery environment on a daily basis.

The process of gathering LK can provide a number of benefits to scientists. Communication with fishermen enables scientists to gain a better appreciation of the interaction of the fishery with the species and habitats in question. The process of engagement with stakeholders can break down barriers to communication and facilitate the development of improved relationships between fishers and scientists (Mackinson *et al*., 2010). However, spatial data gathered from fishers has limitations. For example, it is not as accurate as that obtained from vessel monitoring systems (e.g. Shepperson *et al.,* 2014), which can reveal the precise location of fishing activities. The collection and possession of LK by scientists poses the issue of disclosing confidential and commercially sensitive information (Rudd, 2001; Maurstad, 2002). However, in some cases fisher knowledge represents the best, or only, available data. In the UK, the value of LK to inform the spatial management of inshore fisheries has been recognised and comparable projects to ascertain spatial patterns of fishing activity and the economic value of fishing grounds have been undertaken in Scotland (Kafas *et al.,* 2014), Ireland (Yates & Shoeman 2013) and north Wales ('Fish Map Môn' project). In particular, the ScotMap project data has been useful in marine spatial planning in areas where multiple uses such as renewable energy and conservation objectives must be considered (Kafas *et al.,* 2014). Despite the importance of such information, it is recognised that the accuracy of the data is limited at fine spatial scales (Kafas *et al.*, 2014).

Mapping fishing activity

For areas where electronic tracking of fishing vessels occurs, delineating the boundaries of a fishery is relatively straight-forward (Lee *et al*., 2010). In the UK, all commercial vessels >12 m LOA (length overall) must carry a working Vessel Monitoring System (VMS) when at sea. In the European Union (EU), a VMS record is transmitted approximately every 2 hours when a vessel is at sea, enabling vessel activity to be monitored retrospectively. Although these data are gathered primarily for enforcement purposes, the data can used to analyse spatial fishing patterns and estimate fishing effort (e.g. Mills *et al.*, 2007; Hintzen *et al.*, 2010; Lee *et al.*, 2010; Gerritsen *et al.*, 2013). When joined with landings data the economic importance of different fishing grounds can also be determined (Gerritsen & Lordan, 2012). In the English Channel, VMS data have been used to estimate fishing intensity from towed mobile gears in specific locations but such studies have been temporally restricted (Vanstaen, 2010; Campbell *et al.*, 2014). In the EU, VMS has been compulsory for vessels >15 m LOA since 2005 and for vessels >12 m LOA since 2012. However, over 90 % of registered fishing vessels in England and Wales are <15 m LOA (MMO, 2012), which means that there is a lack of spatial data for this sector of the fleet. In the absence of VMS data, other methods are employed to describe the location and intensity of inshore fishing activity such as combining environmental data with expert information on the location of fishing to estimate the area of sea impacted (Dunn *et al.*, 2010). Breen *et al.* (2014) used records of observed fishing activity from fisheries enforcement data to calculate sightings-per-unit-effort (SPUE) as a measure of relative fishing intensity. In the latter study, although correlation with VMS data (where this was available) was high, limitations included a low density of sightings data and positional accuracy in some areas, the sporadic nature of data collection and gaps in the data set for areas that are never visited by fisheries enforcement vessels.

LK data can provide a reasonable estimation of the spatial extent of fishing; verified by comparing maps of fishing effort derived from LK data to 100 % VMS coverage for a fleet (Shepperson *et al.*, 2014). Aggregation of data at a finer scale provides a more accurate representation of the spatial extent of the fishery. However, when using LK to estimate fishing intensity; accuracy increases with the proportion of the fleet sampled and aggregation of the data at a coarser scale (Shepperson *et al.*, 2014). To date, there has been no attempt to collect LK data to map the activity of the English Channel king scallop fishery. Due to commitments under the EU Habitats Directive ((92/43/EEC) and the Marine Strategy Framework Directive (MSFD, 2008/56/EC) to develop networks of Marine Protected Areas (MPAs), and the number of livelihoods reliant on inshore fisheries in the UK (Breen *et al.,* 2014), establishing the location and intensity of inshore fishing activity is essential for marine spatial planning. The aim of the present study was to gather LK from a representative range of scallop fishermen that have fished in the English Channel over the last decade to address the following objectives:

1. Map the spatial extent and relative intensity of the inshore (<15 m LOA vessels) and offshore (>15 m LOA vessels) scallop fisheries in the English Channel.
2. Assess the validity of using fishers' LK to estimate the extent and relative intensity of scallop dredging by comparing maps of LK with VMS data (for vessels >15 m LOA).
3. Obtain information to describe the characteristics of the fleet, changes in fleet behaviour over a recent (10 year) time period, and industry views on current management measures.

2.2 Methods

In the UK vessels >10 m LOA must hold a scallop licence in order to fish for scallops but vessels <10 m LOA are permitted to land scallops without such a licence. Some scallop licences are inactive, or used on a seasonal or ad-hoc basis depending on weather conditions, scallop abundance or the availability of quota for other species. Data for all UK vessels that land scallops are held by the Marine Management Organisation (MMO). For some vessels this will reflect ad-hoc landings (as a result of bycatch when using other gear such as beam trawls, gill nets or otter trawls). Based on economic decisions, some vessels switch between scallop dredge gear and beam-trawling (or other fishing gears), whereas other vessels fish for scallops year-round. Data from the MMO were used to provide an accurate estimate of the number of full-time scallop vessels active in the English Channel (ICES sub-areas VIId and VIIe) fishery over the last decade, by calculating a mean for the total number of vessels that have targeted scallops for trips where dredge gear was used, or scallops were the main retained species (Table 2.1). The mean number of vessels that exploited the scallop fishery in ICES sub-areas VIId and VIIe, between 2006 and 2014, was 151.

Table 2.1: Total number of vessels targeting king scallops ±S.E. (includes data from trips by vessels where scallops were the main retained species, or scallop dredges were used) caught in ICES sub-areas VIId and VIIe, split by vessel length.

Year	<10 m vessels	10 - 15 m vessels	>15 m vessels	Total vessels
2006	57	39	37	133
2007	73	38	31	142
2008	89	38	23	150
2009	90	35	28	153
2010	63	39	35	137
2011	88	44	41	173
2012	88	43	36	167
2013	93	49	39	181
2014	45	35	42	122
mean 2006-2014	**76 (±5.8)**	**40 (±1.5)**	**35 (±2.1)**	**151 (±6.5)**

A questionnaire (Appendix 2.1) was administered to scallop fishermen, contacted via the UK Scallop Association, the South-West Fish Producers Organisation (SWFPO) and referrals provided by fishermen. All of the participants were full-time skippers of vessels that target scallops for all or part of the year. All interviews were conducted in person, by the author, either on board vessels or in a suitable meeting place such as a café, office or the skipper's home. The first section of the questionnaire involved a series of 39 quantitative and qualitative questions regarding vessel and gear characteristics, fishing habits, economics and opinion regarding the management of the fishery. Questions were either: closed; required an answer based on a Likert scale; or were structured in an open format to encourage greater sharing of information.

During the second section of the questionnaire, fishermen were asked to identify all locations in the English Channel that they had actively fished for scallops with their current vessel, over the previous 10 year period. This was done by drawing polygons directly onto a geo-referenced admiralty chart of the English Channel in ArcMap v.9.1 using software developed for the 'FisherMap' project (des Clers, 2008). Some skippers had worked on the same vessel for the full 10 year period while others had recently changed vessels, or were more recently qualified as skippers. Data for fishing locations was only recorded for the time period the interviewee had been the skipper of the vessel. This was to avoid any duplication of data if >1 fisher had skippered a particular vessel (which occurred a number of times). For each polygon drawn, participants were asked to indicate which months in the year they normally visited the area to fish, and on average how many days per month fishing activity occurred. They were also asked to indicate how many years in the last 10 (or as long as they had been skipper of the vessel, if <10 years) they had fished within the specified polygon. Interviews were conducted with 19 skippers of vessels >15 m LOA (length overall) and 29 skippers of vessels <15 m LOA between summer 2012 and autumn 2013. Based on data provided by the MMO for scallop vessel activity in recent years (Table 2.1) this constitutes approximately 54 % and 25 % respectively of the mean number of full and part-time scallop vessels operating in ICES sub-areas VIId and VIIe over the past decade. Full-time scallop vessels are defined as those that use only scallop gear. Part-time scallop vessels are those that target scallops during certain times of the year but target other species with different gear (e.g. beam-trawl) the remainder of the year. There were more frequent opportunities to interview skippers of vessels <15 m LOA, as vessels of this size tend to return to port each day and are limited by weather conditions. There were fewer opportunities to interview skippers of larger vessels as they spend up to a week at sea and after landing the catch often leave port immediately for the next fishing trip. There are 19 landing ports along the south coast of England (Figure 2.1). Interviews were conducted with skippers of vessels either registered at, or landing into 13 of these ports. This included a number of Scottish and, to a lesser extent, Welsh owned vessels.

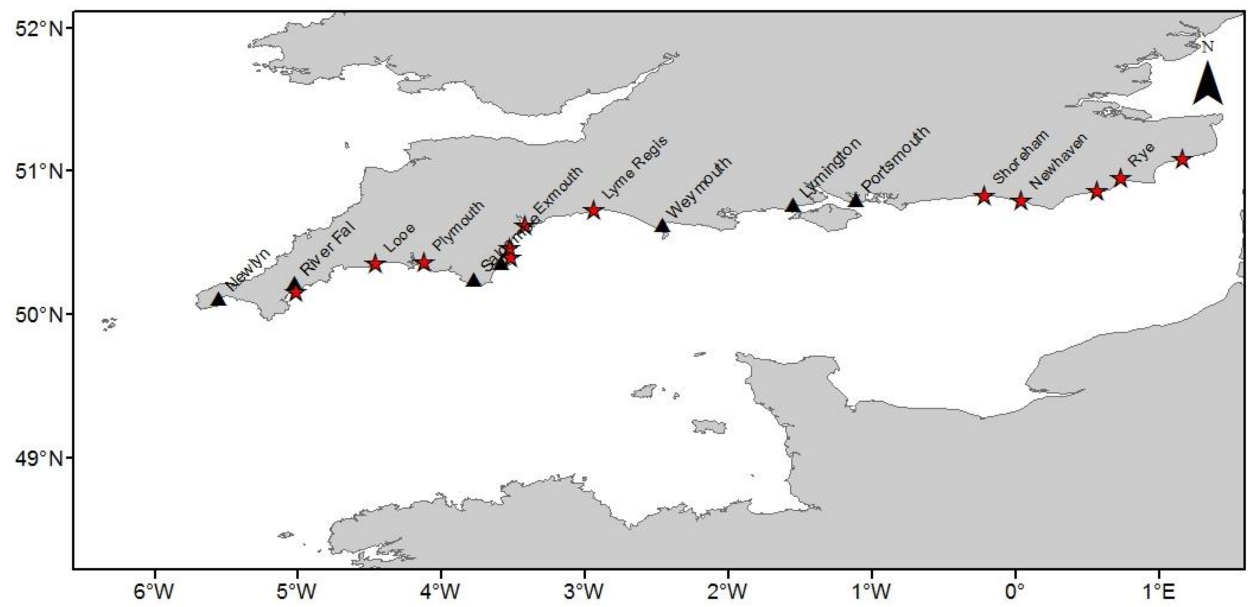

Figure 2.1: The location of ports in the English Channel where scallops are landed. Red stars indicate the home ports or landing ports of scallop fishermen that were interviewed. No scallop fishermen were interviewed from ports indicated by black triangles. Data provided by the Marine Management Organisation.

2.3 Data Analysis

Vessel characteristics

To assess the characteristics of the inshore and offshore fleets, a multivariate analysis of vessel characteristics was performed using PRIMER-E (Clarke & Gorley, 2006). The variables characteristics included in the analysis were: total number of dredges; maximum hours fishing per day; total days activity in last 12 months; minimum tow duration; maximum tow duration; minimum tow speed; maximum tow speed; minimum mean catch weight (scallops) per day; maximum mean catch weight (scallops) per day; minimum trip length; maximum trip length; maximum wind force fished; % grounds visited in last 12 months that have been fished previously; maximum distance travelled to fish; increase in distance travelled in last 10 years; vessel length; engine power; normal number of crew; minimum number of crew; maximum number of crew; zone in which majority of fishing has occurred (distance from shore). For the purposes of analysis, where skippers were unable, or chose not to provide an answer to a question, missing data was dealt with by entering the average response for vessels with similar characteristics e.g. similar LOA or total number of dredges.

Data were normalised and then a resemblance matrix of the similarity between vessels was created based on Euclidean distance. An ANOSIM test was used to ascertain whether characteristics were significantly different between vessels grouped by LOA (<15 m; 15-25 m; >25 m). The SIMPER function was used to ascertain the percentage similarity of characteristics within groups and percentage dissimilarity between groups. Skippers were asked to state which current management measures they felt were most effective (up to a maximum of 3). Each chosen measure was given a score of 1 and those not chosen were given a score of 0. This dataset was then subjected to a similar multivariate analysis to assess differences in opinions of skippers based on vessel size (<15 m, 15-25 m, >25 m).

LK Fishing polygons

Polygons of fishing activity recorded during fisher interviews were weighted according to the frequency of use indicated. The total number of fishing days per year (days yr^{-1}) was calculated for each polygon by summing the number of days visited over the 12 months. To provide a total value of fishing effort over the 10 year period, a weighting (0-1) was then applied. For example, if a skipper had fished in a polygon area once in the previous 10 years, a weighting of 0.1 was applied to the total days per year; whereas if the area had been fished biennially (5/10 years), a weighting of 0.5 was applied. This enabled integration of data from all interviews, which covered different time periods, to provide a measure of relative fishing intensity. Then all polygons were joined using the 'Union' tool, to produce a map of relative fishing effort over the 10 year period. Polygons for >15 m LOA and <15 m vessels were treated separately.

For each of the two length groups of vessels, fishing polygons were converted to a continuous raster layer using the mean of all values within a cell and the cell centre assignment method, with an output cell size of 0.025 decimal degrees (approximately 1.8 x 2.8 km at 50°N), as this was the scale at which the VMS data was aggregated (see below). If a skipper of an >15 m LOA vessel had drawn a polygon on the map that fell inside the 6 NM zone (0-6 NM from the shore) it was assumed to be a result of the coarse method of recording, rather than an intentional indication of fishing effort. To eliminate this error, the raster layer for the >15 m vessels was converted to a point grid layer of 0.025°, points that fell inside of this zone were removed and the resultant point data was then converted back to a raster of cell size of 0.025° using a mean cell assignment type.

Comparison of VMS and LK data

Up until January 2013, VMS data relate to vessels >15 m LOA only. Vessels of this size are not permitted to fish within 6 NM of the coastline in the English Channel due to restrictions on vessel length and the maximum number of dredges permitted per vessel (Appendix 1.1). For this reason, a 6 NM buffer was applied to VMS data and only records outside of this zone were retained for the comparison of VMS data with LK. Data from ICES sub-areas VIId, e and h (outside of the 6 NM mile zone) were included, as fisher polygons included fishing effort in all of these areas. Anonymised VMS point data (aggregated at a scale of 0.025°) for all UK and foreign scallop vessels, for the period 2005-2013 inclusive, were entered into ArcMap v.10. The sum of the time interval between VMS transmissions was used as a measure of relative fishing effort (total hours) over the time period and the point data were converted to a continuous raster using 0.025° grid cells. The VMS data were also aggregated using the 'Aggregate' tool, into grid cells of 0.1 and 0.3 decimal degrees (using the mean value) for comparison with the LK data.

The size of grid cell used for the aggregation of VMS data can over- or under-estimate the spatial extent and intensity of fishing activity (Piet and Quirjins, 2009; Gerritsen *et al.*, 2013). Therefore, vector analysis grids of differing cell sizes (0.1; 0.2; 0.25 and 0.3 decimal degrees) were created using the 'Create Fishnet' tool in order to visually assess the suitability of different scales. Due to the trade-off between resolution and accuracy, and the distortion that occurs at the boundaries of the data, 0.3° grid cells were the largest size of cell used for aggregation. The 'Zonal Statistics as Table' tool was used to obtain mean VMS and LK fishing effort values for each fishnet polygon, at each spatial scale. The resultant tables for VMS and fisher data were joined and the data points for each corresponding polygon plotted against each other. Correlations were tested for significance using a generalised linear modelling approach in R (R Development Core Team, 2008) and models were evaluated by checking for homogeneity of residuals. Visual assessment of frequency histograms of intensity values indicated that the data distribution was skewed towards low activity values. Therefore aggregated relative fishing intensity data at each resolution were displayed on maps in 7 breaks using the Jenks natural breaks classification (Jenks, 1967). This maximises the variation between groups in order to optimise visualisation of the relative spatial distribution of fishing activity.

2.4 Results

2.4.1 Vessel characteristics

An MDS plot and accompanying ANOSIM test of normalised vessel characteristics indicated that each group of vessels (LOA of <15, 15-25 and >25 m) had significantly different characteristics (ANOSIM: R=0.839, p=0.001, Figure 2.2, Table 2.2). A summary of vessel characteristics, by group is given in Appendix 2.2. SIMPER revealed high within group similarity for <15 m, 15-15 m and >25 m LOA vessels (82.9, 90.7 and 94.8 % respectively). The greatest dissimilarity occurred between <15 m and >25 m vessels (31.1 %), followed by 15-25 m and <15 m LOA vessels (25.8 %), with the least dissimilarity occurring between 15-25 m and >25 m LOA vessels (9.47 %).

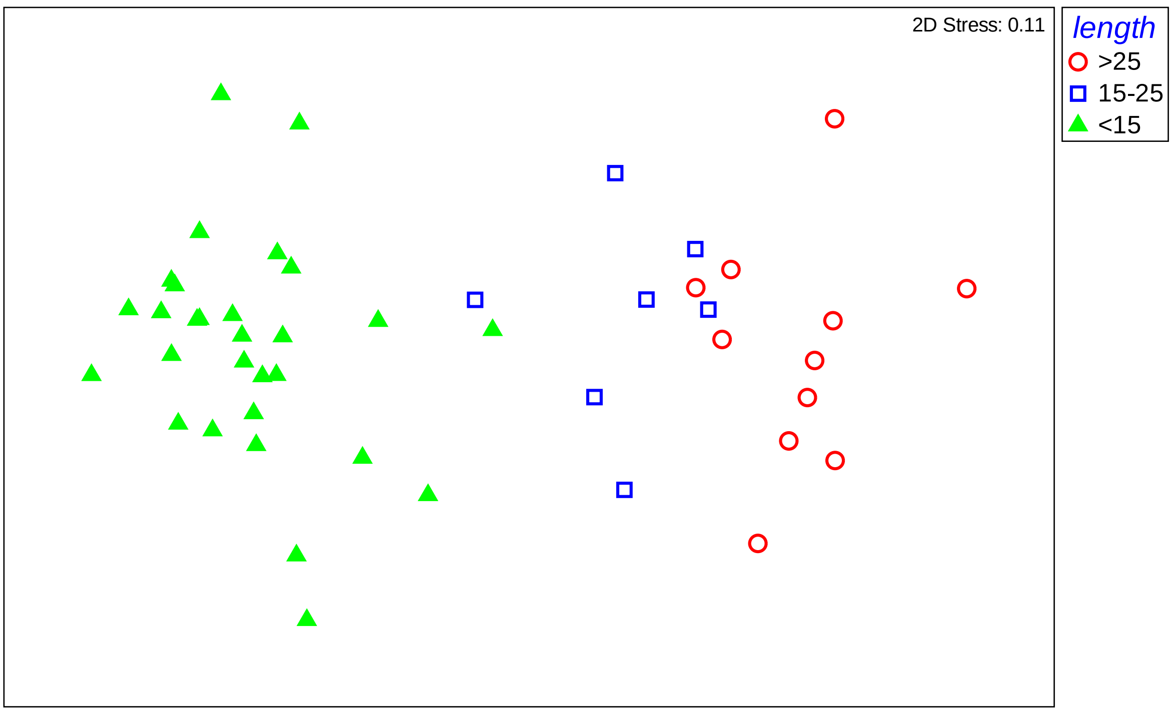

Figure 2.2: A multi-dimensional ordination plot of scores assigned to scallop vessel characteristics. Data was normalised prior to creating the resemblance matrix. Vessel characteristics included in the analysis are listed in the methods section. Symbols represent vessels <10 m LOA; 15-25 m LOA and >25 m LOA.

Table 2.2: Results from pairwise tests following ANOSIM testing for differences in vessels characteristics, between vessel length groups.

Pairwise tests	R Statistic	p value
>25, 15-25	0.28	0.007
>25, <15	0.98	0.001
15-25, <15	0.83	0.001

2.4.2 Fishing effort maps

As the grid cell size used for data aggregation increased, so does the estimate of the total area impacted. This effect was most pronounced for the VMS data, due to the high resolution of the original data set (Table 2.3). There was a marked increase in area impacted for the offshore LK data when the grid cell was increased from 0.1 to 0.3 decimal degrees. In contrast, there was a slight decrease in the overall area impacted for the inshore LK data when the grid cell was increased from 0.1 to 0.2 (Table 2.3).

As the grid cell size increased there was an increase in the correlation between relative fishing effort estimated from aggregated VMS and offshore LK data (Figure 2.3), however all correlations were significant (Table 2.4). As grid cell size increased so did the spatial boundaries of the fishery, and this effect was most evident using the VMS data (Figure 2.4). This resulted in grid cells covering areas that had not been identified as fishing grounds from LK polygons (Figure 2.5, 2.6). The boundaries of the data also became increasingly abstract.

Table 2.3: Estimate of the area impacted by the scallop fishery in the English Channel using VMS data and LK data for the inshore and offshore scallop fleets, with data aggregated at increasing grid cell sizes.

Data	Grid cell size (decimal degrees)	Area (km^2)
VMS	0.025 decimal degree cells	44,821
	0.1 raster	83,326
	0.3 raster	124,300
LK offshore	raw polygons	81,636
	0.1 raster	88,024
	0.3 raster	110,489
LK inshore	raw polygons	33,586
	0.1 raster	39,848
	0.2 raster	39,097

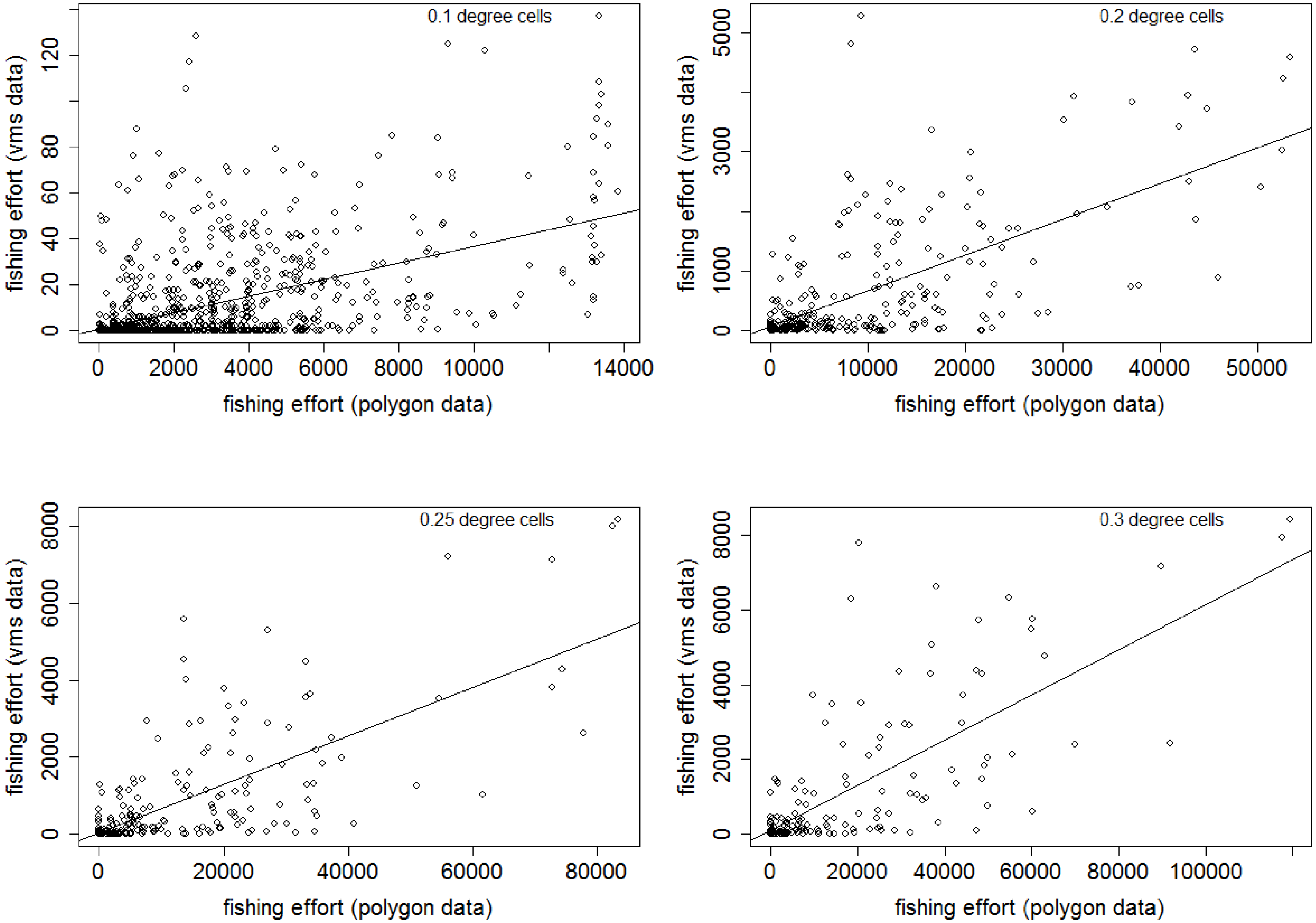

Figure 2.3: Plots of scallop dredge fishing effort values derived from VMS data and fisher polygons; data extracted at four different spatial scales: 0.1; 0.2; 0.25; 0.3 decimal degrees. Significant modelled linear regression lines are displayed. The r^2 and p-values are given in table 2.3.

Table 2.4: Results of linear regressions for fishing effort data calculated from VMS data and fisher polygons extracted at different cell sizes, d.f. = degrees of freedom.

Grid cell size (decimal degrees)	cell dimensions	cell area	R-squared value	d.f.	p value
0.1	7.2 x 11.1	80 km^2	0.28	1, 1083	<0.001
0.2	14.4 x 22.2	320 km^2	0.45	1, 332	<0.001
0.25	18.0 x 27.8	500 km^2	0.51	1, 231	<0.001
0.3	21.0 x 33.0	693 km^2	0.53	1, 175	<0.001

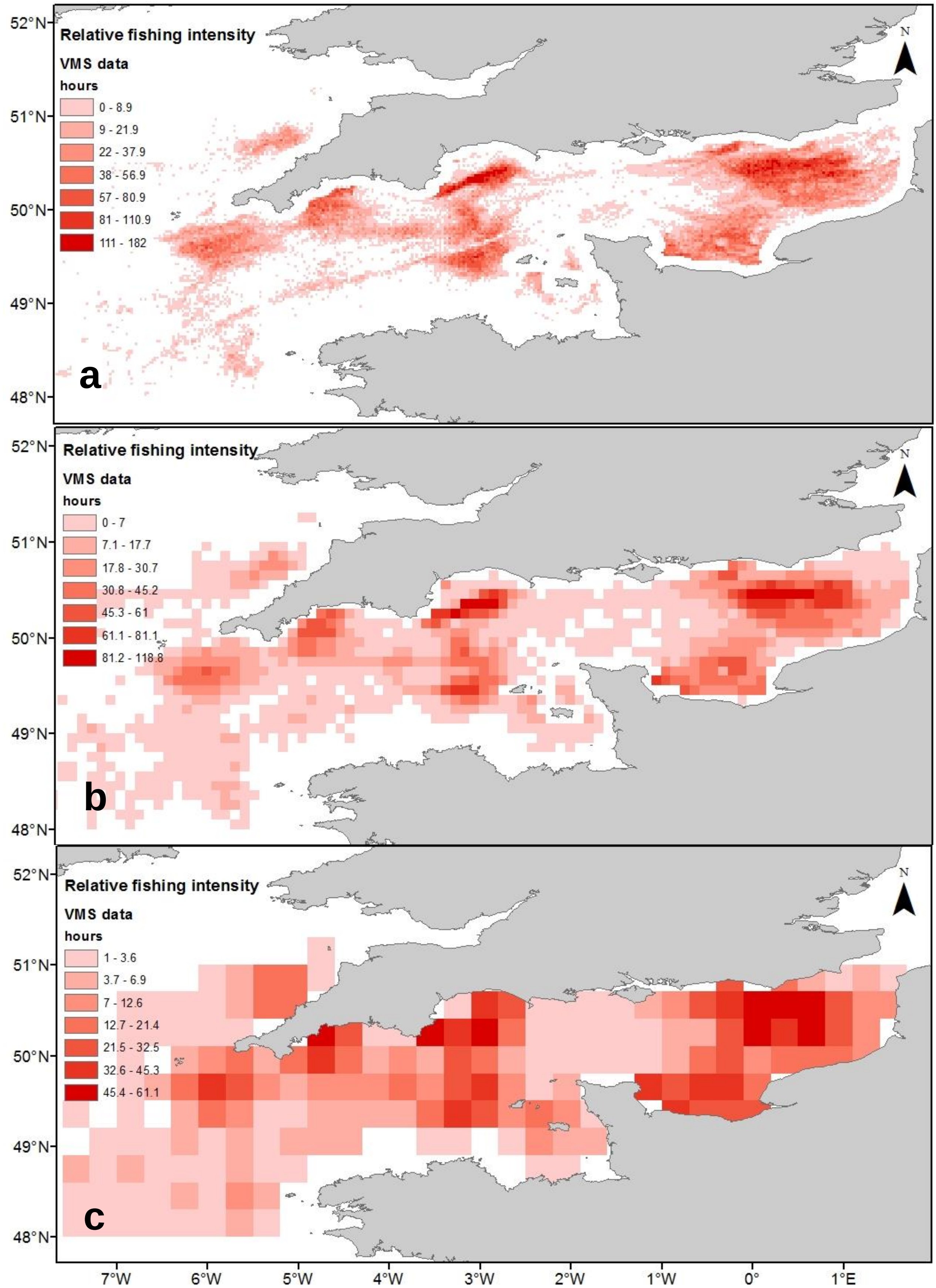

Figure 2.4: Relative scallop fishing intensity for all UK and foreign scallop vessels in the English Channel, expressed as the total number of hours fishing activity for the reference period 2005 to 2013, derived from VMS data, aggregated at: a) 0.025 decimal degree grid cells; b) 0.1 decimal degree grid cells; c) 0.3 degree grid cells.

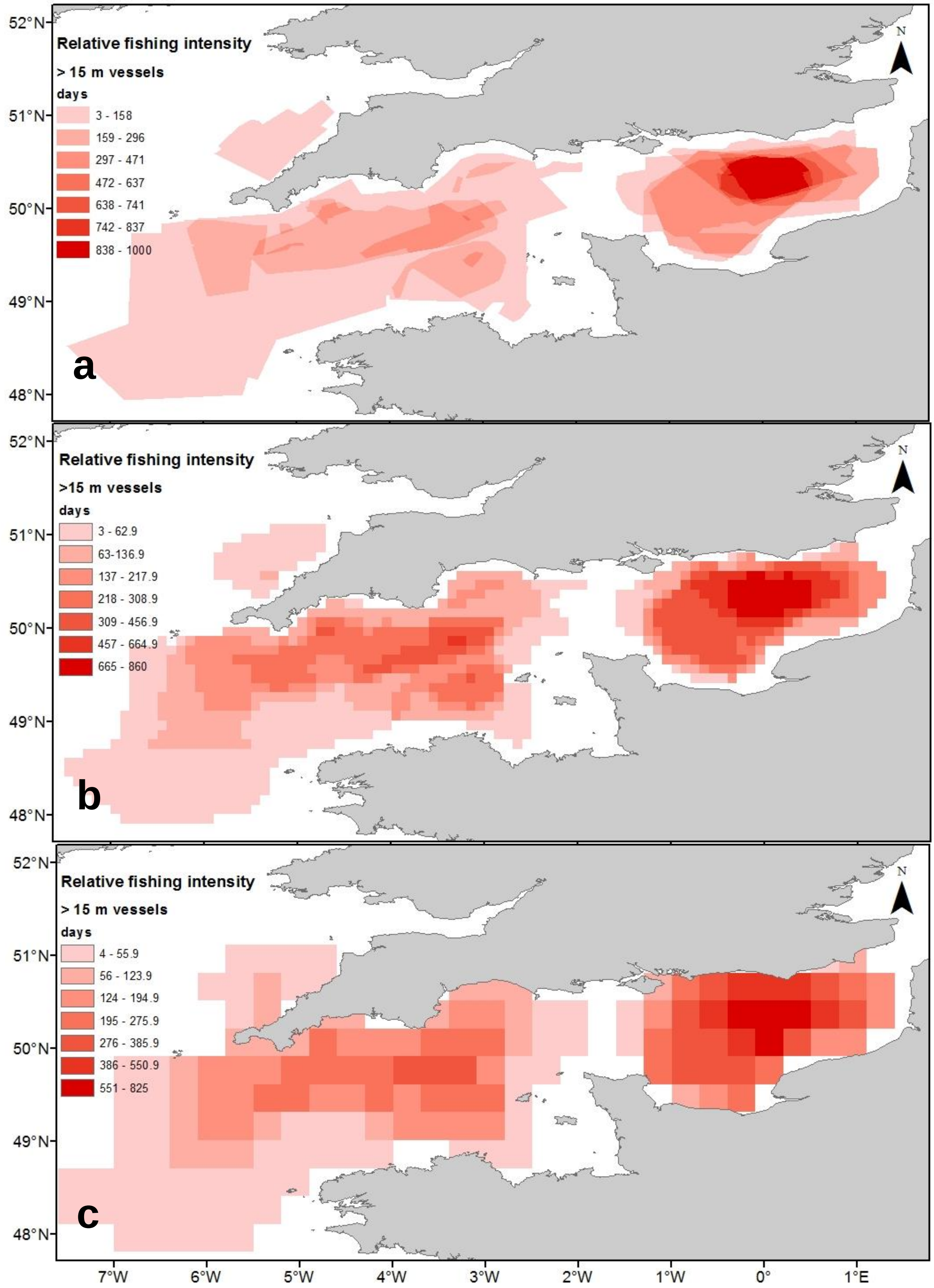

Figure 2.5: LK data for >15 m LOA scallop vessels in the English Channel displayed as: a) raw data (polygons); b) data aggregated at 0.1° grid cells; c) data aggregated at 0.3° grid cells. Data represents the total number of vessel days fishing over a 10 year reference period. Darker shading indicates higher values of fishing intensity.

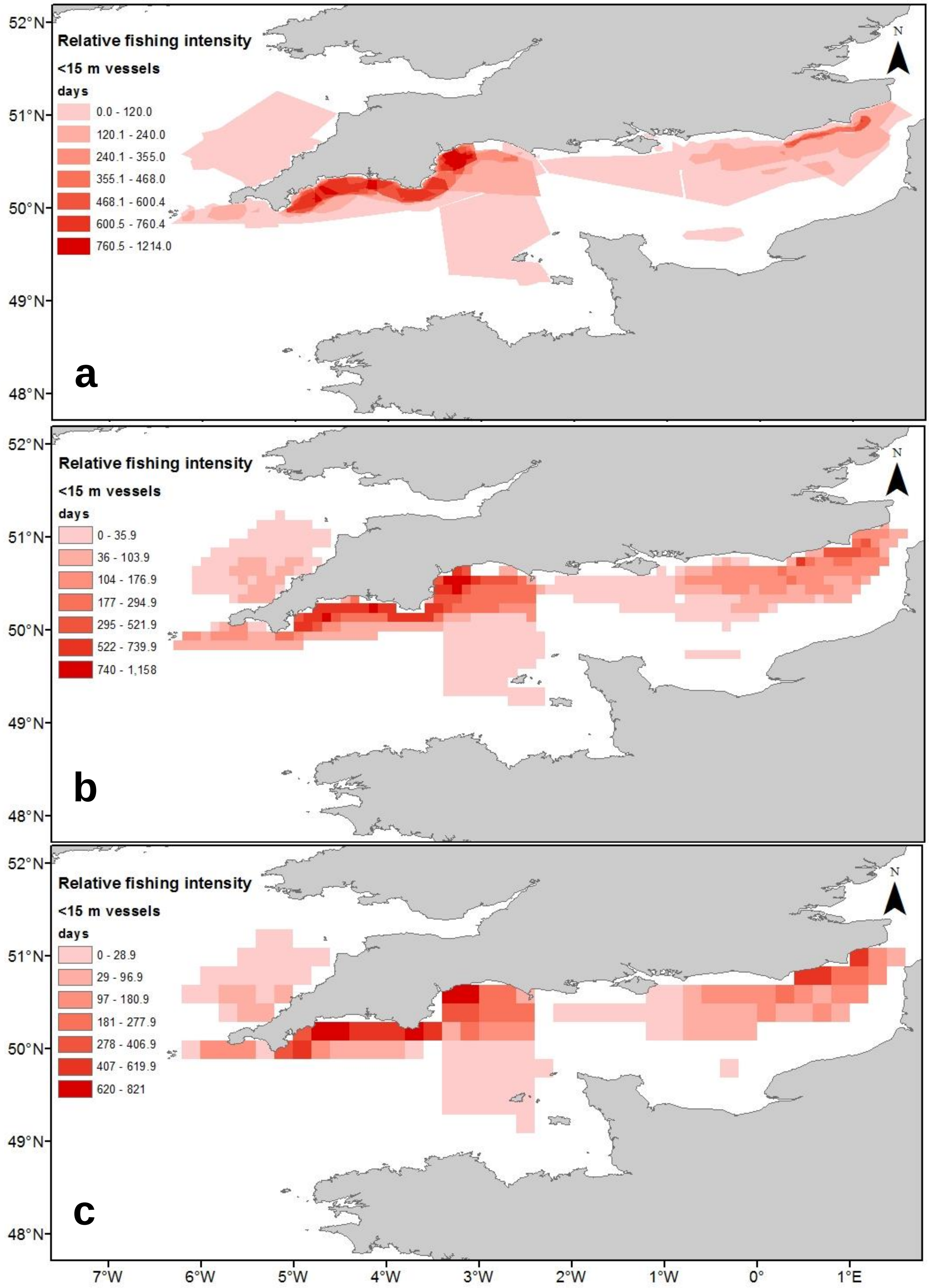

Figure 2.6: LK data for <15 m LOA scallop vessels in the English Channel displayed as: a) raw data (polygons); b) data aggregated at 0.1° grid cells; c) data aggregated at 0.2° grid cells. Data represents the total number of vessel days fishing over a 10 year reference period. Darker shading indicates higher values of fishing intensity.

2.4.3 Information derived from the industry

Ninety percent of the skippers of the <15 m LOA vessels were also the vessel owner. These skippers/vessels are referred to as the 'inshore fleet'. For >15 m vessels this dropped to 38 % and these are referred to as the 'offshore fleet'. Most inshore skippers (79 %) did the majority of their scallop fishing between 3-6 NM of the coast and 17 % did the majority of their fishing in the 3-12 NM zone. One skipper fished both inside and outside 12 NM. Fewer offshore skippers did the majority of their fishing in the 6-12 NM zone (17 %); one skipper fished in both the 6-12 NM and 12+ NM zones; and 78 % did the majority of their fishing outside 12 NM.

Patterns of use and choice of fishing ground

A minority (24 %) of inshore skippers fished in both areas VIId and VIIe, while the remainder fished only in the same area as their home port. This was in contrast to 85 % of offshore skippers that fished in both areas VIId and VIIe, which confirms the common understanding that >15 m LOA vessels tend to be nomadic while <15 m vessels are more locally restricted in areas where they fish.

When skippers were asked to state three factors that had the greatest influence on their choice of fishing grounds, 'weather' was the most frequent answer. 'Weather' relates to wind speed and direction (temperature and precipitation do not influence scallop fishing). Recent catch levels, previous fishing success in an area, and the quality of the catch (e.g. percentage of meat weight and gonad condition) were jointly important factors in choice of fishing ground. 'Tide' (state (spring/neap) and direction) was cited by 22 % of all skippers as an influencing factor. Other less frequently cited responses (in decreasing order of importance) included: season (which relates to both variations in scallop condition and seasonal closure of certain grounds); distance from landing port; location of other fishing vessels (either following or avoiding other scallop vessels, or avoiding other sectors such as vessels with static gears); substrate type. For vessels >15 m LOA, effort restrictions and available fishing days for ICES area VII also influenced the location of fishing.

The majority of skippers shared knowledge of good fishing grounds with a selected number of other skippers, while 14 % of skippers preferred to keep this information to themselves. Vessels >15 m LOA are required by law to have AIS (Automatic Identification System), to enable the location of the vessel to be electronically tracked while at sea. Real-time vessel positions are publicly available on the internet (e.g. www.marinetraffic.com). Skippers of >15 m LOA vessels reported that if word spread of a large catch being landed, many other vessels would subsequently visit the fishing ground from where the catch originated. When this occurs, it is possible for a number of the larger scallop vessels to reduce the density of scallops in a patch below economically viable levels within a week or two of concentrated fishing effort.

Skippers reported that they rotated activity between fishing grounds either seasonally or for time periods >1 year. Reasons for rotational fishing included allowing replenishment of scallops of minimum landing size (MLS) through growth of under-sized scallops, seasonal variation in the size of the scallop roe and quality of scallop meat and forced rotations through the use of static fishing gear in an area which prevents or inhibits scallop dredging. Over half of offshore skippers (61 %) actively prospected for new scallop beds while fishing, however many of the inshore skippers had fished in the same areas for many years and offshore skippers reported that the amount of time they spent trying new grounds had significantly reduced in recent years. This reduction in time was attributed to the increase in fuel costs that made trips to distant grounds unviable (due to uncertain financial returns) as well as recent limitations on the number of days available to fish in area VII.

With regard to scallop condition (scallops with a fully developed roe are more valuable and considered in better condition than scallops that have recently spawned), 68 % of inshore skippers preferred to avoid areas where scallops recently had spawned. However, one inshore fisher said he would not avoid areas of recently spawned scallops as he felt the price received for the catch was fairly consistent whether roes were full or not. Vessels that had the facility to convert to beam trawling gear did so if scallop condition reduced or catch rates dropped. Most offshore skippers (57 %) actively avoided areas where scallops had recently spawned, whereas 15 % would carry on fishing regardless and placed more importance on the quantity caught, rather than the quality.

Impact of legislation

Skippers of inshore vessels reported that in recent years their fishing had been impacted by area closures in Lyme Bay; Start Bay; Torbay; Falmouth; the Scilly Isles; Cardigan Bay and Caernarfon Bay in Wales as a result of habitat conservation measures. Due to these closures, 28 % of inshore fishers reported having to travel further from their home port to fish, while 72 % travelled the same distance to fish as 10 years ago. This has resulted in more time spent at sea, increasing fuel costs and greater vulnerability to weather conditions (closures generally occur in areas close to shore that provide greater shelter from the wind).

Offshore skippers reported that in recent years, legislation has displaced their fishing activity in England, Scotland, the Isle of Man and the Channel Islands (due to legislation on the maximum number of dredges allowed); the Baie de Seine (there was an agreed temporary closure of this fishery in 2013 and 2014 in return for effort (kW days, under the Western Waters management regime (Council Regulation (EC) No 1415/2004), from the French fleet); and Wales (a reduction in the maximum number of dredges allowed has effectively closed the entire Welsh fishery to larger vessels). Since 2011 the >15 m UK fleet have been restricted in the number of kW days available per year in ICES area VII. This restriction on effort has caused both an increase and a decrease in distance travelled by vessels. When vessels have used all their allocated fishing days in area VII, activity is displaced to other areas and vessels travel as far away as Scotland to fish. However, as soon as an >15 m vessel leaves a port on a fishing trip, the entire duration of the trip is deducted from the available days, even if the vessel is steaming (and not fishing) on departure and return from fishing grounds. This means that vessels are more likely to fish closer to a landing port when in area VII, to reduce this loss of fishing time, hence reducing the spatial footprint of the fishery in this area. As a result of the above issues, 31 % of offshore skippers reported that they now had to travel further to fish, while 21 % said they now travelled less distance, in order to conserve fuel and effort days. The remaining skippers reported no change in the distance they normally travel to fish over the last 10 years.

Technical measures

For inshore vessels, the number of dredges used varied depending on the location of fishing. For example, vessels are restricted to 6 dredges per side within all Inshore Fisheries and Conservation Authority (IFCA) regions in the English Channel. IFCAs are responsible for fisheries management within 6 NM of the coast. Vessel length and engine size also dictate the maximum possible number of dredges that can be used, and in Wales the maximum engine size permitted is 221 kW. Skippers of smaller vessels may choose to reduce the number of dredges used, based on local restrictions, if they believe that the quantity and quality of the scallops in the area will negate the inevitable reduction in overall catch. For skippers of offshore vessels, the total number of dredges was only reduced when fishing in Scottish waters, due to the regulation that allows a maximum of 14 dredges per side outside 12 NM;

whereas in the rest of the UK, there is no limit on the total number of dredges used outside of 12 NM.

Legislation in Northern Ireland and the Isle of Man dictates that the internal diameter of the dredge belly rings must not be less than 75 mm. This was the smallest internal diameter reportedly used by vessels in the English Channel, however 31 % of skippers of inshore vessels used rings of a larger diameter (up to 100 mm) in order to reduce the catch of scallops under MLS, or reduce the amount of stones retained. Some offshore skippers (42 %) also used >75 mm diameter belly rings.

The engines of 21 % of all vessels had been reduced in size (power) since purchase for reasons such as conversion from a beam trawler or to provide greater fuel efficiency. Only two vessels had slightly increased engine power due to replacement of an old engine. In the last 10 years, conveyor belts and tipping rails have been installed on some vessels. Both of these measures improve safety. Conveyor belts also enable easier and quicker sorting of the catch and prevent the crew having to bend or kneel down on the deck. One skipper had purchased a larger vessel (15 m LOA) that enabled him to do longer trips, as a direct result of local/inshore area closures. Skippers of some offshore vessels had improved their fishing strategy through the installation of acoustic systems. Such systems allow the substrate type on the seabed to be identified and recorded. This allows the skipper to target areas where there are more likely to be scallops and avoid areas where catches are likely to be low.

Views on management measures

Skippers were asked what they felt the three most effective management measures to conserve scallop stocks would be. The total number of measures cited, and the number of responses for each are given in table 2.5. Restriction on total dredge number; engine size; and temporary closures were considered the most effective management measures by skippers of inshore vessels. For offshore skippers temporary closures; restriction on total dredge number and a cap on the number of scallop licences were considered the most effective measures to conserve scallop stocks. ANOSIM testing revealed significant difference in responses between skippers of <15 m vessels and those of >15 m vessels (R=0.106, p=0.04); however the difference between skippers of 15-25 m vessels and >25 m vessels was non-significant (R=0.099, p=0.056). The overlap between the groups can be seen in figure 2.7.

Table 2.5: Management measures considered by scallop fishermen to be the most effective at conserving scallop stocks in the English Channel. Numbers reflect the total number of responses for each measure.

Management measure	Total responses <15m vessel skippers (n=20)	Total responses 15-25m vessels skippers (n=8)	Total responses >25m vessel skippers (n=11)
Total dredges restriction	14	7	8
Vessel size restriction	10	2	3
Seasonal/temporary closures	9	6	9
Engine size restriction	6	1	1
Curfew	4	1	0
Restricted effort	2	1	2
Minimum landing size	2	1	0
Minimum belly ring size	1	1	1
Cap on licences	0	2	4
New dredge design	0	2	0
Maximum number of teeth	0	1	0
Permanent closed areas	0	0	1
TACs	0	0	1

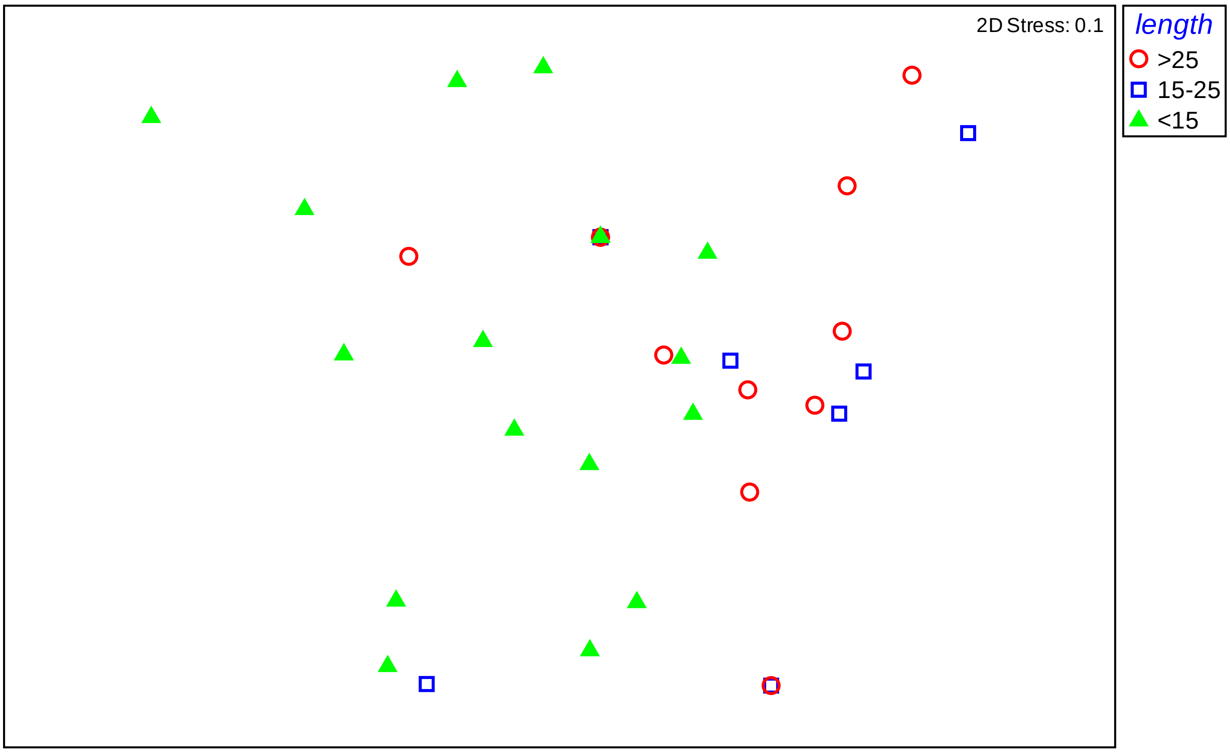

Figure 2.7: A multi-dimensional ordination plot of skipper opinion on the effectiveness of the management measures listed in table 2.4. Symbols represent skippers of vessels <10 m LOA; 15-25 m LOA and >25 m LOA.

Catches and profit

There was no consistent pattern in reported catch with most inshore skippers reporting no change (31 %), a decrease (24%), or an increase (45 %) in catch levels over the last 10 years. A similar pattern existed for offshore skippers who reported either no change or fluctuating catch levels (37 %), increased catches (32 %) or reduced catches (10 %). Increased catches were attributed to: the developing experience of the skipper; improved technology such as the use of sonar for seabed substrate discrimination and conveyor belts; a particularly high recruitment of scallops to the fishery in the eastern English Channel in 2009-2010; introduction of management restrictions such as the inshore curfew and dredge limitations; and fewer vessels (inshore fishery only). Where skippers reported their catches had decreased, this was attributed to an increase in vessel numbers and restrictions on days at sea (both in relation to the offshore fishery). Some fishermen reported an increase in profit since 10 years ago; however increases in the cost of fuel and gear, combined with a reduction in the sale price of scallops resulted in 26 % of skippers reporting a decrease in profit. Skippers who were not the owner of the vessel were unable to provide financial information and some chose not to disclose such information.

In Rye (East Sussex), a combination of environmental and economic factors had a major impact on scallop fishing. Fishermen reported a reduction in the amount of scallop fishing they had done in recent years and none of them had fished for scallops the previous winter (2012). This was due to windier conditions, lower catch levels and a reduction in the sale price of scallops that made it uneconomical and practically challenging for them to profit from the fishery.

Sustainability

When asked if the fishery was currently fished at a sustainable level, 17 % of inshore skippers disagreed; 10 % were unsure and 65 % felt that current fishing activity was sustainable. For this sector of the fleet, skippers felt that inshore regulations protected the stocks and weather also restricted the total amount of fishing activity. One skipper mentioned that some vessels were switching to scallop fishing due to a shortage of fish quota. Some offshore skippers (30 %) felt that the current level of fishing in the English Channel was unsustainable, largely due to the increase in the number of vessels, some were unsure (15 %), however more than half (55 %) felt that the current level of fishing was sustainable.

When asked if they felt the fishery was 'at risk' of overfishing, 45 % of inshore skippers disagreed; 6 % were unsure and 34 % agreed (one of these skippers felt this was true for the offshore, but not the inshore fishery). This risk was associated with displacement of effort causing greater pressure on stocks in open areas; concern over the number of >15 m vessels entering the fishery; and beam trawlers switching to scallop fishing due to a lack of fish quota. Some offshore skippers (40 %) disagreed that the fishery was at risk, attributing this to consistent landings and the large area that the fishery covers, however the majority (60 %) felt that the fishery was at risk largely due to the lack of management measures, the increasing number of vessels joining the fleet and the concentration of effort in certain areas.

A number of fishermen did not express disagreement with any of the current management measures, however some disagreed with various measures (Table 2.6). The final question was open-ended, asking for any further comments on the management of the fishery; or anything that had been discussed during the interview (Table 2.7). This demonstrated that opinions can vary widely both within and between fleet sectors and fishers' often had novel suggestions for management.

Table 2.6: Comments from UK scallop fishermen (skippers of <15 m LOA vessels) on the current management of the English Channel king scallop fishery.

Current measures that fishers disagree with (<15 m vessel skippers)	**Other comments**
MCZs and the Lyme Bay SAC. There used to be 70ft vessels fishing in within the 6 NM limit and if sea life is present now, why won't it be there in 10 years' time?	Gear restrictions are fair to protect grounds and stocks. Too many big boats are switching to scalloping.
The 7-7 curfew as it does not work from a tidal harbour.	Areas should be open for 1 month, closed for 11, which will always allow recovery.
Permanent closures (e.g. Lyme Bay) as this allows starfish to dominate the seabed	Would like to see a 3 mile limit for smaller boats e.g. 4 dredges a side
West Bay closure	Concerned about MPAs now regarding ground closures
Closed areas for pot boxes as this agreement favours potters over scallopers	Would like all boats to have VMS
There are more restrictions on smaller boats than larger boats which is not fair	Agrees with the curfew
Legislation that prevents the use of rubber to prevent wear to the steel belly rings. This affects people who were using them honestly.	Has VMS and thinks areas with coral should be opened up to boats with VMS. Rotational closures would be better than permanent closures
Having to land bycatch every time they land. They are not allowed to land it in a different port.	Can catch whole of monthly sole quota in one day in Plymouth, so has to throw it away
Would like to be able to fish inside 3 NM (eastern English Channel), thinks there will be plenty of scallops there as fishing is currently not allowed there.	Larger boats are needed to supply the market but effort needs to be managed. Need to balance supply and effort between the large and small boats.
There should be 110 mm landing size in ICES sub-area VIIe, in line with VIId	The fishermen asked for the 6 month summer closure about 8 years ago and it works well for them (Sussex IFCA)
Disagree with 'Carte blanche' closures of particular areas e.g. MCZs, without proper reasons and not being reopened even when vessels have inshore VMS on-board.	Wants to see closed areas reopened on a controlled basis with VMS. Doesn't have Succorfish (inshore VMS system) at the moment as there is no benefit but he would get it if it led to fishing grounds being reopened.
Would like to see more flexibility in the 12 hour curfew.	Could possibly implement a maximum landing size, e.g. 150 mm as scallops of that size are old and not good for sale but can still breed.
Around Selsey Bill, larger boats can fish up to the 6 NM limit. Inside 6 NM there should be <6 dredge per side limit.	Would like harmonisation with the French summer closed season (which has been in place for 18 years)
	More boats have gone scalloping in recent years as there is no quota and also larger boats e.g. 6 a side. Would like to see a 3 or 4 a side limit inside 6 NM limit (currently 6 aside is allowed between 3 to 6 NM).

	Caps on licences would be an effective management measure. Would like to see more management and effort control. The inshore potting agreement in Devon works really well so why can’t this be implemented across the Channel. Dredge limits should be reduced on big boats. Need regulations between 6-12nm as big boats are still allowed in. Engine size should be 300hp max inside 6nm. Should have 85mm ring size from 0-6nm.

Table 2.7: Comments from UK scallop fishermen (skippers of >15 m LOA vessels) on the current management of the English Channel king scallop fishery.

Current measures that fishers disagree with (>15 m vessel skippers)	**Other comments**
Area VII effort restrictions	Does not disagree with days at sea but would like 60 days per quarter, which would allow for 3 landings and a week off per month for the crew.
Closure of the Eastern Channel fishery to the 1st October is an arbitrary period. The closure should be based on research to protect stocks while they are spawning	Has received unfair/ disproportionate charges for minor offences compared to other boats, feels he has been singled out for prosecution by the authorities
Lyme Bay closure - as starfish will overtake the ecosystem	Thinks fishing has been better in last few years due to increased water temperature e.g. it was 18.5°C in Bay de Seine last year. Small boats suffer as the larger boats fish on the 6 NM limit and fish out the stocks. E.g. In the Irish Sea the smaller boats can't fish in bad weather but larger boats can. They may have to look at buying a fish license for the boat as they have a category C licence at the moment so can only keep 5 % of their bycatch.
Disagree with how the 'days at sea' regime is managed (in ICES area VII) and the lack of a cap on scallop licences	England should match the French winter closure of sub-area VIId. Should not have permanent closures as this allows starfish to proliferate.
10 dredges per side should be allowed inside 12 NM	The quality of scallops in the eastern English Channel has decreased and good crew can't afford to stay working on the boats as wages have decreased.
There should be seasonal management, not just days at sea	Need to prevent more boats from entering the fishery. The <15 m fleet should be managed in the same way as the >15 m fleet.
Fully closed areas inshore (should be seasonal closures). Should be greater dredge limitation e.g. 14 per side outside 12 NM limit	If they can make a living using 12 dredges a side so can other boats
Disagree with bycatch restrictions on fish, should be controlled by TAC, not bycatch	If he is stopped from fishing inside 12 NM due to dredge limits, that would affect him.
Yes, in the sense it is not a level playing field for all	Would like to see 14-side limit outside 12 NM in all areas.

Being restricted on total number of dredges inside 12 NM	There should be a limit on entry of <15 m vessels to fishery as well as limitations on effort as they can have a large impact close to shore (also as most have cat C licence they are discarding most or all of the fish caught).
Would like a higher effort allocation	Area VII is too large, which doesn't make sense for management. Scallop stocks on the east coast of Scotland can't cope with the extra effort displaced from the English Channel. Days at sea should be allocated in each area to spread total effort.
Each vessel should get its full allocation of days at the start of the year to manage themselves.	Would like to see rotational closures for the fleet rather than closing a whole area e.g. the whole of VIId in the winter. He would like each vessel to be allowed to manage its own days and where they choose to fish.
Disagrees with year-round fishing. The fishery should close from when the scallops begin to shoot their roes until November. Fishing should rotate between the east and west English Channel.	Restrictions on foreign boats are needed as well as UK vessels otherwise the management is pointless. Sometimes he can't fish on home grounds as he has no effort remaining, but foreign boats can. When his scallop days at sea have run out he is forced into beam-trawling but he is not happy about this.
There are too many management measures, making things too complex. Effort limitation is enough to deal with	
The maximum dredge number should align with Scottish regulations (14 per side)	

2.5 Discussion

Semi-structured interviews create open dialogue and offer opportunities for scientists and policy makers to better understand socio-economic drivers for the fishing industry and inform long-term solutions to issues in fisheries management (Yates *et al.,* 2014). Older fishers that were interviewed had decades of experience and were able to impart very specific knowledge for stocks and areas they had fished for many years. This sharing of knowledge is invaluable to scientists, particularly for areas where historical data is lacking, as in the case of the English Channel scallop fishery for which only landings data exist. The reliability and accuracy of LK varies with context and species (Gilchrist *et al*., 2005; O'Donnel *et al.,* 2012). However, for a species such as the king scallop that has a consistent association with seabed habitat the reliability of LK data can be high (Shepperson *et al.,* 2014).

The terms 'inshore' and 'offshore' are often used when describing fishing fleets (e.g. Charles & Reed, 1985). In the context of the scallop industry, inshore and offshore vessels are characterised by size (e.g. vessel length; engine power), which in turn dictates the number of dredges that can be towed from the vessel. The maximum number of dredges used by inshore vessels was the minimum number used on offshore vessels. Average towing speed and tow duration were similar for both fleet sectors, and therefore are likely to reflect the most efficient fishing practises. Information such as this can be used by scientists when conducting surveys with scallop dredge gear. LK data revealed differences in fishing behaviour and opinions on management between fleet sectors (Nutters & da Silva, 2012). Unlike the inshore fleet, larger offshore vessels complete longer trips, travel greater distances and can fish in more extreme weather conditions. This provides more fishing opportunities and greater access to a wider variety of fishing grounds and the number of >15 m vessels that target king scallops has remained fairly consistent over the last decade. Vulnerability to economic drivers of fishing activity varies between fleet sectors (Tuler *et al.,* 2008). As many of the smaller vessels tend to fish scallops only when it is most profitable they are likely to be more sensitive to such forces, which may explain why the number of <15 m vessels has fluctuated between 45 and 93 over the last decade.

Assessing the validity of Local Knowledge

Local knowledge derived from half (54 %) of the offshore fleet gave a good visual representation of the maximum spatial extent of fishing activity when compared to 100 %

VMS coverage, however the estimate of the total area impacted will be inflated due to the coarse resolution of the LK polygons. LK data is limited by the precision at which individual fishers record fishing grounds and the accuracy is also affected by sample size and analysis grid resolution (Shepperson *et al.,* 2014). In relation to both the VMS and LK data, as the grid cell size used for aggregation increases the border of the area of impact becomes increasingly abstract, which can be critical if overlaps between fisheries activities and conservation features need to be identified. Thus, the smallest grid cell size may be useful when delineating fishing grounds. Using larger grid cells reduces the inherent variability in the data and mitigates against individual error or deception in reporting, however the spatial extent of the fishery can be over-estimated (Shepperson *et al.,* 2014). When data was aggregated at grid cells of 0.3 decimal degrees the estimate of the area impacted by the offshore fishery increased by 35 % in comparison to the raw polygon data. For the inshore fleet, the estimate of area impacted increased by 16 % when the data was aggregated using grid cells of 0.2 decimal degrees. Hence, there is a necessary trade-off when evaluating spatial patterns of fishing intensity and the appropriate scale should be chosen depending on the intended use of the data.

When considering the distribution of fishing effort, there were significant correlations between the LK and VMS data. Correlation with VMS data increased with increasing cell size, with moderate correlation (0.51; 0.53) at the two largest grid cell sizes (500 and 693 km^2 respectively). Using a larger grid cell size when assessing fishing intensity will also buffer against inaccuracies in the data (Shepperson *et al.,* 2014). In the study by Shepperson *et al*. (2014), grid cells of 25 km^2 were the largest used in analysis and gave the highest agreement between LK and VMS data. In the present study, the smallest grid cells used were substantially larger (approximately 80 km^2), therefore we consider that the scale of analysis of LK data will yield reasonable accuracy for the English Channel scallop fishery. Shepperson *et al*. (2014) also found that a larger sample size of the fleet increased the accuracy of estimated fishing intensity. A subsequent reduction in sample size from 100 % of the fleet to 33 % led to a 9 % reduction in the Kappa agreement statistic, which accounts for the likelihood of chance agreement between datasets (Cohen, 1968). In the latter study, the resultant kappa value based on a 33 % sample was 0.57 using a 25 km^2 grid cell, which falls just below the threshold of 0.6 that is considered to indicate ‘substantial agreement’ between data sources (Landis & Koch, 1977). Based on this, for the offshore fleet in the present study, of which 54 % were sampled, the largest grid cell (693 km^2) is considered to provide a reasonably accurate estimation of the distribution of fishing effort.

Individuals demarcated fished areas with varying levels of precision; offshore skippers tended to map activity with a few large polygons, whereas inshore fishermen more frequently drew small polygons in specific locations. In the western English Channel, offshore fishing activity is sparse (indicated by patchy VMS data). Offshore skippers drew polygons that covered large areas of the western English Channel to reflect the maximum range that they had travelled to fish in the previous 10 years. However, this failed to represent the fine scale detail in fishing activity that can be revealed by VMS data and led to an overestimation of the total area impacted by the offshore fishery. It also resulted in many zero hour VMS records overlaying low intensity LK data and therefore reducing the overall correlation between the two datasets. Thus, it appears that the representation of the extent of the offshore fishery using LK is a conservative method. There was greater visible correlation between the VMS and LK data in areas of concentrated fishing intensity; therefore the LK is likely to be more accurate where fishing activity occurs most often.

As VMS data do not exist for the inshore fishery a comparison with LK data was not possible. The inshore fishery is more spatially concentrated than the offshore fishery. Skippers were interviewed at a range of landing ports along the coast; therefore the maximum extent of the inshore fishery is likely to be reasonably accurate. This is supported by the comparison of VMS and LK polygon data for the offshore fleet, which gave a good visual representation of the maximum spatial extent of the offshore fishery. It is possible that some of the inshore activity may have been missed as some ports were not visited, or relative effort is underestimated in certain areas as only 25 % of the inshore fleet were sampled. No inshore skippers from Southampton were interviewed; however the scallop fishery in this area is very limited. A byelaw in the Southern IFCA district restricts vessels to 12 m LOA or less, towing 3 or 4 dredges in total and there are only 5 or 6 vessels that land scallops into Southampton (Neil Richardson, Southern IFCA, pers. comm.). However, fishing grounds to the east of the Isle of Wight were identified by a Welsh skipper that had fished in that area. Many of the polygons drawn by different fishermen overlapped indicating that skippers visited the same traditional fishing grounds. Therefore, the LK data would seem to provide a good representation of the spatial extent of fishing activity of the inshore fleet. An increased sample size would increase the accuracy of estimates of relative fishing intensity but is unlikely to alter the predicted spatial extent of inshore fishing activity by a significant amount. Verifying the map of inshore activity with local fishers would substantiate this.

'Hotspots' of scallop fishing activity are highlighted by the inshore LK map, which reflect traditional fishing grounds along the coast of south-western Cornwall, Devon and Dorset. There is less inshore scallop activity in the eastern English Channel; however the highest levels of activity are concentrated close to the Sussex shoreline (Vanstaen *et al.*, 2010; Vanstaen & Silva, 2010). Areas of lower activity for the inshore fleet tend to be in locations that are further from shore or landing ports, or are only visited during extended periods of good weather, such as the Channel Islands (as smaller vessels are more vulnerable in windy conditions).

Changes in fishing behaviour

For the offshore fleet, a restriction on the annual fishing effort (measured as kW days) for the >15 m fleet in ICES area VII has led to a reduction in the spatial footprint of this sector in recent years (Campbell *et al.*, 2014). Although there has always been an annual limit on the amount of effort available to the fleet, this limit has never been reached prior to 2011. An increase in the total number of >15 m LOA vessels, and an increase in the overall engine capacity of the fleet has created a situation where the number of kW days is not sufficient to allow each >15 m scallop vessel unlimited fishing in ICES area VII. As well as restricting overall fishing effort in the English Channel, this has also influenced the behaviour of the fleet. As soon as an >15 m vessel leaves port, this counts as fishing time to be deducted from the annual allowance. Some fishing grounds in the far western English Channel require 12 or more hours of steaming to reach, therefore skippers no longer travel to distant fishing grounds to avoid this loss of fishing time.

Although fishing patterns can be predicted through social or economic drivers, unpredictable natural events have the potential to alter the behaviour of the fleet and the spatial extent of fishing activity. Traditionally, offshore skippers reported that they would target the more sheltered grounds of the eastern English Channel during the winter months, when the scallops were in the best condition (before the spawning season, which occurs between January and March). The fleet would then move to fishing grounds in the western English Channel in the summer months; when weather conditions were better and scallops were in good condition. In 2009-2010 there was an unusually large recruitment of scallops on fishing grounds in the eastern English Channel. Skippers reported such high catches that it made economic sense to remain fishing those grounds for a 12 month period, rather than moving west during the summer months.

Socio-economic considerations

LK provides valuable qualitative information on socio-economic changes in fisheries over time (Murray *et al.*, 2006). Different sectors of the fleet experience varying levels of sensitivity to environmental or socio-economic pressures (Tuler *et al.*, 2008). Although governments give consideration to the impacts of management on fishers and fishing communities, often this fails to manifest as social objectives in fisheries policies (Symes & Phillipson, 2009). This affects inshore fishers in particular who are more likely to suffer in the event of reduced fishing opportunities. Although environmental factors (weather; tide; season) were the strongest influencers of choice of fishing ground, interviews revealed that the last 2-3 years have shown step-changes in fishing activity for both the inshore and offshore fleets related to economic and legislative drivers. Nearly all fishermen highlighted that the price of fuel and gear has increased disproportionately compared to the sale price of scallops over the last 10 years. Fuel price increases have a much greater negative economic impact on vessels with mobile gear than those with static gear (Abernethy *et al.*, 2010).

Legislation has restricted both the inshore and offshore fleets due to area closures and effort restrictions, respectively. Area closures such as inshore potting zones and marine protected areas (MPA's) have prevented access to traditional fishing grounds in inshore areas. Ground closures displace the impacts of fishing to other locations (Greenstreet *et al*., 2009), with financial and socio-economic impacts on fishers. Therefore, when proposing sites to meet conservation objectives careful consideration should be made of the potential impacts on fleet behaviour. In particular, fishermen cited a number of ground closures that have impacted their activities in recent years, including the special area of conservation (SAC) in Lyme Bay, Marine Conservation Zones (MCZs) in Falmouth Bay and recent closures around the Isle of Wight. Existing and proposed Marine Conservation zones in the English Channel are shown in Figure 2.8. A productive scallop fishery around the Isle of Wight was recently closed to towed bottom fishing gears on a precautionary basis due to the occurrence of natural features (such as reefs) that are protected under the EC Habitats Directive. Vessels that operate in this region are <12 m in length and have historically exploited a range of species. Recent declines in the clam and cockle fisheries and the collapse of the oyster fishery mean that local fishers are increasingly reliant on scallop fishing (Patrick Cooper, Southern IFCA, pers. comm.). The closure of productive scallop grounds on a precautionary basis therefore has large ramifications for local vessels in times of ever diminishing fishing opportunities. Currently, the total area of designated MCZs and Special Areas of Conservation (SACs) in the English Channel is over 6000 km^2 (data from http://jncc.defra.gov.uk/). This represents 3 % of the area impacted by the inshore fishery (calculated from LK data). However, due to the overestimation of the area impacted, this proportion is likely to be higher in reality. A further 3400 km^2 of areas have been proposed as MCZs for the next round of implementations under the Marine and Coastal Access Act 2009 (Hill *et al.* 2010). The public consultation regarding these closed in April 2015, with a decision due in early 2016.

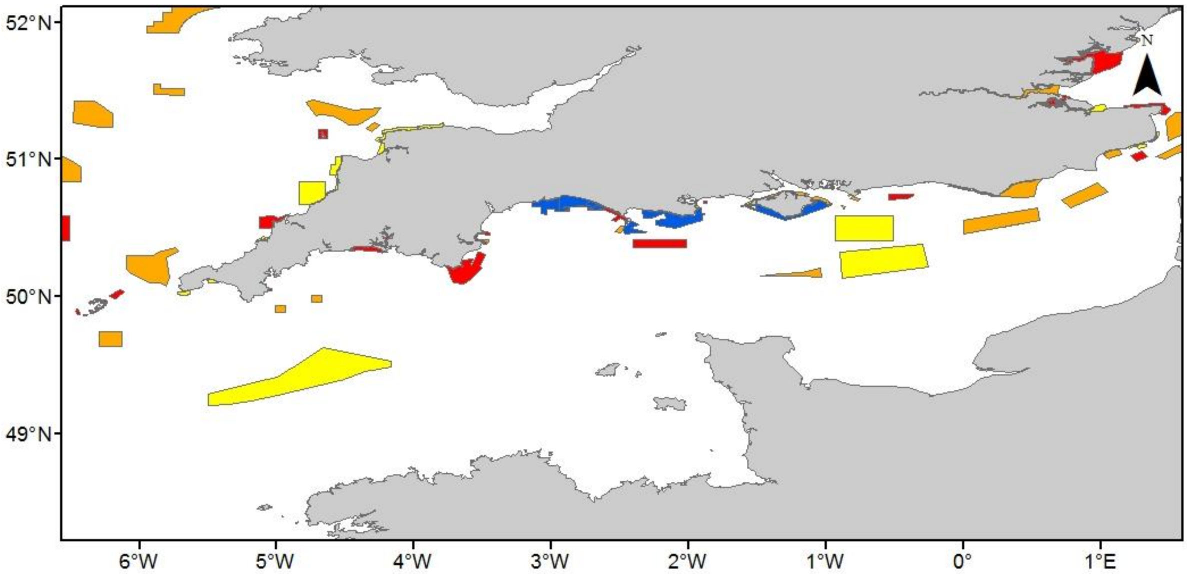

Figure 2.8: Location of designated (red), recommended (orange) and proposed (yellow) Marine Conservations Zones (MCZs) in the English Channel. The Lyme Bay Special Area of Conservation (SAC) and other areas closed to bottom-towed fishing gear are shown in blue. Data from http://jncc.defra.gov.uk/.

The introduction of a 12 hour curfew has also affected inshore vessels in the Cornwall; Devon & Severn and Southern IFCA (Inshore Fisheries & Conservation Authorities) districts. Many harbours in this region are tidal, meaning that vessels can only exit and enter a few hours either side of the high tide. Due to tidal cycles this period often does not align with the time of the curfew. Despite this, a number of fishermen said that they supported the curfew as a measure to restrict fishing effort in the region. Traditionally inshore fishermen complete day trips of up to 12 hours, returning to port to land the catch each evening. However, interviews revealed a change in fleet behaviour in recent years with an increased number of smaller boats (<15 m LOA) completing trips of up to 3 days. This involves anchoring at sea overnight, to avoid the costs of fuel and berthing fees associated with travelling back to port, or by moving outside the 6 NM limit to continue fishing during curfew hours. Some of the smaller vessels lack basic facilities such as a flushing toilet or shower, therefore conditions for the crew during trips of >1 day are not ideal. Some skippers did not like having to spend longer periods away from home but felt they had to do so in order to maintain a profitable business. A third of inshore fishermen reported having to travel further and complete longer trips, due to the closure of nearby traditional fishing grounds. This demonstrates the negative social and economic impacts that legislation can have on the inshore fleet. In contrast, the trip length for offshore vessels varied between 3-7 days and this has not altered in the last 10 years.

Formulating future management

Although the majority of fishermen felt that the fishery was currently exploited at a sustainable level, fishers recognised the risks from increasing numbers of vessels exploiting the offshore fishery and displacement of effort, which has the effect of concentrating effort in certain areas. The latter is a key factor for consideration by fisheries managers when designing management plans (Greenstreet *et al.,* 2009). There was considerable variation in views of both inshore and offshore fishermen when asked whether total catch levels had changed over time. This may be due to the wide range of reasons cited that ranged from improvements in experience or technology to the number of vessels (fewer inshore vessels; more offshore vessels) and changes in scallop abundance. Although there were differences in opinions on what the most effective management measures are, the overall difference in response between fleet sectors was not significant.

Examples of where industry engagement has led to the development of successful management plans in scallop fisheries include the Shetland Islands (Goodlad, 2005), the north-western Atlantic sea scallop fishery (Murwaski *et al.,* 2000), the Isle of Man (Dignan *et al*., 2014) and the Baie St Brieuc (OECD, 2012). In general fishers are not opposed to regulations and often support the view that improved management is necessary (Yates *et al.,* 2014). However, for management to be both effective and complied with, it is essential that the industry understand the benefits of measures that are imposed. This can be challenging, not least as fishers from different sectors have different requirements meaning a united view on management across the industry can be difficult to achieve. This study demonstrated that individual fisher opinion can vary widely both within and between size classes of vessel (see table 2.5, 2.6).

Another issue is the perception of fishers that management is imposed without prior or fair consultation with industry, which leads to frustration and serves to divide fishers and scientists or policy makers (Yates *et al.*, 2014). In the present study both inshore and offshore fishermen agreed that temporary ground closures and a cap on the total number of dredges permitted are two of the most effective ways of ensuring a sustainable scallop fishery. The concept of permanent closed areas as a management tool was unpopular with all but one fisher in the present study. Differences in opinion also occurred between the inshore and offshore fleets. Inshore fishermen more frequently expressed that vessel and engine size limitations were important for sustainable management of the fishery. Inshore grounds are spatially limited and therefore to maintain stock levels it is logical that such restrictions are used. However, this view was not upheld by the majority of offshore fishermen and these factors are less relevant in the offshore environment as larger vessels are necessary to cope with the physical conditions encountered.

Conclusions

Views from across the inshore and offshore sectors indicate that many UK scallop fishers support aspects of the current management measures for the king scallop fishery in the English Channel. However, there are disparate opinions both within and between skippers in the inshore and offshore sectors. The process of engagement with fishers provides an extensive resource of information to scientists. Engagement with the industry can greatly improve the success of management schemes and stakeholder input should be an integral part of the governance process. Various environmental and socio-economic factors influence scallop fishing activity. The inshore scallop fleet fish on traditional grounds and are impacted considerably by ground closures, including existing MCZs and SACs. In comparison to this, the offshore fleet have large areas of productive ground available to fish, however economic drivers have reduced the spatial extent of the offshore fishery in recent years.

The LK data in the present study has certain limitations, with <100 % fleet coverage and a trade-off between scale and accuracy. However, it provides a tangible alternative in data poor situations and is proven to be accurate enough to be of use for king scallop fisheries (Shepperson *et al.*, 2014). However, for management decisions that require more precise estimates of fishing effort, sampling the entire fleet is desirable (Shepperson *et al.*, 2014).

Insight gained from fishers can be incorporated into future management plans to meet the objectives of developing effective management and an economically and environmentally sustainable scallop fishery. The present study represents a useful resource for fisheries managers, highlighting important fishing grounds for the fleet. Such data can be overlaid with habitat and stock information to enable evaluation of potential benefits and conflicts of future management measures. This is particularly pertinent in the process of extending the UK network of MCZ's.

CHAPTER 3: SCALING OF AGGREGATED VESSEL MONITORING SYSTEM DATA FOR IMPACT ASSESSMENT OF SCALLOP DREDGING

Abstract

Understanding and quantifying the impacts of bottom-towed fishing gears on seabed habitats is an essential part of informing management and improving the sustainability of fisheries. In EU waters, commercial vessels >12 m Length Overall (LOA) have been required by law to carry working vessel monitoring systems (VMS) since 2013 and vessels >15 m LOA since 2005. VMS data can be combined with species and habitat data to aid in the investigation of the relative impacts of fishing on the seabed. VMS data are subject to strict confidentiality legislation and as a result are rarely available in raw format. Instead the data are often anonymised and aggregated, before being released to scientists. The spatial scale at which VMS data are aggregated can have a large effect when using the data to estimate fishery impacts. The latter issue coupled with VMS transmission frequency (*c*. 2 hours) leads to potential errors when calculating the absolute impact and spatial footprint of a fishery. Aggregating VMS data at larger spatial scales tends to over-estimate the extent of the fishery in terms of seabed impacts; whereas aggregation at smaller spatial scales can underestimate relative effort. VMS data for the king scallop dredge fishery in the English Channel were aggregated at two spatial scales (0.025º and 0.05º grid cells), and the correlation in estimated fishing activity between years at selected sites was investigated. In the present study, aggregation of VMS data at the smaller scale provided greater spatial correlation in relative fishing effort between years. Spatial correlation in fishing activity between some years but not others suggests that effort at certain sites can be consistent year on year, while others are fished sporadically on a timescale of >1 year. Limitations of the methods are discussed along with the application of measures of relative fishing intensity to studies on the habitat impacts of the scallop dredge fishery.

3.1 Introduction

The ability to accurately quantify the spatial and temporal distribution of fishing activity is an important pre-requisite to ascertaining the environmental footprint of fishing activities. Fishing effort is one of a suite of indicators that can be used to ascertain impacts on 'sea-floor integrity'; a descriptor that is used to assess the health of marine ecosystems under the European Marine Strategy Framework Directive (MSFD, CEC 2008). Recent advances in the use of tracking technology, such as vessel monitoring system (VMS) data, have provided novel insights into fleet behaviour and dynamics (Rijnsdorp *et al.*, 1998; Poos & Rijnsdorp, 2007). VMS was designed for fisheries control and enforcement; however its use for science is an additional benefit. VMS data can be used to estimate the area swept, or frequency of disturbance from towed fishing gears. It can also be used to identify spatial aggregations of fishing activity; however there can be high spatial variation in these aggregations between years (Vanstaen, 2008). In the UK, the > 15 m LOA scallop fleet is nomadic and the spatial and temporal distribution of fishing activity reflects weather conditions, variability in scallop abundance and recruitment, displacement of fishing through management measures such as the Western Waters effort regime (Council Regulation (EC) No 1415/2004) and economic trade-offs or personal habits (Bastardie *et al.*, 2010; Tidd *et al.*, 2012). The distribution of fishing effort can be variable at localised (2x2 km) spatial scales (Vanstaen, 2008). Similarly, patchiness in beam trawling activity in the North Sea occurs at a scale of 9 km^2 scale (Rijnsdorp *et al.*, 1998).

Various measures of fishing intensity have been used in the context of understanding the potential environmental impact of towed bottom fishing gears. These methods include estimation of: time spent fishing in a specified area (Lee *et al.*, 2010); the number of times a specified area has been swept per year (Rijnsdorp *et al.*, 1998; Lambert *et al.*, 2011); the area of seabed impacted using observations aggregated over one year (Diesing *et al.*, 2013). Individual VMS tracks are rarely available to scientists due to restrictions imposed by confidentiality (Hinz *et al.*, 2012). As a result, in the UK, VMS data are often provided to scientists in an aggregated and anonymous format. This necessitates the aggregation of the data over a given time period into grid cells of a specified size, using the point summation method in order to estimate relative fishing effort. This prevents identification of individual vessel activity. Use of the point summation method has the following assumptions: the entire cell has been impacted by fishing; fishing effort is constant across the area of each cell. These assumptions inevitably mean that the larger the grid cells used, the greater the over-estimation of the area of seabed impacted (Piet and Quirjins, 2009; Gerritsen *et al.*, 2013). However, as the frequency of VMS transmissions is generally every 2 hours, using a smaller grid cell can produce values of zero impact for cells in which no VMS transmission occurred as the vessel passed across an area of seabed. Hence, there is a trade-off between resolution and accuracy, which has implications when using the data to investigate the impacts of fishing activity on species and habitats (Gerritsen *et al.*, 2013). For locations that are least impacted, the predicted level of impact increases with increasing resolution of data, whereas predicted impact at more frequently fished areas is curtailed when resolution is increased. Lambert *et al.* (2012) found that using a 3 NM^2 aggregated resolution of VMS data, estimates of fishing effort were 17 % greater than when using a scale of 1.5 NM^2. The finer-scale data gave a much greater negative correlation with the biomass of sedentary organisms (such as anemones) that are vulnerable to fishing disturbance and fishing effort. This demonstrates that the spatial scale at which VMS data are aggregated has important implications when trying to understand the effects of fishing disturbance on biological communities and habitats. Similarly, Piet and Quirjins (2009) demonstrated that using a spatial scale of 3.43 km^2 indicated that 66 % of a given area was fished, whereas a scale of 0.003 km^2 indicated that 14 % of the same given area was fished. Smaller spatial scales are more appropriate when considering fisheries for sedentary target species, whereas aggregation of VMS data at larger spatial scales might be more appropriate when considering mobile or migratory target species (Piet and Quirjins, 2009). Some studies have been able to estimate the total area of the seabed swept by fishing gear (using the width of the fishing gear or engine size combined with estimated fishing time, e.g. Rijnsdorp *et al.*, 1998; Murray *et al.*, 2013). However, such estimates are not possible with anonymised data as they rely on knowledge of vessel characteristics such as engine size or gear width. If raw VMS data are available, vessel tracks can be interpolated using different methods such as straight lines (Eastwood *et al.*, 2007); or cubic Hermite splines (Hintzen *et al.*, 2010). However, due to the nature of vessel movements that are often non-linear, interpolated tracks can deviate >3 km from the actual track (Skaar *et al.*, 2011, Lambert *et al.*, 2013).

The time period over which VMS data are aggregated is also relevant when trying to estimate dredge impacts. Recovery of sand and gravel seabed habitats may occur at time-scales of 2-8 years (Tillin *et al.*, 2006, Kaiser *et al.*, 2006; Hiddink *et al.*, 2007; Lambert *et al.* 2014a) and subsequent recovery may mask the effects of the dredge fishery. However, cumulative impacts of fishing will occur over longer timescales. It is difficult to infer the last time a site was fished unless raw VMS data are available and all vessels in a fleet are monitored (Lambert *et al.,* 2012). The ability to detect ecosystem changes that are attributed to fishing activity relies on our ability to accurately determine fishing pressure at an appropriate scale, and distinguish those effects from the natural variability in an ecosystem. The aim of this study was to investigate annual variation in the spatial distribution of scallop dredging activity and estimates of fishing intensity. This was calculated using anonymised VMS data aggregated at two different spatial scales, to assess the implications of using data aggregated at different scales. Inter- and intra-annual correlation in fishing activity at selected sites was used to assess spatial and temporal patterns of fishing activity. We hypothesise that if there is 100 % correlation in activity at sites, between years then it can be assumed that fishermen fish the same grounds year on year. If there is zero correlation, the footprint of fishing activity differs completely between years. An intermediate level of correlation would suggest that certain areas of the seabed experience fishing more consistently each year, whereas other patches are depleted sporadically with the fleet returning to fish these patches at a frequency >1 year. Under this hypothesis the inter-annual correlation may increase with time. High correlations over short timescales (<3 years) would suggest that the same areas are revisited for a couple of years, after which the spatial pattern of fishing may evolve. Knowledge of spatial and temporal patterns in fishing activity are necessary when using fishing effort data to investigate the impact of towed-bottom fishing gear on the seabed.

3.2 Methods

Since 2005, VMS (Vessel Monitoring System) data has been recorded for all commercial fishing vessels >15 m LOA fishing in EU waters. In the UK, VMS data are managed by the Marine Management Organisation (MMO). In January 2013 legislation was updated such that all vessels >12 m LOA were monitored using VMS. For this reason, until 2013 the VMS data represents mainly offshore (> 6 NM (nautical miles) from land), rather than inshore fishing activity. Records of vessel position and speed are obtained approximately every 2 hours when vessels are at sea. The periods of time when vessels are not engaging in fishing activity can be identified by removing VMS records that fall outside the thresholds of normal fishing speeds for the specific gear types (Lee *et al.,* 2010). Raw VMS data were not available due to confidentiality restrictions. This meant that it was not possible to attempt the reconstruction of individual vessel tracks or estimate the total area swept as in other studies (e.g. Piet &

Hintzen, 2012). Anonymised VMS data for dredge fishing gears (classified as either boat dredges or mechanised dredges) for UK and non-UK vessels, for the period October 2005 to September 2013 (inclusive) and covering the English Channel (ICES sub-areas VIId and VIIe), were obtained from the MMO. Data provided for each VMS record included: month; year; latitude; longitude; IFISH (UK Sea Fisheries Data Warehouse) gear code; time interval between successive records; and whether the vessel was of UK or non-UK origin. After data exploration, records classified as 'unknown' gear type were omitted from the analysis due to the potential for over-estimation of fishing effort by gear type. All 'dredge' gear codes were assumed to be scallop dredgers as there is unlikely to be activity from any other type of dredgers (oyster, mussel, cockle) at locations of high scallop fishing intensity in the English Channel (E. Bell, CEFAS, pers. comm.). Previous studies have shown that grid cells of between 0.06 x 0.03° and 0.6 x 0.3° are appropriate for the analysis of VMS data (Piet & Hintzen, 2012). Data were aggregated at resolutions of 0.025 and 0.05 decimal degree grid cells, equivalent to approximately 5.0 km^2 and 20 km^2 respectively. Scallop vessels tend to fish at speeds of >2 knots (nautical miles per hour) and <3.5 knots (Lee *et al.*, 2010; Lambert *et al.*, 2012). During transmission, the speed at which the vessel is travelling at the time of each individual VMS record is recorded as an integer and rounded to the nearest knot (Janette Lee, CEFAS, pers. comm.). Therefore VMS records with speeds of 2, 3 and 4 knots were retained for analysis. A VMS record is supposedly transmitted approximately every two hours when vessels are at sea. However, during data exploration, the transmission poll rate was found to be highly variable. The mean transmission interval (±S.D.) was 2.92 ± 1.98 hours (0.05° cells) and 2.65 ± 1.9 hours (0.025° cells). The observed variation in VMS transmission interval could be due to the transmission rate increasing on occasion for vessel monitoring purposes, or through the occurrence of duplicates not captured during validation checks. Intervals >2 hours can result from failure of the VMS signal or equipment. As the time required to travel back to port from the fishing grounds, refuel, and unload the catch is considered to be >8 hours, this threshold was used and successive records with speeds of <2 or >4 knots and time intervals of >8 hours were omitted from the dataset (this represented 32 % of records). Duplicate records were also removed. Due to the observed variation in transmission interval, summation of the number of VMS records in each grid cell would not be an accurate measure of relative fishing intensity. Therefore, the sum of the time interval between successive records, by grid cell, provided an estimate of the total number of hours fishing activity within each grid cell. These values were used as a measure of relative fishing intensity (referred to as 'FI' from here on in). The VMS data were imported into ArcMap v.10 and the 'Point to Raster' tool used to create continuous rasters with cell sizes of 0.025 and 0.05 decimal degrees. Forty sample sites, situated across eight locations of known scallop fishing activity in the English Channel were identified (Figure 3.1). An estimate of FI was extracted from the grid cell in which each site fell, for each of the 12 month time periods.

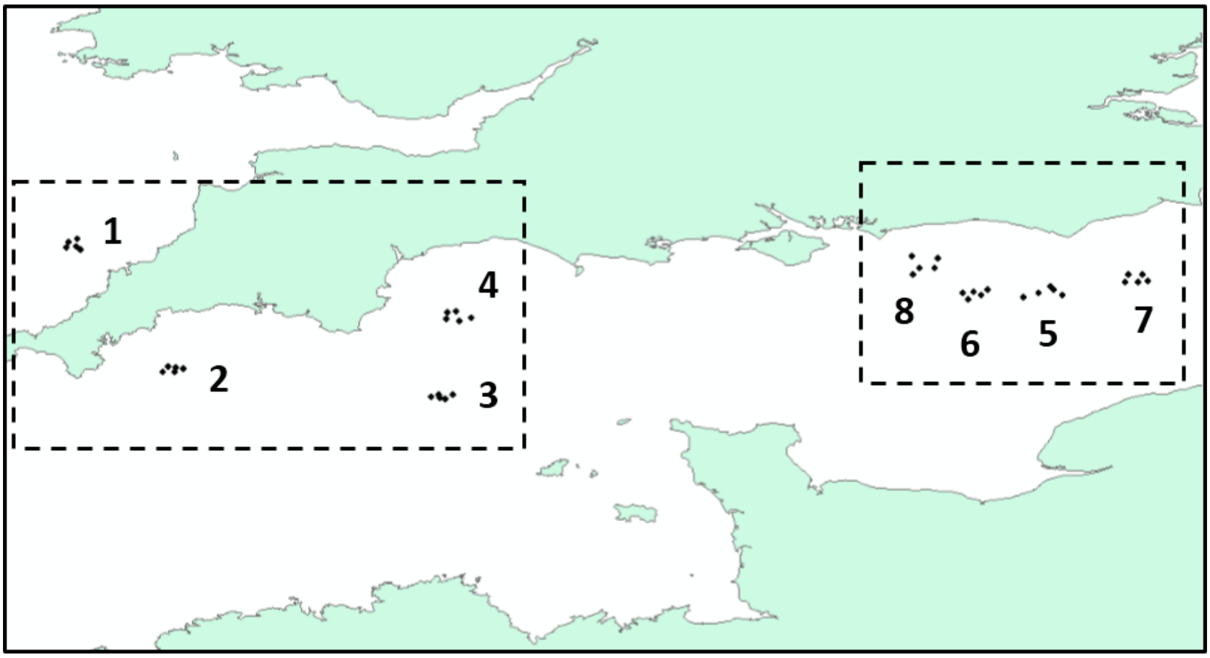

Figure 3.1: Location of the eight sample areas (numbered 1-8) and the forty sample sites (identified by black dots), in the English Channel. Sites were numbered numerically, e.g. 1.1; 1.2; 1.3; 1.4; 1.5; however individual sites are not labelled on the figure.

3.3 Data Analysis

Spatial and temporal correlation

Initially, the ranking of sites in terms of relative FI at the two scales of data aggregation was assessed. The distribution of FI values across sites, calculated at the 0.025° scale was skewed with many low, and few high values. Significant correlations derived from such skewed data will therefore be driven by a few key sites which consistently see very high effort levels. The utilisation of lower effort areas might be variable between years, but this effect would be masked by the higher intensity areas. To compensate for this the FI values for the 0.025° scale were log-transformed, thus giving the lower intensity areas greater weighting. Each pairwise combination of 12 month time periods, at both spatial scales (0.025° and 0.05°), was tested using Pearson's correlation coefficient to assess the spatial correlation in FI at each site, between years. Regression plots of the fishing intensity at each sample site by year were used to visualise the change in FI over time at individual sites (e.g. an increase, decrease or consistent pattern). Paired t-tests were used to investigate significant differences in mean FI at sites over the eight year period, at both spatial scales. The mean of the correlation coefficients for each year against all other years was calculated to indicate in which years the spatial pattern of fishing deviated most compared to other years. A bootstrapping analysis, using a random sample of correlation values was performed to assess whether the time lag between years had an impact on the mean correlation between years.

Total area impacted

The proportion of the English Channel impacted by scallop dredging was calculated, to provide an insight into the total spatial footprint of the fishery (but not taking into account the variation in the level of impact in different areas). The percentage of grid cells impacted by scallop fishing gear during the full eight year period (January 2005 to September 2013) within ICES sub-area VIId and VIIe offshore areas (> 6 NM from the coast) was estimated. First, a buffer was applied to the data to remove any records within 6 NM of the coastline, as vessels >15 m LOA are not allowed to fish within this boundary (due to restrictions on vessel size or the total number of dredges permitted within this zone). Any VMS records of between 2 and 4 knots within the 6 NM zone are therefore likely to be due to the vessel slowing for navigational reasons, rather than fishing. The percentage of grid cells impacted by increasing levels of fishing intensity (0-10, 10.1-20, 20.1-40, 40.1-80, >80 hours) was calculated from a continuous raster created from the 0.025° data set using a 9 km^2 raster cell size. A Kolmogorov-Smirnov test was used to assess the similarity in distribution of the degree of impact (FI) and total area impacted between the most recent 3 and 8 years of fishing activity.

3.4 Results

Spatial and temporal correlation

The mean difference in FI over the 8 year period and between the two scales (0.025° and 0.05°) was significant at 7 of the 40 sites (Table 3.1). For those sites the mean difference in total FI between years ranged from 3.01 to 13.27 hours. The rank order of sites based on FI also varied between the two spatial scales. The difference in FI between data aggregated at the two scales at any one site in a single year ranged from 0-30 hours. There was no difference in FI at individual sites, between the two scales, for 20 of the 320 site/year combinations (6 %). At the larger scale of aggregation (0.05°), there were 15 occurrences of zero activity at all sites over the eight year period, however at the smaller scale (0.025°) there were 45 occurrences of zero activity over the same time period. At the smaller scale each site had been impacted by scallop dredges (>0 hours fishing) at least once in the last three years. At the larger scale this increased to at least once in the last two years. There were a greater number of significant correlations in FI between years at the smaller scale (0.025º grid cells). Out of the 28 possible pairwise combinations, 20 correlations were significant (r^2 0.31-0.60, $p<0.05$) for 0.025º grid cells (Figure 3.2) and 8 were significant (r^2 0.36-0.49, $p<0.05$) for 0.05º grid cells (Figure 3.3). For the 0.025º grid cells there was significant correlation between FI in 2012-13 (the most recent year of data) and all previous 12 month periods (r^2 = 0.35-0.63) except for 2009-10. For the 0.05º degree grid cells, 2012-13 FI significantly correlated with the preceding year, as well as three other non-consecutive years.

Table 3.1: Mean differences in relative fishing intensity (FI) over eight 12 month periods, at sites in the English Channel, between data aggregated using 0.025º and 0.05º gird cells. Comparisons were made using paired t-tests. Only samples sites where significant differences occurred are listed.

Site	**t**	**d.f.**	**p**	**mean difference in FI calculated using 0.025° and 0.05° grid cells (hrs)**
3.1	2.851	7	0.025	3.01
3.5	2.936	7	0.022	5.93
4.4	4.155	7	0.004	13.27
5.4	2.366	7	0.050	6.12
8.1	3.664	7	0.008	10.75
8.4	4.046	7	0.005	9.11
8.5	3.170	7	0.016	3.98

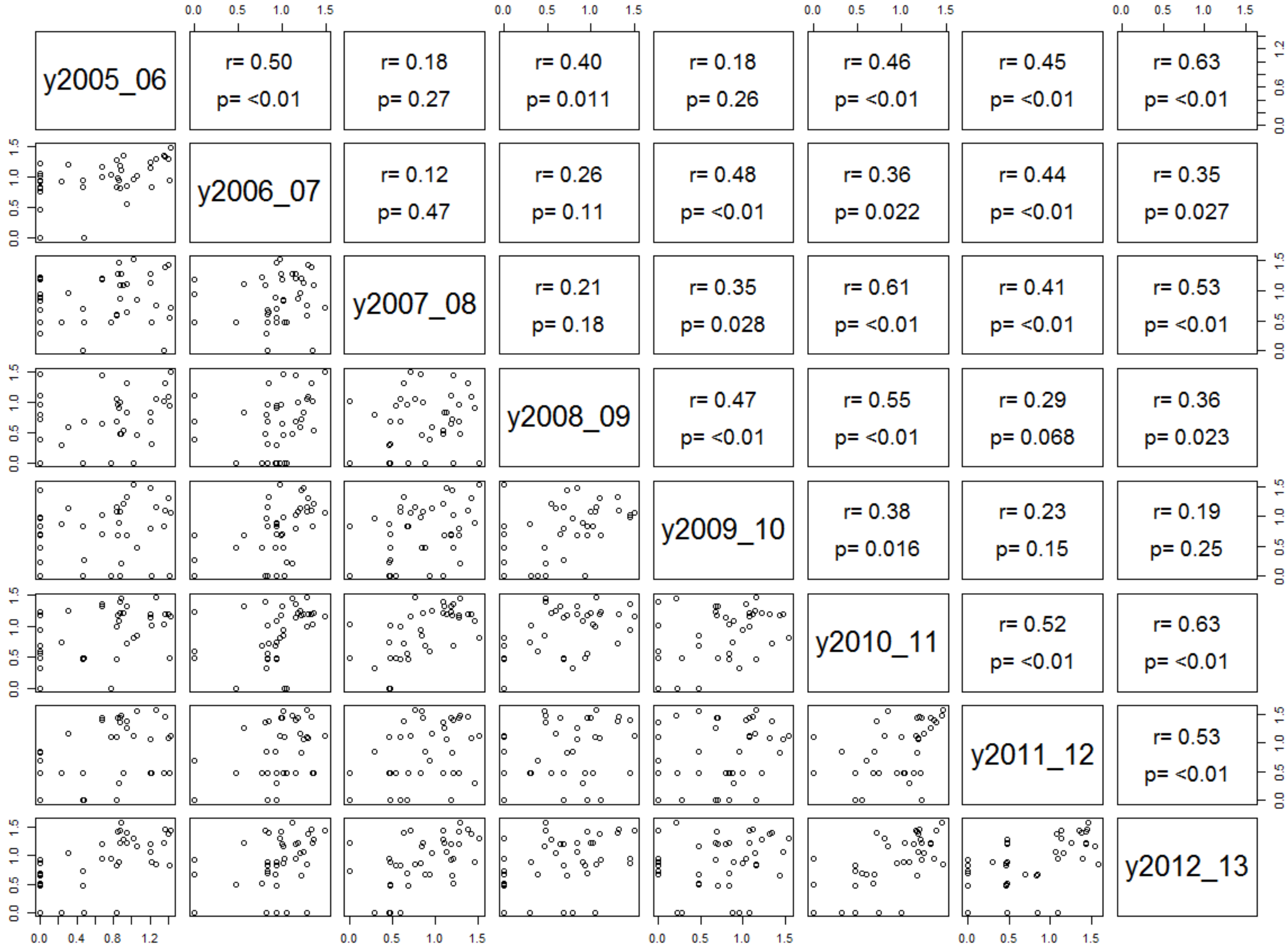

Figure 3.2: Scatter plots with Pearson's correlation coefficients and p-values for pairwise comparisons between each 12 month period of fishing activity, aggregated using 0.025° grid cells, using log-transformed FI data.

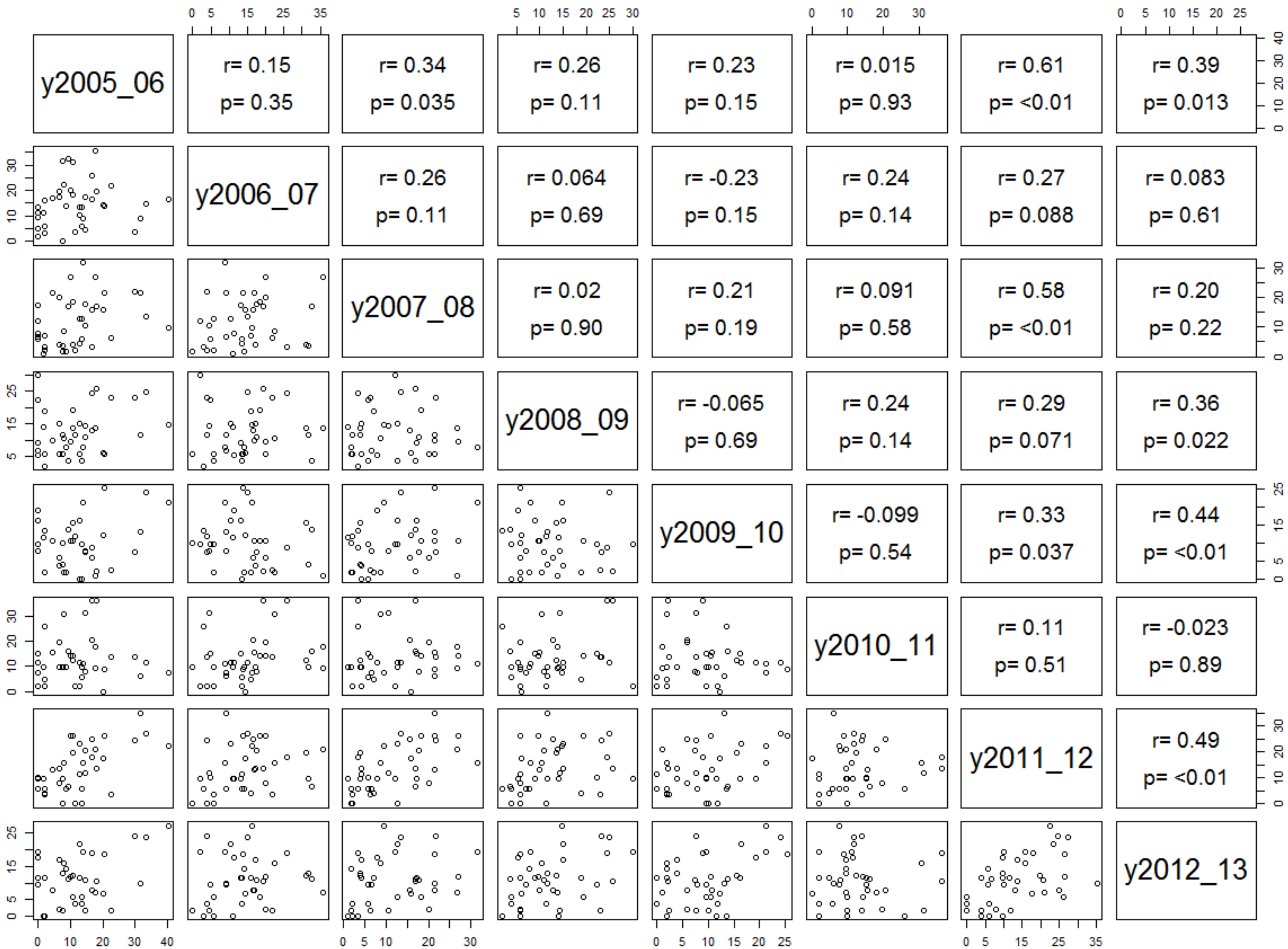

Figure 3.3: Scatter plots with Pearson's correlation coefficients and p-values for pairwise comparisons between each 12 month period of fishing activity, aggregated using 0.05º grid cells.

For data aggregated at the smaller scale (0.025º), the period 2009-10 had a significant correlation with 2006-07 and 2007-08, 2008-09 and 2010-11, suggesting that a more spatially consistent pattern of fishing occurred during that time period (2006-2011). The number of significant correlations of each year compared to every other year ranged from 4, to the maximum of 7 (activity in 2010-11 correlated with all other years). When comparing the correlation coefficient of 2012-13 with other years, the mean correlation was higher for the 0.025º grid cells (0.46 ±0.06) than for the 0.05º grid cells (0.28 ±0.07) (Figure 3.4). However, the mean correlation coefficients were not significantly different (t=1.168, d.f.=6, p=0.157). The spatial pattern of fishing in 2007-08, 2008-09 and 2009-10 is on average, less similar to all the other years, indicated by lower overall mean correlation coefficients for these years (Figure 3.5). A bootstrapping analysis, using a random sample of correlation values, revealed a similar pattern of correlation values (compared to using the whole dataset) between all years except for 2006-07 and 2010-11.

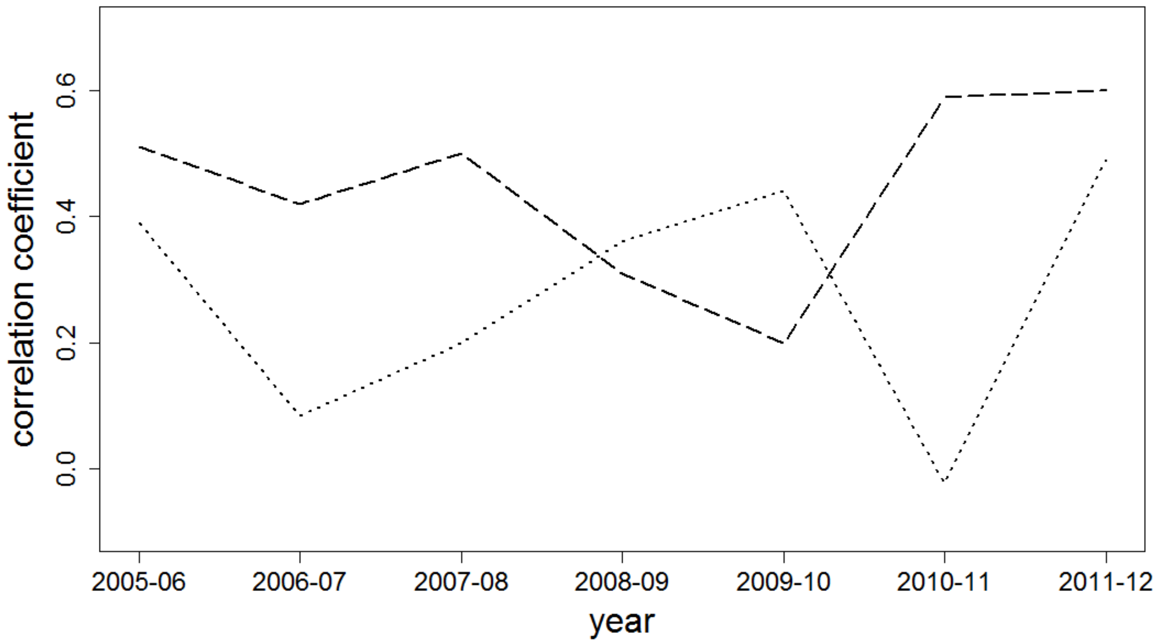

Figure 3.4: Plot of the correlation coefficient of the total hours scallop dredging activity at each site for 2012-13 against all other 12 month periods. Dashed line represents data aggregated into 0.025˚ grid cells; dotted line represents data aggregated into 0.05˚ grid cells.

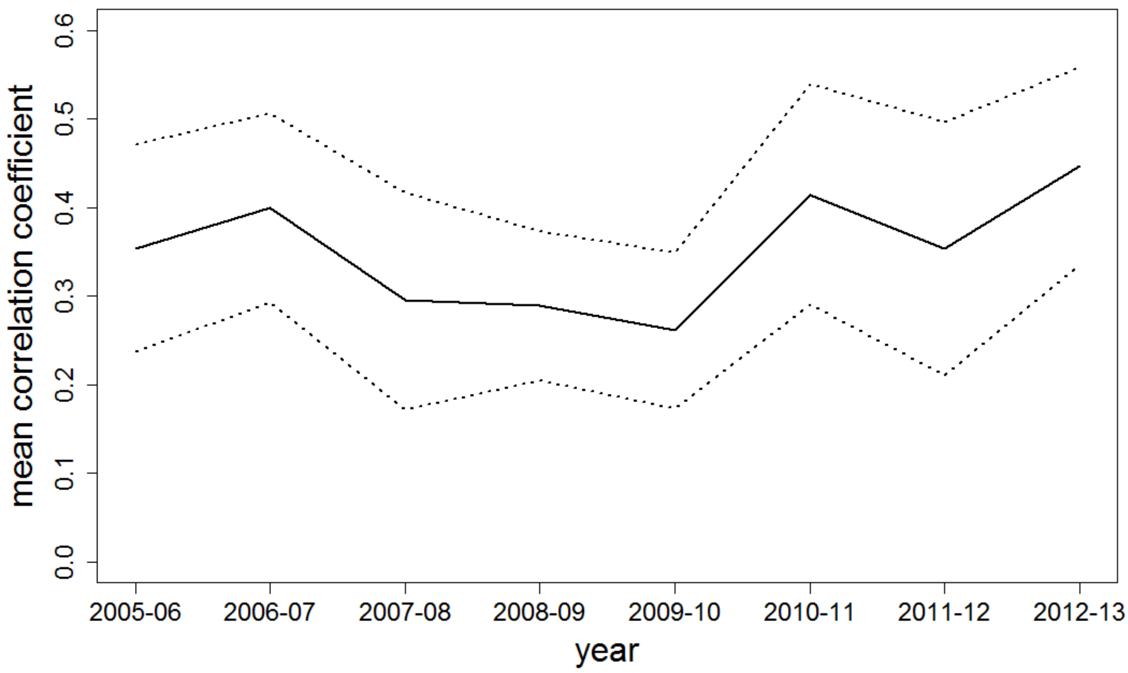

Figure 3.5: The mean of the correlation coefficients of fishing intensity for each year against all other years (solid line). Upper and lower confidence intervals representing 2 standard errors of the mean are shown (black dotted lines).

Total area impacted

The total area of grid cells for which no fishing activity was recorded between 2010 and 2013 covered 50 % of ICES sub-areas VIId and VIIe, outside of the 6 NM limit. Less than 10 % of ICES sub-areas VIId and VIIe experienced >40 hours of fishing during that period (Figure 3.6a). The proportion of seabed fished and not fished remained consistent between recent (3 year) and longer (8 year) periods (Figure 3.6a, 3.6b). There was no significant difference in the distribution of the degree of impact and total area impacted between recent fishing intensity over a 3 or 8 year period (Kolmogorov-Smirnov test: D=0.333, p=0.893).

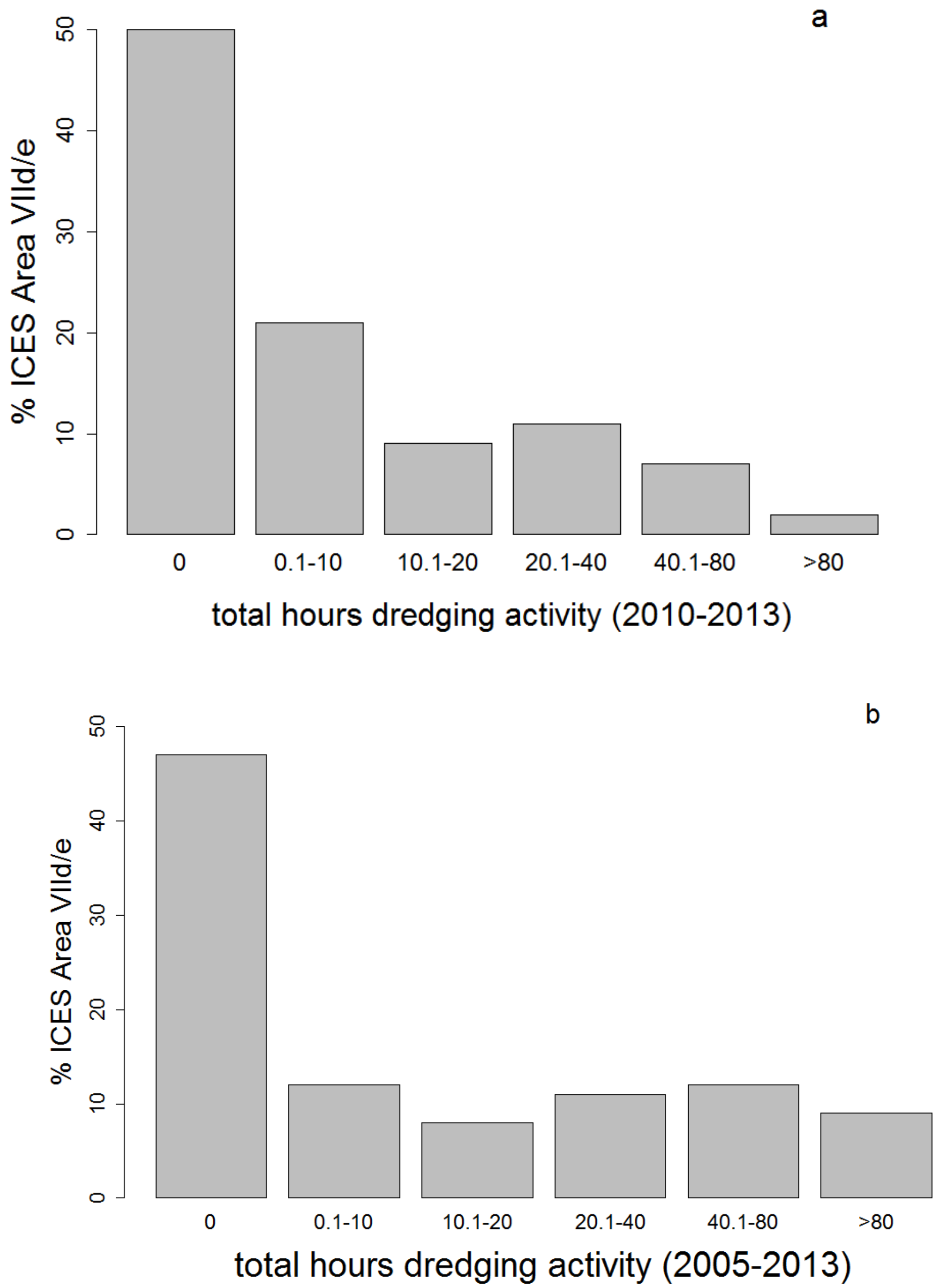

Figure 3.6: Histogram of the total number of hours dredging activity recorded and the area impacted as a percentage of the total of ICES Areas VIID and VIIE (excluding the area within 6 NM of the coastline) for a) October 2010 to October 2013 and b) from October 2005 to October 2013.

3.5 Discussion

Spatial scale

Scallops occur in discrete aggregations on specific habitat types and therefore scallop dredging activity is inherently patchy. The use of aggregated VMS data at broad temporal and spatial scales reduces our ability to observe small-scale patchiness in fishing behaviour (Rijnsdorp *et al.*, 1998), however this is sometimes the best data available to scientists. In the present study, estimates of fishing intensity varied with the scale of data aggregation for 94 % of the site/year combinations that were investigated, with the actual difference in estimates of fishing activity ranging from 0-30 hours. Thus, the scale of aggregation of individual VMS data points, together with the associated over- or under- estimation of fishing intensity, will have a direct effect on the ability to differentiate the gradient of fishing intensity that has occurred between specific sites. Use of a smaller scale is expected to provide a more accurate picture of FI and in the present study a smaller scale elicited a higher number of significant correlations in FI at sites between years. Although, the methods used in the present study provide an estimate of relative fishing effort, due to the fact that individual vessel records and details (such as gear width) were not available, the total area of seabed impacted could not be determined.

In the present study, VMS data aggregated at a smaller scale (0.025˚ grid cells) resulted in much greater correlation in fishing intensity between years than aggregation at a larger scale (0.05˚ grid cells). This suggests that activity in the scallop fishery is patchy at small (5 km^2) scales. Due to the assumption that effort is equal across each grid cell (even though only a fraction of the cell may have been dredged), precision is reduced at larger cell sizes and the total area impacted is over-estimated. Hence, there were a greater number of zero hour FI values when the data was aggregated using 0.025˚ grid cells. This is due to the fact that a vessel may pass through a grid cell without a VMS transmission occurring. Scallop dredgers in the English Channel tend to fish at speeds of 2.0-3.5 knots, in a figure of eight pattern, turning halfway through the tow (based on interviews undertaken with 46 scallop vessel skippers). Tow duration typically ranges between 30-120 minutes and is influenced by seabed type and the rate at which the dredges fill up. At an average speed of 3 knots this equates to a total distance of just over 5.5 km in one hour or 11 km in two hours. As the vessel turns 180º halfway through the tow, it may pass through one to three 0.05º grid cells (approx. 3.5 x 5.5

km), or between three and six 0.025º grid cells (approx. 1.8 x 2.8 km). If the transmission interval is ≤ 2 hours then at least one of those cells (but perhaps not all) will have a record of fishing activity. The longer the interval between successive VMS records, the fewer cells will be associated with fishing activity; with this issue occurring more frequently at smaller scales. Therefore, there is also a risk of underestimating the total area impacted by the dredges. Aggregating records over time will reduce the effect of, but not remove, such limitations. This effect is reduced as the number of data points increases. As VMS data reflect random samples of fishing effort during a trip, the precision of the estimated effort within a grid cell is determined by the number of observations in each cell, rather than the size of the cell (Gerrtisen *et al.,* 2013). The greater the number of observations, the greater precision can be obtained with a finer scale of aggregation. For fisheries that are habitat specific and that target sessile species, the finer the resolution of the fishing intensity data used, the more accurate the estimate of ecosystem effects on different areas of the seabed. Using larger scales introduces large errors and doesn't capture the spatial and temporal dynamics of the fishery. This makes management in relation to the European Marine Strategy Framework Directive (MSFD) difficult. For this reason, and based on the results of the present study, 0.025º was considered a more appropriate scale than 0.05° for use in assessing the impact of the scallop dredge fishery on species and habitats.

Temporal scale

At the 0.025˚ scale, the highest number of significant correlations in fishing activity across all sites occurred between 2012 and 2013, and 6 of the 7 other years. The number of significant correlations between other years varied between 4 and 7 (activity in 2010-11 was significantly spatially correlated with all other years). This suggests that while some sites are repeatedly visited and fished, other sites are fished on a more sporadic basis, at timescales >1 year. Correlation coefficients ranged from 0.15 (non-significant) to 0.63. Therefore, just under 40 % of fishing each year occurs in different locations to the previous year. Experimental and comparative studies of scallop dredging on mixed sand and gravel habitats (that predominate in the English Channel) indicate that recovery occurs on time-scales of 2-8 years (Tillin *et al.,* 2006, Kaiser *et al.,* 2006; Hiddink *et al.,* 2007) and scallops spawn at least once per year. This suggests that the nature and habits of fishing patterns could serve to maintain the fished population (albeit through an altered habitat, see Figure 7) and sustain the fishery and stocks. In recent years rising fuel and gear costs, as well as restrictions in Western

Waters effort allocation for the >15 m scallop vessels have resulted in a reduced fishery footprint, resulting in a higher spatial correlation between years. Economic considerations drive fishers to target grounds closer to ports, to limit the costs (of time and money) associated with steaming to offshore fishing grounds where economic returns are uncertain. Historically, scallop fishing has only occurred in the eastern English Channel during winter months (October to March) when the roes are larger and the catch more valuable. However, there was an unusually large recruitment of scallops to the fishery in the eastern English Channel in 2009-10. During 2009-10 fishermen reported spending a full 12 months fishing in the eastern English Channel due to high yields from the fishery (unpublished data from interviews conducted with scallop fishermen). The most recent period of activity (2012-13) was significantly correlated with every year except for 2009-10. This pattern of behaviour suggests a redistribution of fishing activity after 2010 to a wider variety of locations, with less focus on the eastern English Channel, no doubt due to depletion of the stocks in this area following the intensive fishing activity during 2009-2010. Spatial variation in fishing activity between years could be due time lags between good year classes, or the industry instinctively reducing effort in particular areas for a few years to allow the stock to rebuild after fishing. Extending the analysis with data from subsequent years may reveal further temporal patterns. The years 2007-08, 2008-09 and 2009-10 showed low correlation with the other years in the current dataset; however if the pattern of fishing is cyclical, correlations with future years may be significant. Before 2005, VMS was only obligatory for vessels >18 m LOA; therefore investigating previous patterns is only possible for vessels above this size.

Displacement of fishing can lead to a more homogenous pattern of activity (Dinmore *et al.*, 2003). Drivers that influence the habits of fishers and hence the spatial variation in fishing activity have potential consequences for the intensity of fishing in specific locations and the length of recovery time between fishing events. French legislation prevents French scallop vessels catching scallops in ICES sub-area VIId between May and October each year. The English >15 m LOA scallop fleet agreed to a temporary closure to scallop dredging in VIId for the months of August and September in 2013 and 2014, in return for additional Western Waters effort (in terms of kW days) from the French. This agreement is likely to be repeated in future years. Although this serves to reduce the total annual fishing pressure on scallop stocks in VIId, the duration of closure is not long enough to allow recovery of benthic communities that occur within the fishing grounds (see below). Also, the resulting displacement of scallop fishing effort to other locations will have consequences for the associated benthic communities in those areas. At smaller scales, implementation of Marine Conservation Zones in UK waters (such as the recent designation of a traditional scallop fishing ground called the 'Manacles' in Cornwall as a Marine Conservation Zone, thereby prohibiting scallop dredging) will cause similar displacement of the fleet.

Recovery

The impacts of disturbance from fishing gears on benthic communities are likely to be less significant in habitats where natural disturbance is high, as organisms are adapted to physical disturbance. For example, no difference was found in scallop density and community composition between fished and unfished areas of seabed in Cardigan Bay, Wales where strong natural disturbance is present in the form of tidal and wave induced currents and storm events (Sciberras *et al*., 2013). Similarly recovery rates are negatively correlated with natural disturbance levels, e.g. the more dynamic the habitat, the faster the recovery time (Hiddink *et al.,* 2007; Lambert *et al.,* 2014a). The morphology and life-history traits of organisms influence their susceptibility and recoverability after fishing disturbance. Erect, fragile, large or soft organisms are more susceptible to damage from contact with the dredge. Slow-growing, long-lived species will take longer to recover than fast-growing, opportunistic species (Kaiser *et al.,* 2006; Tillen *et al.,* 2006; Hiddink *et al.,* 2007). Community composition, and subsequently, recovery time is explicitly dependant on seabed type, which is correlated with the amount of natural disturbance at the seabed. The patchiness of fishing therefore influences recovery rate; recolonisation from nearby undisturbed or less disturbed patches can speed recovery compared to large disturbed areas with minimal immigration of fauna from outside (Collie *et al.,* 2000; Lambert *et al.,* 2014a).

The number of times an area of seabed has been impacted by fishing is an important indicator of pressure. The regime of harvesting by scallop vessels is cyclical, whereby areas of the seabed are revisited and fished intermittently. The scale of aggregation of VMS data affects the estimation of time since the last impact; in the present study a much higher frequency of zero activity records occurred when data were aggregated at the smaller scale. The exact time since the last fishing disturbance will influence community composition at a site; this can only be reliably estimated from interpolation of individual vessel tracks for all towed bottom fishing gear within an area. Due to confidentiality restrictions on VMS data in Europe this is outside the scope of the current study. Results indicated that at least 35 of the 40 sites had been impacted by scallop dredging in the most recent 12 month period and all sites had been impacted in the last three years. This suggests that the time between fishing events is too short to allow substantial recovery of the habitat, following disturbance from the dredge. If more substantial recovery of the seabed was a desired outcome a rotational management regime, allowing adequate time for benthic communities to recover between fishing events, would be required. The fleet of commercial scallop vessels in the English Channel expanded significantly in the mid-1970's and has remained high ever since. Therefore, the sites used in this analysis have potentially been impacted for over 40 years. Hence, the 'natural' state of the ecosystem is likely to have shifted and 'recovery' following cessation of scallop dredging may be to a permanently altered state, represented by an altered community structure (Bradshaw *et al.*, 2001, Hiddink *et al.*, 2006a, Figure 3.7). Such communities may be dominated by species resilient to the effects of fishing. This makes it difficult to distinguish between the effects of recent dredging and longer-term changes that can occur on grounds that have been scallop-dredged for many years (Bradshaw *et al.*, 2002).

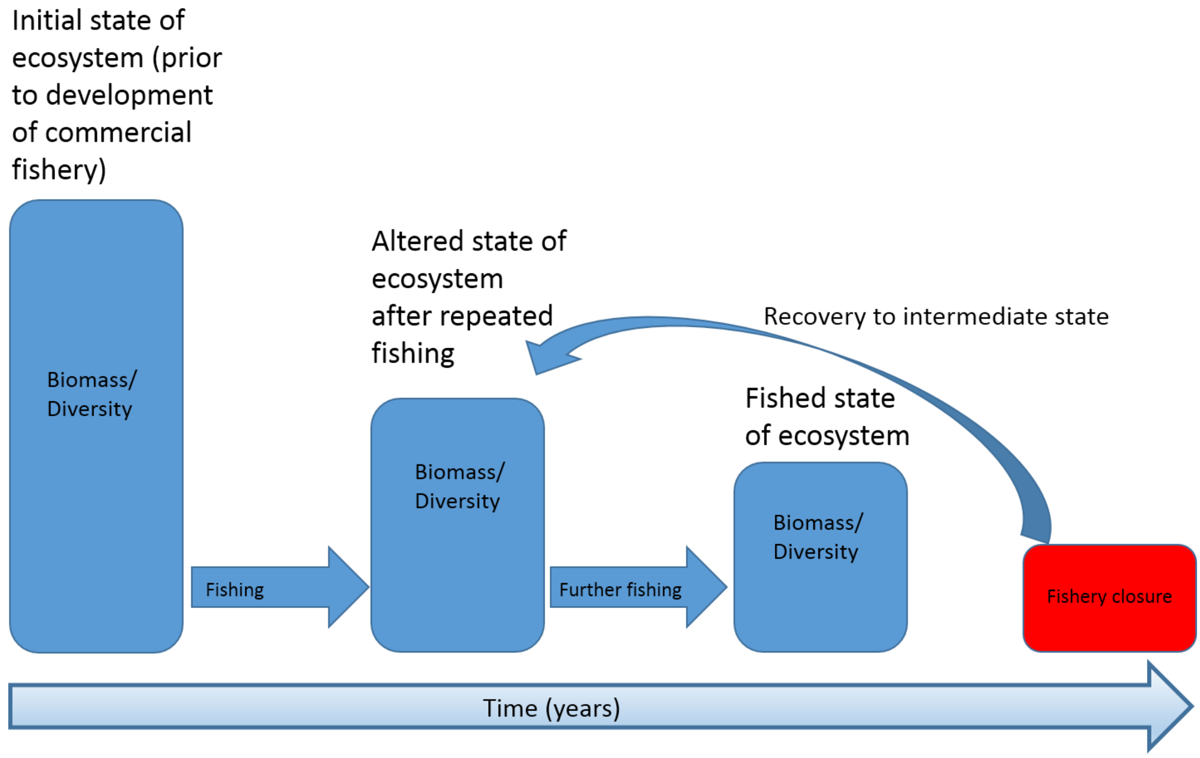

Figure 3.7: Conceptual diagram illustrating the initial and cumulative effects of commercial fishing on the biomass and diversity of an ecosystem and indicating how repetitive patterns of fishing could potentially hold the system in a permanently altered state.

Conclusions

Currently VMS records are transmitted at approximate 2 hour intervals (however this was found to vary greatly in reality). For the purposes of estimating total area of seabed impacted and the frequency of impacts from towed bottom fishing gears using aggregated VMS data, a polling frequency of <2 hours would improve accuracy. However, there is an associated cost with increasing poll rate and the optimal rate will vary depending on the species being targeted and behavioural patterns associated with the fishing fleet (Lambert *et al.*, 2012). As the vessel speed associated with each VMS record is rounded to the nearest knot it was necessary to include all records between 2 and 4 knots. This will over-estimate total fishing effort as it is unusual for scallop vessels to fish at speeds of 4 knots.

There was significant spatial correlation in dredge activity for the most recent 3 year period using data aggregated to 0.025º grid cells, suggesting that in recent years the location of scallop fishing activity in the English Channel has been fairly spatially consistent. This clustering of high correlations is unusual, albeit in the short time series analysed. This could be due to spatial and temporal restriction of fishing activity related to limits on days at sea available to vessels >15 m LOA and the economic trade-off of fishing closer to a landing port to minimise fuel costs. Estimates of relative fishing intensity can be applied to studies investigating changes in species diversity and composition that may occur between areas of high and low fishing intensity. To rank sites in order of relative FI for this purpose it is deemed appropriate to sum fishing effort over the most recent 3 years. In the present study scallop fishing activity was found to be relatively spatially consistent over this period of time. Based on predicted recovery time scales for the type of seabed in question, using data from a three year period would minimise the likelihood of habitat recovery occurring and masking the impacts of fishing. In the English Channel, our results show that 50 % of grid cells outside of the 6 NM limit have not been impacted by scallop dredges in the last 8 years, and a relatively small proportion of that area of seabed experiences the most intensive dredging activity. Thus, controlling scallop fishing effort by ensuring that a large proportion of the area remains unfished may be a more effective way of reducing the environmental impacts of the fishery compared to a rotational management regime.

CHAPTER 4: NATURAL VERSUS FISHING DISTURBANCE: DRIVERS OF COMMUNITY COMPOSITION ON TRADITIONAL SCALLOP FISHING GROUNDS

MSC data requirements addressed:

P2	Data requirements
Impact on Ecosystem	The fishery should not cause, or pose risk of, serious or irreversible harm to habitat structure or ecosystem function. There should be knowledge of the impacts of the fishery on the ecosystem and bycatch. The nature, distribution and vulnerability of all main habitat types in the fishery should be known at a level of detail relevant to the scale and intensity of the fishery.

This chapter has been accepted as a scientific paper to the ICES Journal of Marine Science and is currently in review:

Natural versus fishing disturbance: drivers of community composition on traditional king scallop, *Pecten maximus*, fishing grounds.

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Abstract

Scallop dredges are considered to be one of the most damaging forms of fishing to benthic habitats, although these effects vary among different habitats. The present study characterised the biological communities that occur within the spatial limits of the English Channel king scallop dredge fishery in relation to key environmental drivers (mean seabed temperature; seabed temperature range; interannual temperature variation; bed shear stress; substrate characteristics and depth) and across a gradient of scallop dredging intensity derived from vessel monitoring system data. Dredge fishing intensity was not correlated with species richness, species diversity or species composition. However, increasing tidal bed shear stress had a significant negative correlation with species richness and diversity. This outcome indicates that it is not possible to demonstrate that there is an effect of scallop fishing within the current spatial limits of the king scallop dredge fishery. This may be because historic dredge fishing could have altered the benthic communities within the area of the scallop fishery to those that are resilient to scallop dredging, or that fishing disturbance has no impact over and above natural physical disturbance within the fishery. An analysis of biological and life-history traits revealed that there was no relationship between recent fishing intensity, or bed shear stress, and the functional composition of the communities present. However, even the lowest bed shear stress values in the present study could be considered relatively high compared to areas outside the spatial boundaries of the fishery. Two distinct habitat groups were identified, based on the environmental drivers. These two groups were largely characterised by depth: deep (western) and shallow (eastern) sites. Species with traits that increase resilience to physical disturbance were abundant across all sample sites. Management concerning the environmental impacts of the fishery is discussed in terms of the spatial footprint of the fishery and predicted recovery timescales for the associated benthic communities.

4.1 Introduction

The environmental impacts of fishing gear on the seabed have been extensively studied over the last three decades, yet this remains a topical area of knowledge needs due to the increasing focus on managing the effects of marine fisheries on the wider marine environment (Borja *et al.* 2010). Towed bottom fishing gears have variable effects on species richness and biomass depending on the interaction between fishing gear, geographic location, sediment type and intensity of fishing (see Thrush & Dayton 2002). In general, impacts from towed bottom-fishing gears are lower in areas where natural disturbance is high (Kaiser *et al*. 2006; Collie *et al.* 2000; Lambert *et al*. 2011) but are long-lasting in biogenic habitats that are formed by organisms (e.g. Hall-Spencer *et al*. 2000; Cook *et al*. 2014).

Scallops are commercially important with an annual first sale value of more than £60 million in the UK alone (MMO 2012). Scallop dredging is known to lead to the mortality of benthic biota and causes disruption to benthic habitats, but the magnitude of these effects varies among different habitats (Kaiser *et al.* 2006). Biological communities that occur in naturally disturbed environments tend to be more resilient to disturbance and comprise a greater proportion of scavengers and mobile epifauna, compared to relatively undisturbed areas that have communities with greater abundances of sessile, emergent epifauna and burrowing infauna (Collie *et al.* 2000, Dernie *et al.* 2003; Sciberras *et al.* 2013). Collie *et al*. (1997) found greater species diversity and habitat complexity at sites where less intense scallop dredging occurred on Georges Bank in the north-western Atlantic, and Kaiser *et al.* (2000) found similar effects in the coarse sediment habitats around the Isle of Man. However, both Stokesbury & Harris (2006) and Sciberras *et al.* (2013) reported that the short-term effects of scallop dredge disturbance could not be differentiated from the effects of natural disturbance from physical processes (such as shear stress from strong currents, tides or storm events) in certain habitats. Therefore, natural disturbance may result in similar levels of modification of benthic habitats as anthropogenic disturbance such as dredging or trawling of the seabed. However, chronic physical disturbance (for example fishing pressure) that is greater in frequency and/or magnitude than natural disturbance, can alter community structure and function, remove biomass and reduce production (Hiddink *et al.* 2006).

The distribution of biological traits can reveal further insight into community assemblages occurring over different geographic locations or environmental conditions. The latter approach enables the quantification of ecological functioning and can be a useful tool in setting spatial boundaries for management (Frid *et al.* 2008) as different taxa demonstrate varying susceptibilities to both capture and damage from dredges (Hinz *et al.* 2011). Both of the latter studies concluded that the communities present were adapted to the naturally dynamic environment and therefore a limited short-term dredge fishery appeared to have an effect on communities no greater than natural perturbation. A greater shift in sediment composition than community composition in control areas, suggested that the natural environmental regime in that area was highly dynamic. Therefore, in order to assess the impact of scallop dredging on the seabed it is important to consider the following: which species and habitats occur within the area affected by the fishery and their vulnerability to natural and fishing disturbance; the substratum type; the frequency and intensity of disturbance from fishing and natural physical disturbance.

Seabed conditions are influenced by multiple, complex biotic and abiotic processes and the interactions therein (Snelgrove & Butman, 1994); including temperature, sedimentary and hydrodynamic conditions and inter-population relationships between species. Variation in seabed conditions can be characterised by physical parameters (such as chlorophyll-*a* concentration, sediment type, mean sea bed temperature, annual temperature range and inter-annual seabed temperature variation). Although the variables mentioned are not an exhaustive list of those that influence habitat and species composition, they are environmental variables for which data are available for most areas of the continental shelf in northern Europe. These variables are also considered important, at least in part, for helping predict the distribution of benthic species (Kostylev & Hannah 2007) and are partly responsible for variation in community composition and distinct faunal distributions (Table 4.1). Environmental variables influence the growth and survival of benthic organisms. Tidal bed shear stress is a function of the maximum predicted tidal current and the bed friction coefficient and reflects physical disturbance at the seabed. BSS is a strong predictor of sediment type, benthic biomass and production (Wildish & Peer 1983; van Rijn 1993) and influences food availability and the ability of organisms to filter feed. The growth rate and maximum size of king scallops in the English Channel is influenced by chlorophyll-*a* concentration, temperature and bed shear stress (Smith *et al.* 2007). Fish and macro-crustacean community assemblages in the English Channel are also linked to variation in depth, salinity, stratification, temperature and bed shear stress (Vaz *et al.* 2007). Depth is considered a generic descriptor of varying environmental conditions and is correlated with temperature, shear stress and sediment type.

Table 4.1: Environmental parameters that correlate with benthic community composition.

Environmental parameter	**Evidence**
seabed temperature	Holme 1961, 1966; Davies & Guinotte, 2011
sediment	Holme 1961, 1966; Fresi *et al.,* 1983; Roy *et al.,* 2014
chlorophyll-*a* concentration	Holme 1961, 1966; Eleftheriou & Basford 1989; Heip *et al.,* 1992, Giberto *et al.,* 2004
sea surface temperature	Bremner *et al.,* 2006
latitude	Bremner *et al.,* 2006
depth	Bremner *et al.,* 2006; Davies & Guinotte, 2011; Roy *et al.,* 2014
fish taxon richness	Bremner *et al.,* 2006
sediment mobility	Eleftheriou & Basford 1989; Heip *et al.,* 1992; Giberto *et al.,* 2004
salinity	Eleftheriou & Basford 1989; Heip *et al.,* 1992; Giberto *et al.,* 2004
organic carbon	Eleftheriou & Basford 1989; Heip *et al.,* 1992; Giberto *et al.,* 2004
tidal bed shear stress	Snelgrove & Butman 1994; Vaz *et al.,* 2007; Stokesbury *et al.,* 2006; Sciberras *et al.,* 2013

Seabed recovery

When considering the impacts of disturbance it is important to understand the timescales over which recovery of the species and habitats in question may occur. Blyth *et al.* (2004) demonstrated similar species richness and biomass at regularly and periodically (seasonally) trawled sites off Start Point, Devon. However, sites that had never been trawled and those which had not been trawled in the previous 18 or 22 months had greater species richness and biomass. This indicates that benthic communities found in sedimentary habitats in the English Channel can begin to demonstrate signs of recovery >1.5 year post-fishing disturbance. Although recovery in highly dynamic sandy habitats can occur in <1 year (Lindholm *et al*. 2004; Sciberras *et al.* 2013), predicted recovery timescales for communities on sandy gravel seabed, typical of that found in the English Channel range from 2.5 to >10 years (Table 4.2).

Table 4.2: Predicted recovery timescales of benthic communities in sand and sandy gravel seabed habitats.

Authors	**Location**	**Seabed type**	**Estimated recovery time**
Hermsen *et al.*, 2003	Georges Bank, NW Atlantic	gravel	> 6 years for biomass recovery
Blyth *et al.*, 2004	English Channel	mixed coarse sediments	2.5 years for mobile epifauna, 3.5 years for sessile epifauna
Hiddink *et al.*, 2006a	North Sea	gravel, sand	SW North Sea after 1-3 years; NW North Sea after >10 years
Sheehan *et al.*, 2013	Lyme Bay, England	mixed coarse sediments	recovery of 3 indicator species evident after 3 years
Sciberras *et al.*, 2013	Cardigan Bay, Wales	mobile sand	< 1 year
Lambert *et al.*, 2014a	Irish Sea	coarse and hard	< 1 to >10 years

Policy context

Due to the economic importance of the king scallop fishery in the English Channel and requirements under the new EU Common Fisheries Policy (CFP) to maintain good environmental status (GES) of seabed habitats, it is imperative to understand the environmental impacts of the dredge fishery. Although studies regarding the impact of scallop dredging exist in other sea areas, it is clear that effects on biological communities vary widely with the environmental context.

The objectives of this study were to investigate the relative impact of scallop dredging on habitats at the scale of the king scallop fishery in the English Channel, by comparing epifaunal species diversity, composition and biomass at sites subjected to varying levels of dredge fishing, while accounting for variation in environmental parameters between sites (depth, tidal bed shear stress, seabed temperature, food availability, substrate type). Variation in the composition of biological and life-history traits of communities was also investigated. These findings provide insights into the environmental impacts of scallop dredge fisheries in mixed sediment seabed habitats and would inform a more ecosystem-based approach to management of the scallop fishery.

4.2 Methods

Quantification of habitat characteristics

In order to characterise and compare environmental conditions across the scale of the fishery, environmental parameters that can limit growth and/or reproduction for benthic organisms, and for which datasets were available, were selected. These variables were then used to identify locations covering the spatial extent of the scallop fishery in the English Channel that are subject to different environmental regimes. At each selected location, sample sites were chosen to cover a range of fishing intensities (methods described below). Surface chlorophyll-*a* concentration and water stratification can be combined to give an indication of the availability of food to benthic organisms (Kostylev & Hannah, 2007). Mean seabed temperature, seabed temperature range, and seabed temperature variability are measures of physiological stress and may therefore influence growth rates in benthic organisms, as well as define the geographic range of a species. Time series datasets for each of the above parameters were obtained (see below) and used to create raster data layers in ArcMap v. 10. Data were projected to WGS1984 UTM Zone 30N using a grid cell size of 500 m. Tidal bed shear stress (BSS) is a strong predictor of sediment type, benthic biomass and production (Wildish & Peer, 1983; van Rijn, 1993). A modelled data set from Hiddink *et al.* (2006a) was used to generate a raster data layer of BSS for the English Channel.

Seabed temperature

Monthly composites of modelled seabed temperature data were provided by NEODAAS (The Natural Environment Research Council Earth Observation Data Acquisition and Analysis Service) for the years 1990 to 2004 (Holt *et al.,* 2012). Data were provided as NetCDF files and data layers for single months were extracted and imported into ArcMap v.10. The mean temperature (T_{mean}) for each year was calculated from the 12 monthly values and then averaged to give an overall mean seabed temperature for the period 1990-2004. The amplitude of the seasonal cycle (temperature range, T_{range}) was calculated as the range between the minimum and maximum of all mean monthly values. The inter-annual temperature variation (T_{varib}) was calculated as the mean of the standard deviation of the mean temperatures for each month.

Food availability

Monthly composite sea surface chlorophyll-*a* (chl-*a*) data from MODIS satellite data were provided by NEODAAS for the years 2003-2012, in 8-bit GeoTIFF format. Zero values, representing areas where cloud cover obscured the satellite images, were removed. Layers for the spring months of March, April and May were imported into ArcMap and the mean chl-*a* values (mg m^{-3}) for each month were calculated. The mean of these values was then taken as the mean spring chl-*a* value for the period 2003-2012. Modelled monthly mean potential energy anomaly (PEA) data were obtained from the National Oceanography Centre (NOC) for the period 1994-2004. The mean for each of the months June, July and August were used to calculate an overall summer mean value. Food (chl-*a*) availability (*Fa*) was calculated using the following formula from Kostylev & Hannah (2007): $Fa = \log(C/D)\text{-}S$, where C is chl-*a* concentration, D is depth, and S is the stratification index. Log(C/D), S and the resulting *Fa* were all re-scaled to 0-1.

Fishing effort

Anonymised Vessel Monitoring System (VMS) data for all UK and non-UK scallop vessel activity in the English Channel was obtained from the Marine Management Organisation (MMO), and aggregated at a scale of 0.025 decimal degrees (equivalent to approximately 5 km^2). A detailed analysis of the implications of using VMS data aggregated at different spatial and temporal scales is presented in Chapter 3. The data used covered a three year time period from October 2010 to September 2013, as recovery is not expected to occur in the habitat types sampled within three years (see Table 4.1 and Chapter 3). Data provided for each VMS record included month, year, latitude, longitude, IFISH (UK Sea Fisheries Data Warehouse) gear code and time interval between successive records. Scallop vessels fish at speeds of >2 knots (nautical miles per hour) and <3.5 knots (Lee *et al.*, 2010, Lambert *et al.*, 2012), therefore fishing activity was estimated using only those VMS records where speed was 2, 3 or 4 knots (speed on VMS records is recorded as an integer value). Although a VMS transmission supposedly occurs once every two hours while a vessel is at sea, inspection of the data revealed high variation in transmission interval. Therefore, the sum of the time interval between successive VMS records, was used as a measure of relative fishing intensity (FI) and expressed as 'total hours fished' over the 3 year period (h $3y^{-1}$). During this time period, 50 % of 0.025 degree grid cells had been impacted by the dredge fishery.

Selecting sample sites and stations

The VMS data were displayed in ArcMap v.10 and areas frequented by scallop vessels were identified. Eight sampling areas were chosen (Figure 4.1) covering the main fishing locations across the extent of the fishery; as well as a range of environmental conditions. Within each of these areas, five sample stations, within an 8 km radius, were selected to cover a range of dredge fishing intensities (FI). Mean FI values across the five sample stations within each of the eight main sites are given in Table 4.3. Due to limitations on time and money, only sites on the English side of the Channel were sampled. As VMS data does not include vessels <12 metres L.O.A., which generally fish within 12 NM (nautical miles) of the coast, all sample sites were located outside the 12 NM limit.

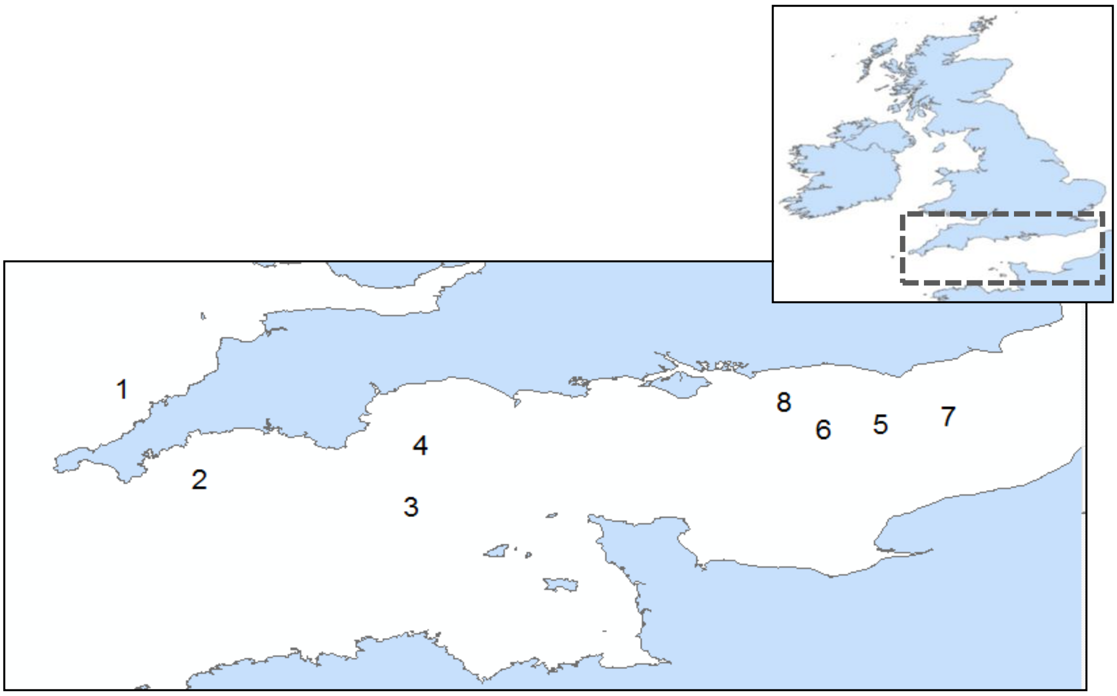

Figure 4.1: The central location of the eight main sample areas in the English Channel. Within each area five sample sites, each with a different estimated value of fishing intensity, were located within an 8 km radius.

Sites 5 to 8 were sampled between 21 and 24th August, and sites 1 to 4 between 20 and 24th September 2013. Scallop dredges were used to sample the abundance of king scallops, a 2-m beam trawl was used to sample the epifaunal community and an underwater camera was used to provide images for assessment of substrate type (further details below). Each sampling gear was deployed close to the station midpoint. The exact time and position of gear deployment and retrieval were noted from the ship's GPS (Global Positioning System).

Values for each of the environmental parameters and relative dredge fishing intensity for each station midpoint were extracted from the data layers in ArcMap. At the eight sample sites, sediment type ranged from sand interspersed with broken shell, to gravelly sand with pebbles and cobbles. Coordinates and values of the environmental variables for each sample site are in Appendix 4.1.

Table 4.3: Mean, median, minimum and maximum fishing intensity values for the five sample sites within each main site (1-8). Location relates to the eastern and western sides of the English Channel. Values relate to dredge fishing intensity expressed as the total number of hours fished over a three year period (hr $3yr^{-1}$).

site	location	mean	median	min	max
1	west	5.55	5.77	4.00	7.70
2	west	24.18	19.44	9.67	53.94
3	west	13.52	13.68	6.43	23.52
4	west	47.95	51.15	18.33	70.29
5	east	57.74	58.05	53.46	61.72
6	east	60.27	65.89	18.83	92.64
7	east	32.15	35.29	23.22	36.62
8	east	7.64	6.47	2.00	20.66

Habitat survey

Scallop dredging

At each sample station a gang of four spring-loaded Newhaven scallop dredges were deployed and towed for 20 minutes, at approximately 2.5 knots. Two king scallop dredges (9 teeth of 110 mm length, belly rings 80 mm diameter) and two queen scallop dredges (10 teeth of 60 mm length, belly rings 55 mm diameter) were used. The mean area sampled by the scallop dredges was 4362 m^2 (S.D.=757) per tow. The king dredges were used to sample the commercial catch of king scallops while the queen dredges were used to retain smaller king scallops (below the minimum landing size). The dredges were emptied on deck between metal dividers to ensure that the contents of each dredge remained separate for recording purposes. All king scallops were counted and weighed. The weight of inert ('*Inert*') material (gravel, rock and broken shell) was recorded as kg Ha^{-1} (kg per hectare). When the total bycatch comprised one or two five-stone fish baskets (one basket holds approx. 30-40 kg of catch), a sub-sample of one fish basket was quantified. All bycatch organisms were identified to species or genus level and then counted and weighed using a motion compensated digital scale (Marel M-Series 1100) to the nearest gram. For colonial species such as hydroids and bryozoans, only the total weight was recorded. When total weight was <0.001 kg, a weight of 0.001 kg was recorded. Samples were omitted if fishing gear failure occurred. The weights of sub-sampled bycatch were raised to estimate the total weight of each bycatch species from the haul.

Sampling of epifaunal communities

Epifaunal communities were sampled with a two metre beam trawl with 4 mm mesh (ICES standard design, Kaiser *et al.*, 2004; Jennings *et al.*, 1999), deployed at each station and towed for 5 minutes (±3 mins) at a speed of approximately 1.5 knots. The start and end coordinates were used to calculate the distance of each towed gear deployment. The width of the scallop dredges (0.76 m per dredge) and beam trawl (2 m) were multiplied by the distance of the tow to calculate the total area of seabed sampled during each deployment, and mean biomass (per tow, dependant on the number of dredges analysed) calculated per hectare (10 000 m^2). The mean area of seabed sampled by the beam trawl was 561 m^2 (S.D.=158) per tow. The whole of site 5 and site 8.1 were not sampled using the beam trawl as there were a high proportion of large stones at these sites and therefore a risk of damaging the gear. Therefore the community analysis and PCA of environmental variables (see below) excludes site 5 (scallop biomass data from the dredge catches at site 5 were retained). The contents of each beam trawl were identified to species or genus level and the total count and/or weight of each species was recorded.

Observations and analysis of seabed type

To provide information on seabed substratum composition, a GoPro Hero3 camera mounted on a steel sledge with LED lights and scaling lasers was deployed and towed for approximately 10 minutes at a speed of 1 knot (depending on the strength of the tide). A 10 megapixel photograph was recorded every 10 seconds. The camera was mounted on the sledge at an angle of approximately 25° relative to the seabed. The distance between the lasers was measured and the width of the field of view calculated so that the images could be used to estimate sediment grain size. Due to the angle of the camera, the field of view was greater at the top of each image than the bottom. The width of view halfway up each image (0.96 m) was therefore used to calibrate the images in order to estimate sediment particle size. Distortion at the image edge was not corrected for; however, as the camera/sledge geometry was kept constant, the effects of distortion will be the same across all sediment types. Seabed sediment was described as the percentage cover of different sediment grain sizes (based on the Wentworth scale; Wentworth, 1922) derived from the camera images using ‘Coral Point Count’ (CPCe) software (Kohler & Gill, 2006). Between 140 and 204 images were available for each station. Ten images spaced at approximately equal intervals were selected for analysis at each station. In CPCe, 10 stratified random points were generated on each image and the type of substratum under the point recorded (Table 4.4). Analysis was trialled using 15 and 20 points per photo but there was no difference in the outcome therefore 10 points were used for each image. From these 10 records the mean percentage cover of each sediment particle size was calculated. ImageJ (Schneider *et al.*, 2012) was used to set the scale and measure individual sediment particles on the photographs. The resolution of the images prevented accurate measurement of particles <4 mm in diameter, hence <4 mm was the smallest particle size class recorded. Therefore mud, fine sand, coarse sand and gravel were combined into one category, ‘*Sand_grv*’ (see Appendix 4.1).

Table 4.4: Substratum types and particle sizes identified and recorded from seabed images for sediment size analysis. n/a = not applicable. Classifications of particle size taken from Wentworth 1922.

Wentworth description	**Particle diameter**	**Image classification**
Mud	< 0.063 mm	< 4 mm Gravel, Sand, Mud
Fine sand	0.063 – 0.25 mm	
Coarse sand	0.25 – 2 mm	
Gravel	2 – 4 mm	
Pebble	4 – 64 mm	Pebble
Cobble	64 – 256 mm	Cobble
Boulder	> 256 mm	Boulder
Bedrock	no specific size	Bedrock
n/a	n/a	Shell
n/a	n/a	Algae
n/a	n/a	Organism
n/a	n/a	Shadow (view obstructed by blur or shadow)

4.3 Data analysis

Quantification of habitat characteristics

Using the software PRIMER v.6 (Clarke & Gorley, 2006), a draftsman plot was used to investigate significant auto-correlation between environmental variables. '*Fa*' was highly correlated with 'T_{mean}' (ρ=0.95), therefore '*Fa*' was excluded from the multivariate analyses described below (Clarke & Warwick 2001). Data for the variable '*Inert*' (the weight of inert material retained in the dredge samples) were square-root transformed and the whole dataset was normalised, then a resemblance matrix was produced by computing the Euclidean distance between each pair of stations. To account for the variation in environmental characteristics across sample sites a cluster analysis with 999 permutations was used to identify significant groupings of sample stations based on their similarity in terms of the environmental variables. A principle component analysis (PCA) was performed to establish which of the environmental variables explained the greatest variation between stations. The BIOENV procedure in PRIMER was used to investigate which environmental variables produced the highest correlation with species composition and the significance of the correlation was tested using the RELATE procedure (site 5, at which the beam trawl was not deployed, was excluded from this analysis).

Effects of natural and fishing disturbance on species diversity and community composition

No single fishing gear samples the entire benthic community, resulting in semi-quantitative sampling (Reiss *et al.*, 2006). Also, catching efficiency of both the beam trawl (Reiss *et al.*, 2006) and scallop dredges (Dave Palmer, Centre for Environment, Fisheries and Aquaculture Science, pers. comm.) reduces with increasing sediment particle size. However, the 2-m beam trawl, due to its small mesh size is considered to be the device that gives the most consistent and comprehensive data in relation to the epibenthic assemblage, therefore this dataset was used for analyses of community composition. Data from the king dredges was used to quantify the retained biomass of king scallops, as they are rarely retained by the beam trawl. Non-parametric multivariate analyses of the community assemblage data were carried out in the statistical package PRIMER using the species biomass dataset for the beam trawl samples from sites 1-4 and 6-8. The DIVERSE function in PRIMER was used to calculate univariate measures of species diversity at each site using the un-transformed, un-aggregated biomass data: mean total number of species (*S*), total biomass at each site (*N*), Species richness (count of the number of different species), Shannon diversity index (*H*), Pielou's evenness index (*J*) and Simpson's index (*D*) (in the form 1-λ'). Simpson's index gives greater weighting to the more abundant species in a sample while the presence of rare species in a sample results in only small changes in the value of *D*. A multiple regression was performed to test for the effects of the environmental parameters (T_{mean}, T_{range}, T_{varib}, BSS, Inert, Sand_grv, stratification; Chl-*a*) on species richness. Model assumptions were assessed using plots of the residuals. Relationships between gross community metrics; *P. maximus* biomass retained by the dredges, and fishing intensity (FI) and tidal bed shear stress (BSS) were tested, controlling for the effect of habitat group ('Deep' or 'Shallow') using a one-way ANCOVA (Analysis of Covariance) in R (R, 2008) ('aov' function), with a Bonferroni correction for multiple comparisons, which reduces the likelihood of a Type I error. Model residuals were checked for normality of distribution and homogeneity of variance.

The beam trawl species dataset was aggregated to genus level and a square-root transformation was applied to down-weight the influence of highly abundant or rare species. In PRIMER, a resemblance matrix was created using the Bray-Curtis similarity index. Sample stations were ranked based on FI values and each sample station was categorised as low (<10 hours), medium (10-30 hours), or high (30.1-93 hours) fishing intensity (FI) based on arbitrary divisions to provide a relatively even number of stations in each category. This meant that in some instances stations within the same site were ranked in different categories. Similar groups were created for low (0.24-0.70 N m^{-2}), medium (0.71-1.04 N m^{-2}) and high (1.05-1.70 N m^{-2}) BSS. A PERMANOVA (Permutational Multivariate Analysis of Variance, Anderson 2005), was used to test for differences in species composition (using the aggregated dataset) between sites, using the factors FI/habitat group and BSS/habitat group. SIMPER analysis was used to identify characteristic species for the groups identified by the cluster analysis and samples grouped by BSS (low, medium and high). Key species identified in the SIMPER analysis were used for further univariate analysis against FI and BSS using a GLM approach.

Biological traits analysis

Thirteen ecological and life-history traits were selected, based on their relevance to ecosystem functioning and the availability of trait information for the species in the beam trawl dataset (Table 4.5). Each trait was separated into a number of modalities (categories) resulting in 53 modalities in total. Information on traits was gathered from various sources (see Appendix 4.2).

Table 4.5: Biological and life-history traits and the related modalities used in a biological traits analysis of species between sample sites.

Trait	Modalities
Maximum size (cm)	<1 1.1-2.0 2.1-10.0 10.1-20.0 20.1-50.0 >50
Morphology	Soft Tunic Exoskeleton (chitin/calcium carbonate) Crustose Cushion Stalked
Longevity (years)	<1 1 to 3 4 to 10 >10
Larval development location	Pelagic (planktotrophic) Pelagic (lecithotrophic) Direct development
Habitat	Infauna Epifauna
Living mode	Burrow-dwelling Free-living Crevice/hole/under stone Epizoic/endozoic/epiphytic Attached to substratum
Feeding mode	Deposit Filter/suspension Browser Scavenger Predator
Mobility	Sessile Swim Crawl/creep/climb Burrower
Age at maturity (years)	<2 2 to 5

	6 to 10 >10
Food type	Algae Benthic organisms Detritus Plankton Suspended organic matter Micro- organisms
Reproductive frequency	Annual Biennial Semelparous
Fragility	Fragile Intermediate Robust
Sociability	Colonial Gregarious Solitary

As species can exhibit more than one trait modality (e.g. deposit and suspension feeding) a 'fuzzy coding' approach was used (Pop, 2001), where each taxa is coded according to the relative extent to which it displays each trait (Chevenet *et al.,* 1994). Fuzzy coding eliminates the effects of outliers or poorly distributed variables from analysis. When information regarding traits was not available, expert judgement was used to assign traits based on a comparison with similar species, genus or family (as per Bolam *et al.,* 2014). If no information on a particular species' trait was available the average score for the trait was assigned, so as to not influence final results (Chevenet *et al.,* 1994). The resulting 'taxa by trait' matrix was converted to proportions providing a total value of 1 for each taxon and trait. The beam trawl biomass for each taxa was then multiplied by the fuzzy-coded trait proportion to produce a biomass-weighted trait-by-station matrix and a fuzzy correspondence analysis (FCA) was carried out using the 'ade4' package in R (Dray *et al.*, 2007). The relationship of FI and BSS with the distribution of biological traits across stations was investigated using the multivariate methods described above in PRIMER and visually, from plots of the primary FCA axis values against FI and BSS. An asymptotic trend between FCA1 and BSS was investigated using non-linear modelling approach using the *SSasymp* function from the *stats* package in R (R, 2008).

4.4 Results

Quantification of habitat characteristics

A multiple regression of the environmental parameters against species richness revealed a significant relationship only with BSS ($F=2.305_{8, 23}$, $p=0.02$). A principal component analysis (PCA) of environmental variables (excluding '*Fa*') indicated that PC1 explained 50 % of the variation between sample stations, and PC2 a further 18 %. PC1 was mainly influenced by a similarly weighted combination of T_{range} and T_{mean} in one direction and depth in the opposite direction. Influencing PC2 were, in order of strength of explanatory power: T_{varib} and *BSS* in one direction, and '*Inert*' in the opposite direction (Figure 4.2). There were two significant groupings of sites ($p=0.01$) based on similarity in environmental variables. The first group included sites 1, 2 & 3 (all in the western English Channel, deeper sites, >53 m); and the second group included sites 4, 6, 7 & 8 (shallower sites, 18-53 m). Hereon in the former group is referred to as 'Deep' and the latter group as 'Shallow'. These groupings are demonstrated on the PCA plot (Figure 4.2). Deeper sites had lower T_{mean} and T_{range} values, lower BSS and a higher proportion of sand and gravel sediments, while shallower sites had higher levels of natural disturbance (BSS), and a higher proportion of large rocks (Table 4.6). BSS was highest at the eastern English Channel sites (5, 7 & 8, see Table 4.6), with similar values at site 3 (offshore western English Channel).

Table 4.6: Mean values for environmental parameters at each of the 8 sample sites.

Site	Group	BSS (N m^{-2})	Depth (m)	T_{mean} (°C)	T_{range} (°C)	T_{varib} (°C)	*Sand_grv* (%)	*Inert* (kg km^{-2})
1	Deep	0.79	57.60	11.28	7.91	1.93	64.36	177.99
2	Deep	0.26	72.80	10.43	8.06	1.97	37.16	61.17
3	Deep	1.00	54.80	10.41	8.19	1.89	28.40	255.39
4	Shallow	0.63	51.20	11.73	9.61	2.00	52.64	9.20
5	Shallow	1.37	48.67	11.96	10.38	1.96	3.67	1035.89
6	Shallow	0.80	38.40	12.01	10.25	1.95	0.00	688.46
7	Shallow	1.39	34.00	11.97	10.53	2.00	6.12	466.85
8	Shallow	1.14	49.40	11.87	10.41	1.99	19.80	206.77

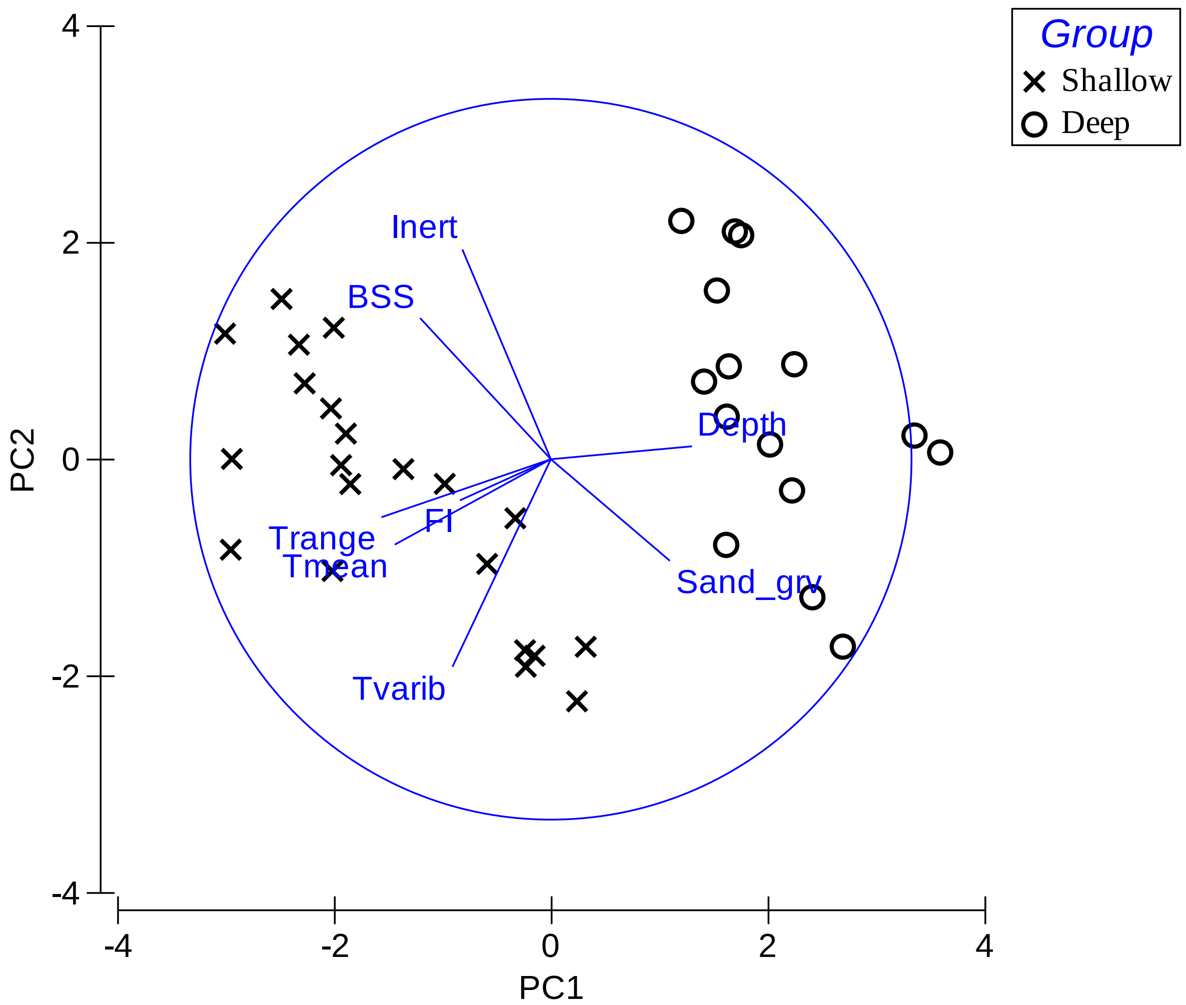

Figure 4.2: Results of a principal components analysis of the environmental variables at each site: Depth, BSS, T_{range}, T_{varib}, T_{mean}, *Inert* and *Sand_grv*. PC1 accounts for 50 % of variation between sample sites and PC2 a further 18 %. Symbols represent sample sites grouped according to their similarity in environmental parameters: 'Shallow' or 'Deep'.

Effects of natural and fishing disturbance on species diversity and community composition

In total, 143 species were identified from all sample stations with all gear types, 23 for which only biomass was recorded. *P. maximus* biomass retained by the dredges increased with FI (Table 4.7, Figure 4.3 lower panel right) with no habitat group ('Deep'/'Shallow') effect. There was no significant relationship between *P. maximus* biomass and BSS. ANCOVA tests revealed no significant relationship between FI and the total number of species, species richness, Shannon diversity index, Pielou's index or Simpson's index, with no interaction between the response variables and habitat group. There was a significant difference in the relationship between the total number of species and FI, for the two habitat groups, however the relationships themselves were not significant. There were significant negative relationships between BSS and species richness, Shannon index and total number of species (Figure 4.3, Table 4.7), with no significant habitat group effect.

Table 4.7: Results from ANCOVA tests for significant relationships of FI (hrs $3yr^{-1}$) or BSS (N m^{-2}) with the gross community metrics and *P. maximus* biomass or habitat group (Deep or Shallow), testing for significant interactions between FI/BSS and group. Intercept and slope values given for significant relationships.

response variable	independent variable	group effect	interaction	d.f.	F	p	intercept	slope
Species richness	FI	no	no	1, 28	0.44	0.512		
Shannon index	FI	no	no	1, 28	1.09	0.307		
Total number of species	FI	yes	no	1, 28	0.80	0.379		
Pielou's evenness	FI	no	no	1, 28	2.94	0.097		
Simpsons index	FI	no	no	1, 28	3.39	0.076		
P. maximus biomass	FI	no	no	1, 34	9.95	0.003	3.81	0.211
Species richness	BSS	no	no	1, 28	18.67	0.0002	12.42	-6.163
Shannon index	BSS	no	no	1, 28	6.83	0.014	2.31	-0.624
Total number of species	BSS	no	no	1, 28	16.12	0.0004	30.87	-9.761
Pielou's evenness	BSS	no	no	1, 28	0.52	0.479		
Simpsons index	BSS	no	no	1, 28	0.67	0.420		
P. maximus biomass	BSS	no	no	1, 34	0.41	0.525		

The community composition was analysed for the two habitat groups ('Deep'/'Shallow'). There were significant differences in species composition between the 'Deep' and 'Shallow' groups (p=0.003), however the effect of FI was non-significant, and there was no interaction between habitat group and FI. There was a significant difference in species composition between levels of BSS (p=0.0001) and there was a significant interaction between BSS and habitat group. A rank correlation coefficient of 0.347 occurred between the species resemblance and environmental variable matrices, with the variables BSS, *Sand_grv* and T_{range}, best explaining the patterns of species composition between sites. The Spearman's rank correlation value was ρ=0.256, p=0.001. When single variables were tested, the highest correlation values were T_{range} (0.285) and BSS (0.276).

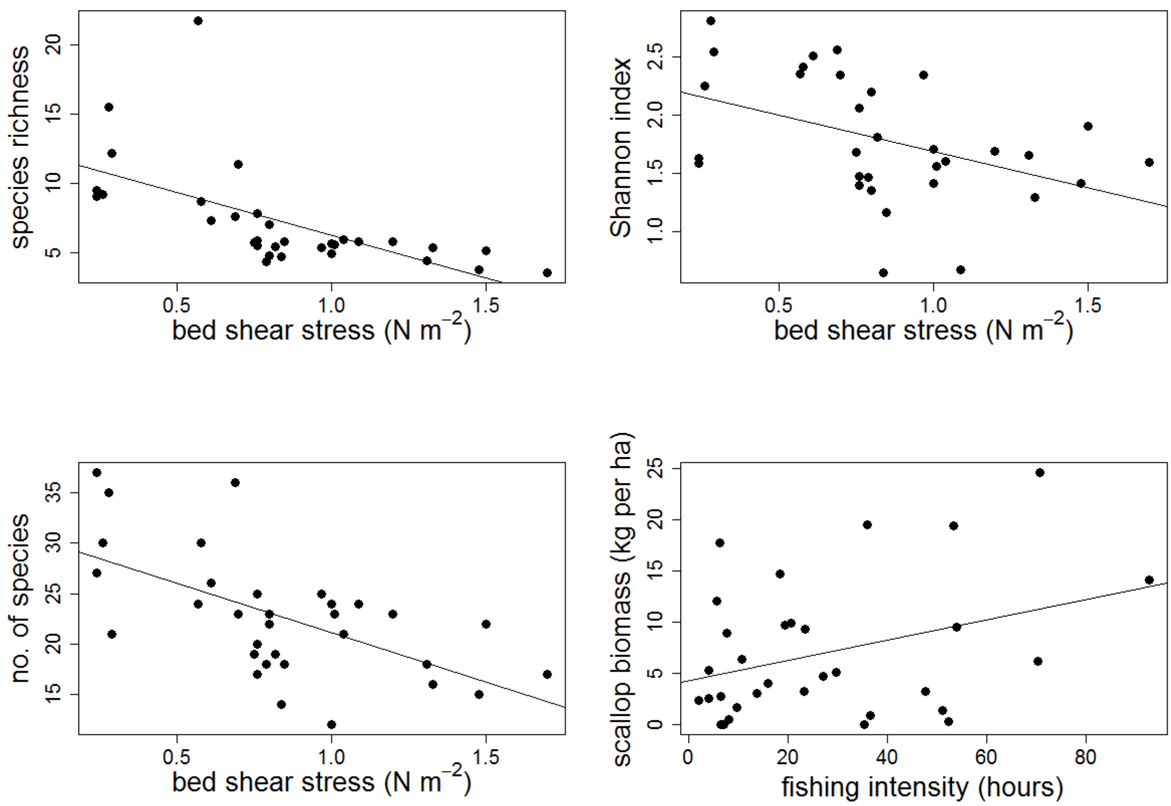

Figure 4.3: Plots of the relationship between species richness, Shannon diversity index, the total number of species and BSS; and the relationship between scallop biomass in the dredges and fishing intensity. Significant fitted relationships are shown by a solid line. See Table 4.7 for intercept and slope values.

'Deep' and 'Shallow' had 14 and 7 taxa respectively that contributed to 80 % of the similarity among stations within each of these groups (full output in Appendix 4.3). In both habitat groups, these species included *Aequipecten opercularis*, *Alcyonium digitatum*, *Pagurus* sp., *Asterias rubens* and *Psammechinus miliaris* (Table 4.8). Brittlestars, *Ophiura* sp. (Deep) and *Ophiothrix* sp. (Shallow) also contributed to the top 80 % similarity within groups. A similarity/standard deviation (Sim/SD) value >1.3 indicates that the biomass of a species is consistent across samples within a group and therefore a typical species for the group (Clarke & Warwick 2001). Six typical species were identified and contributed to the top 80 % between group dissimilarity, including *A. opercularis*, *A. rubens*, *P. miliaris*, *Pagurus* sp., *Macropodia* sp. and *Ophiura* sp. There was a higher biomass of *A. opercularis*, *A. rubens* and *P. miliaris* in 'Shallow', while *Macropodia* sp. and *A. digitatum* had higher biomass in 'Deep'.

Table 4.8: Results from ANCOVA tests for significant relationships of FI (hrs $3yr^{-1}$) or BSS (N m^{-2}) with indicator species or habitat group (Deep or Shallow), testing for significant interactions between FI/BSS and group.

species	dependant variable	group effect	interaction	d.f.	t	p
A. opercularis	FI	no	no	1, 28	1.459	0.155
A. rubens	FI	no	no	1, 28	0.345	0.733
C. intestinalis	FI	no	no	1, 28	-1.371	0.181
P. miliaris	FI	no	no	1, 28	0.958	0.060
Pagurus spp.	FI	no	no	1, 28	1.093	0.283
A. opercularis	BSS	no	no	1, 28	1.267	0.215
A. rubens	BSS	no	no	1, 28	1.925	0.064
C. intestinalis	BSS	no	no	1, 28	-0.253	0.802
P. miliaris	BSS	no	no	1, 28	1.926	0.064
Pagurus spp.	BSS	no	no	1, 28	0.959	0.345

The species contributing the greatest within group similarity at medium and high BSS stations were *A. opercularis*, *Pagurus* sp., *A. rubens*, *P. miliaris* and the brittlestars *Ophiura* sp. (medium) and *Ophiothrix* sp. (high). Species that contributed to the greatest within group similarity for stations with low BSS included sessile, small body-sized and fragile taxa that are less resilient to life in disturbed environments. Typical species included *Pagurus* sp., *A. digitatum*, *Macropodia* sp., *A. opercularis* and *P. miliaris*. Other species that were relatively abundant at low BSS but with Sim/SD values <1.3 were the small crustaceans *Inachus* sp. and *Liocarcinus* sp., branching hydroids and bryozoans (*Nemertesia* sp. and *Cellaria* sp.), the Devonshire cup-coral *Caryophyllia smithii* and small fish species such as *Microchirus variegatus*, *Callionymus* sp. and *Trisopterus* sp..

Univariate analysis of the key species revealed no significant relationships with FI or BSS (Table 4.8). Many other species had low biomass and were present at only a few of the sample sites. Therefore, it was not possible to test the relationship between the biomass of those species with FI or BSS due to the high number of zeros in the dataset.

Biological traits analysis

A plot of the fuzzy correspondence analysis (FCA) revealed that most of the stations were clustered together meaning they have similar biomass weighted trait distributions, although stations from sites 1 and 2 (the most westerly sites) were more distinct from other stations (Figure 4.4). FCA axis 1 accounted for 33 % of the variation in traits between samples and FCA axis 2 contributed a further 19 % (Table 4.9). There are no environmental characteristics clearly distinct to sites 1 and 2, although relatively low temperature ranges occur at both. The traits that contributed most to the variation between stations are those to the left of the main cluster (Figure 4.4). The trait modality ‘Epizoic’ relates to one species *Scalpellum scalpellum*, the goose-necked barnacle that attaches to hydroids and bryozoans, which are abundant at scallop fishing grounds. Longevity of <1 year (‘Long .1’), body size of 1-2 cm (‘size 1.2’), ‘crustose’ morphology and ‘semelparous’ reproduction all relate to less than 0.1 % of the total biomass, so are not considered to be indicator traits. There was a high biomass of ‘sessile’ organisms at sites 1 and 2, and around 50 % of the overall biomass of ‘colonial’ and ‘attached’ organisms and those with ‘soft’, ‘cushion’ or ‘stalked’ morphology occur at these sites. Sites 1 and 2 also had 70-90 % of the overall biomass of the traits; ‘lecithotrophic’ larval development, ‘tunic’ morphology and a life span of 1-3 years (‘Long 1.3’).

Table 4.9: Eigenvalues and projected inertia values from a fuzzy correspondence analysis of biological and life history trait biomasses between samples.

	Axis 1	**Axis 2**	**Axis 3**	**Axis 4**	**Axis 5**
Eigenvalues	0.1516	0.0851	0.0643	0.0412	0.0309
Projected inertia (%)	33.06	18.55	14.02	8.97	6.74
Cumulative inertia (%)	33.06	51.61	65.63	74.60	81.34

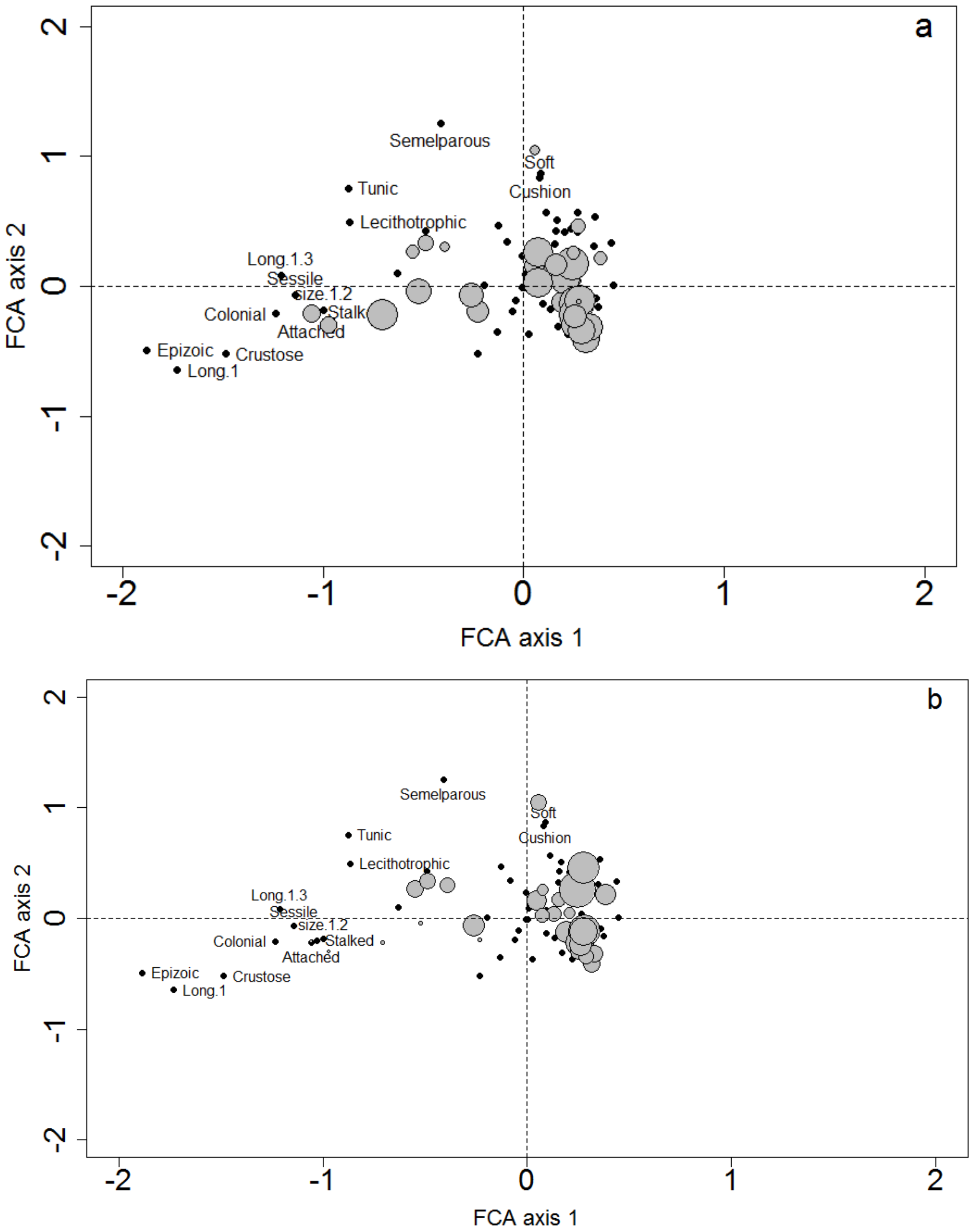

Figure 4.4: Plots of ordination scores from fuzzy correspondence analysis of biological and life-history traits (black dots), only traits with a factor score >=0.7 have been labelled to aid interpretation. Grey circles represent a) the dredging intensity [log(hours $3yr^{-1}$)], b) BSS. Larger circles indicate higher values. Stations that are close together have similar distributions of trait biomasses across modalities. Axis 1 accounts for 33 % of total inertia, Axes 1 and 2 account for 52 % of the variation in trait biomasses between sites.

ANOSIM revealed no significant differences in species composition between low, medium and high FI stations (R=0.005, p=0.376); or between BSS groupings (R=0.052, p=0.115), therefore SIMPER analysis was not carried out on the trait data. A plot of the FCA1 scores against FI revealed no trends; however a plot of FCA1 scores against BSS revealed an asymptotic trend (Figure 4.5). This suggests that there is a threshold at which the effect of BSS on trait composition is absolute. The relationship was significant (p=0.009) and the curve was described by the equation:

FCA1 = *Asym* + (*R*0-*Asym*) * exp(-exp(*lrc*) * *BSS)*

where *Asym*=0.328, is the horizontal asymptote; *R*0= -222.280, the response value when *BSS* is zero; *lrc=2.893*, a value representing the natural logarithm of the rate constant.

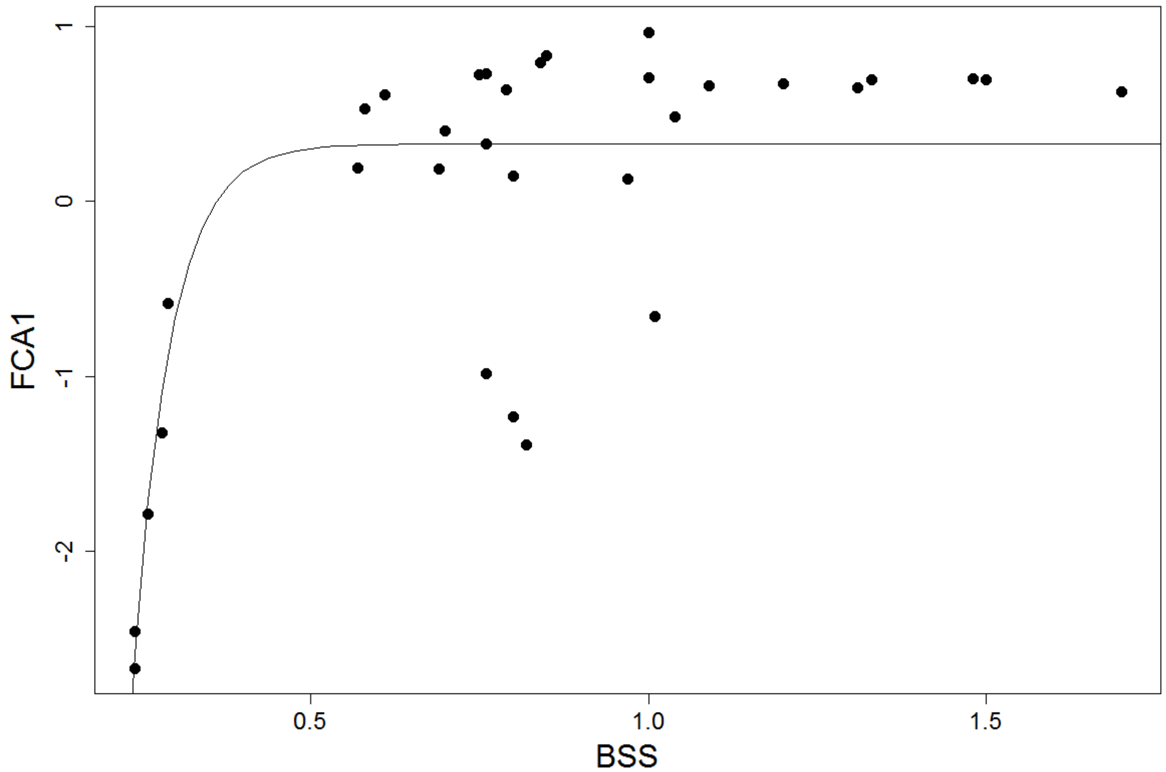

Figure 4.5: The asymptotic relationship between BSS and FCA axis 1 scores for the distribution of biological traits between sample sites. The fitted curve is significant (p=0.009 and described by the equation *FCA1* = 0.3284 + (-222.280 – 0.328) * exp(-exp(log(2.893)) * *BSS*).

4.5 Discussion

Habitat characteristics within the fishery

Of the environmental parameters included in the analysis, only BSS had a significant relationship with species richness. Hence, although all the parameters are known to influence benthic species composition, BSS is considered the most important driver of species richness on scallop fishing grounds in the English Channel. In addition to BSS, seabed temperature range (T_{range}) was also a key driver of community composition. These are both environmental factors to which benthic species generally have a tolerance range (Hall 1994; Hiscock *et al*. 2004). Of the sites sampled, two habitat groups were identified based on similar environmental characteristics, defined largely by water depth (Kaiser & Spence 2002). There were significant differences in species composition between the 'Deep' and 'Shallow' habitat groups (with a degree of overlap between the groups). There was a moderate correlation (0.347) between the environmental and species resemblance matrices, indicating that a more complex suite of variables than those included in the present study influence species composition.

Effects of natural and fishing disturbance on species diversity and community composition

Dynamic environments mask potential community changes caused by fishing impacts. Therefore, it can be difficult to separate the effects of fishing from the inherent natural variability in dynamic shelf-sea systems. Furthermore, there is no well-defined reference data for which comparisons of fished and unfished habitats can be made. Commercial scallop vessels have operated in the English Channel for over half a century therefore any changes caused by historical fishing activity remain unknown (Gislason 1994).

Our ability to detect changes attributed to fishing pressure also relates to our ability to reliably determine fishing pressure at an appropriate scale and distinguish those effects from the natural variability of the system. Aggregated VMS data has inherent limitations that can either over or under-estimate fishing pressure, depending on the scale of aggregation and the number of data points available (Gerritsen *et al.* 2013). However, the likelihood of any over- or under-estimation of fishing effort is considered to be consistent between sample stations; and it is relative, rather than absolute, fishing pressure that provides the basis for this study.

When environmental differences between sites were accounted for in multivariate analyses, no significant relationships were found between recent (3 yr) dredge fishing intensity and measures of gross community metrics (total number of species, species richness, Shannon diversity index, Pielou's evenness index and Simpson's index) across scallop fishing grounds in the English Channel. However, a significant negative relationship occurred between bed shear stress (BSS) and the total no. of species, species richness and diversity. As BSS increases so does the amount of stress on benthic organisms and high BSS is associated with highly dynamic systems where only a reduced assemblage of the most resilient species are able to succeed. There are three potential hypotheses for a lack of relationship between fishing intensity and species diversity: the environmental regime (including natural physical disturbance) has a greater influence on species composition at the sites studied (Stokesbury & Harris 2006) and scallop dredging has no further impact; the effects of low dredge intensity are absolute (Hall-Spencer & Moore 2000) and the frequency of fishing events does not allow recovery of the seabed and communities (the most recent dredging activity occurred <1 year ago at all sites sampled); or the communities at the sites are held in an altered state due to historical scallop dredging activity (Collie *et al.* 2000). Scallops have been commercially fished in the English Channel for much of the last century; however, the number of vessels in the fishery increased dramatically in the mid-1970s and has remained high ever since. In the English Channel, scallop beds also overlap with beam trawl fishing grounds, resulting in additional anthropogenic disturbance, although this was not included when calculating fishing effort in the present study. The benthic communities that result from this pattern of disturbance within trawl and dredge disturbed areas may be tolerant of this regime of disturbance, although this hypothesis requires further investigation. In addition to the above considerations, the lowest levels of BSS occurring at sites in the present study are relatively high compared to other studies on the impacts of towed bottom-fishing gear (Tillin *et al.* 2006). Therefore, it may be expected that the communities present are already tolerant of physical disturbance.

A caveat of the sampling method is that the catching efficiency of beam trawls is likely to be correlated with particle size, and the effect might vary for different sizes of organisms. However, there is no data available with which to construct correction coefficients. As sediment particle size is correlated with bed shear stress there may be some confounding between these two variables.

Results from the multivariate community analysis indicated similar species composition across all the sites sampled, irrespective of fishing intensity. Mobile opportunistic scavengers, including starfish, brittlestars, urchins and small crustaceans, were common across all sites. Bed shear stress is a significant modifier of gross community metrics such that more species (and less common species) occurred in areas with lower bed shear stress. Differences in species composition between sites with varying levels of bed shear stress were largely attributed to variation in the biomass of six taxa (*Aequipecten opercularis, Asterias rubens, Ciona intestinalis, Cellaria* sp., *A. digitatum* and *P. miliaris*) and the number of rarer species that occurred at each site. Species found at sites with higher BSS are those which are better able to either survive, or avoid bottom-towed fishing gear, can utilise detritus and dead organisms left in the path of the dredge and are faster to recolonize an area post-dredging. This is further supported by analysis of biological traits (see below). These species included: *A. opercularis*; the brittlestars *Ophiura* sp. and *Ophiothrix* sp., *Pagurus* sp.; *A. rubens* and *P. miliaris*. The latter three species are resilient scavengers (Lawrence 1975; Groenewold, & Fonds 2000) and are likely to be common on seabed areas subject to fishing disturbance. Collie *et al.* (1997) also found that scavenging biota were dominant in areas subject to scallop dredging activity. The queen scallop, *A. opercularis* is considered resilient to fishing (Nall 2011).

Relationships between typical species and FI or BSS were not found to be significant. *P. miliaris* and *A. rubens* are both opportunistic scavengers with life-history or morphological traits that aid survival in disturbed conditions. The biomass of both species appeared to be positively correlated with both increasing fishing intensity and bed shear stress (although in all cases on the margin of significance). Small spider crabs, *Macropodia* sp. and *Inachus* sp., were prevalent at many sites. Although these species appear to be fragile, they demonstrate the ability to curl into a ball to prevent damage during dredging (Bradshaw *et al.* 2002). The positive relationship between fishing intensity and catches of king scallops in the dredges was unsurprising, given that scallop fishermen actively target areas with higher concentrations of scallops in order to maximise catch rates. However, the biomass of scallops in an area will also be influenced by recent fishing activity that leads to localised depletion of stocks. Scallop biomass did not vary with BSS, indicating that scallops are able to tolerate a greater range of BSS than many of the benthic organisms identified in this study. This makes them an ideal species for harvesting; resilient to physical disturbance and able to survive over a broad range of habitat types. The species assemblage described above represents the community typical of commercial scallop fishing grounds in the English Channel. Species that contributed most to the similarity within groups were those with behavioural or morphological traits that aid survival in disturbed habitats (Table 4.10). Traits occurring at sites 1 and 2 (sessile, 'colonial' 'attached', 'soft', 'cushion' and 'stalked') are typical of less disturbed habitats. Sites 1 and 2 had the lowest and 3rd lowest BSS respectively.

Bradshaw *et al.* (2001) demonstrated long-term community changes on fishing grounds around the Isle of Man by comparing taxonomic datasets spanning a 60 year time period. The study found no relationship between the level of fishing intensity and taxonomic distinctness, although certain species had decreased in abundance over time. These included, amongst others: *Ophiothrix fragilis*; three hydroid species; two upright, and a number of encrusting bryozoans; encrusting worms (*Serpulidae* and *Spirorbidae*). Other species had increased in abundance: the brittlestars *Ophiocomina nigra*, *Ophiura albida* and *Amphiura filiformis*; squat lobsters (*Galathea intermedia*); spider crabs including *Macropodia* sp. and *Inachus* sp.; some upright sessile organisms such as *Nemertesia* sp., *Ascidiella* sp., and *Hydrallmania* sp.; the gastropods *Calliostoma zizyphinum* and *Buccinum undatum*; the common starfish *Asterias rubens*; hermit crabs, *Pagurus* sp.; and pectinid bivalves. All the latter taxa commonly occurred at sites in the present study. Thus around the south and west coast of the British Isles there are similarities in community composition and traits associated with areas subject to scallop fishing activity.

Table 4.10: Biological species traits that aid survival in disturbed habitats, with examples of species that exhibit those traits that were commonly found across sites in the present study.

Biological trait	**% of biomass represented across all samples**	**Species**
mobile	62%	*Aequipecten opercularis, Asterias rubens, Ophuira albida, Pagurus* sp., *Ophiothrix* sp.
broadcast spawner/ fast coloniser	67%	*Aequipecten opercularis, Nemertesia* sp.
scavenger/predator	50%	*Asterias rubens, Ophuira albida, Buccinum undatum, Calliostoma zizyphinum, Pagurus sp.*
intermediate/robust morphology	66%	*Aequipecten opercularis, Buccinum undatum, Calliostoma zizyphinum, Pagurus* sp.

Biological traits

Biological trait analysis (BTA) provides an objective measure of functional diversity and provides greater detail on ecosystem functioning than a taxon approach alone. BTA is resistant to large-scale biogeographic variation (Bremner *et al.* 2003) and can also usefully highlight the dominance of traits associated with adaptations to disturbed environments such as a short life span, fast growth rate, mobility and robust morphology (Tillin *et al.* 2006, Bremner *et al.* 2003). At sites around the Isle of Man, Lambert *et al.* (2011) found a negative relationship between wave stress and the maximum size of emergent, hard bodied and flexible organisms and the total biomass of emergent, colonial and flexible organisms. This may reflect impairment of feeding and larval settlement abilities with increased wave stress. The same study found a decrease in solitary, colonial and flexible organisms with increasing fishing frequency. In the present study, trait biomass was found to be similar at the majority of sample sites regardless of fishing intensity. Variation in biological traits was observed at lower levels of observed BSS; beyond a threshold of BSS (> *c.* 0.3 N m^{-2}) trait composition was similar across sample stations. Beyond this threshold trait composition is heavily weighted towards traits that enhance survival in disturbed habitats. The variation in trait composition at lower levels of BSS was attributable to a few rare species. The two most westerly sites (site 1 and 2) had less similar trait composition to other sites, this was due to a higher biomass of 'sessile', 'colonial' and 'attached' organisms and those with 'soft', 'cushion' or 'stalked' morphology. Such traits are common in less disturbed habitats. Site 2 had the lowest overall BSS and site 1 the third lowest BSS. This suggests that the functional composition of communities at the scale of the English Channel scallop fishery reflects spatial variation in levels of natural physical disturbance. It is also possible that repeated fishing disturbance over decadal timescales has altered community trait composition. Longer term effects of trawling in the North Sea indicate that scavenger and predator species are more abundant in recent times, possibly attributed to fishing activity over the last century (Rumohr & Kujawski 2000), although the ability to draw conclusions are limited due to differences in datasets and samples.

Management implications

Habitat forming taxa such as bryozoans (e.g. *P. fascialis)*, maerl and macroalgae provide substrate on which juvenile scallops (spat) can settle, as well as providing protection from predators, but these taxa suffer negative impacts from scallop dredging (Howarth *et al.* 2011, Bradshaw *et al.* 2001; Stokesbury & Harris 2006). Closed areas can provide benefits to the commercial scallop fishery by preserving habitat complexity to enhance spat settlement. Densities of scallops and the age and size structure of the population can increase both within the boundaries of a closed area, as well as on adjacent grounds that are fished, which can lead to a ten-fold increase in the exploitable biomass (Beukers-Stewart *et al.* 2005), an increase in the density of breeding adults (Stokesbury & Harris 2006) and increased reproductive output (Kaiser *et al.* 2007). Indirect effects of closure (e.g. trophic effects, such as a decline in prey species as the population of target species recovers) take on average 13.1 (± 2.0) years to become evident (Babcock *et al.* 2010). Three large closed areas, coupled with controlled, rotational opening of grounds successfully reformed the *Placopecten magellanicus* (sea scallop) fishery of Georges Bank (north-western Atlantic) from near depletion in the early 1990's, to become the most valuable fishery in the United States at the present time. This provides evidence that management incorporating systematic, prolonged (6-8 years) closure of areas adjacent to exploited grounds can maximise yield and profit, while also providing ecosystem benefits (Valderrama & Anderson 2006). The present study has not identified any effect of scallop fishing intensity on the habitats and communities present. All grounds surveyed had been fished at least once in the previous three years. Therefore, long-term (>5-8 years) or permanent closures may be more beneficial than the current cyclical regime of harvesting if an improved status of the seabed was a desirable outcome (Murwaski *et al*. 2000; Blyth *et al*. 2004; Howarth *et al*. 2014).

Due to the consequences of scallop dredging on the seabed a future management strategy might also retain the fishery within the current spatial extent to negate any damage to grounds that are outside of the boundaries of the current fishery. Closure of areas that have experienced historically low fishing pressure would enable relatively large areas of the seabed to recover, whilst minimising the economic impact to the scallop industry. The potential impacts of such ground closures include displacement of effort to the core fishing grounds (therefore increasing fishing mortality in these zones) and the lack of ability to utilise patches of scallop settlement outside of traditional areas, and respond to changing oceanic conditions. However, closures will only benefit communities in areas where natural disturbance is lower than that caused by fishing gear. Recent storm events in the UK have highlighted that in certain coastal areas of the UK such as Cardigan Bay and Lyme Bay this is not likely to be the case (Lambert *et al*. 2014b).

Conclusions

The impact of specific fishing gears depends on the environmental regime of the habitat in question. The present study was conducted on moderately dynamic, temperate sand and gravel habitats that have experienced prolonged (>40 years) periods of scallop dredging. This historic fishing activity could have altered the habitats and shaped the benthic communities within the boundaries of the fishery (Bradshaw *et al*. 2002; Garcia *et al*. 2006), resulting in communities resilient to fishing disturbance, although we are unable to demonstrate this in the present study. No effect of recent scallop dredging intensity on species diversity or composition was demonstrated (total scallop dredging activity ranged from 2 to 93 hours over a 3 year period at the sites sampled). The statistical results of this study imply that natural physical disturbance may exert a greater influence on species richness and community composition than fishing pressure. Studies of fishing impacts in other areas (Bradshaw *et al*. 2002; Blyth *et al.* 2004; Sciberras *et al*. 2013) have had the advantage of adjacent areas that have been closed to towed bottom gears for a period of time, enabling comparison with fished areas. Hence, forthcoming Marine Conservation Zones (MCZs) in the English Channel that prevent the use of bottom-towed fishing gears, will enable more precise evaluation of the effects of scallop dredging, irrespective of the temporal and spatial patterns of fishing (e.g. Strain *et al*. 2012)

CHAPTER 5: BYCATCH COMPOSITION OF THE ENGLISH CHANNEL KING SCALLOP DREDGE FISHERY

MSC data requirements addressed:

P2	Data requirements
Impact on Ecosystem	Bycatch species are within biologically based limits and management strategy must ensure that the fishery does not pose a risk of serious or irreversible harm to retained/bycatch species. Information allows estimation of mortality rate, injuries and the impact of fishing on endangered, threatened or protected (ETP) species.

Abstract

The issues of bycatch and discards are at the forefront of concern in modern fisheries management. For a fishery to attain sustainability credentials, scientific evidence must show that impacts on non-target species and undersized individuals of the target species are reduced as much as possible. The biomass and composition of bycatch from the English Channel scallop dredge fishery was evaluated and compared to bycatch composition at scallop fishing grounds in Wales and the Isle of Man. Bycatch composition varied significantly at localised and broad geographic scales. This is partly explained by differences in environmental parameters such as depth and seabed temperature, although seasonal variation in abundances and catchability of certain species are also likely to be important factors. Overall, bycatch in the English Channel is relatively low compared to other mobile fishing gears, with a mean of 19 % of total catch biomass. The proportion of scallop dredge bycatch biomass in Cardigan Bay is similar (15 %), whereas bycatch biomass is notably higher in the Isle of Man (53 %). These differences are likely due to environmental conditions that shape the habitat and species composition at the seabed. The main bycatch species of the fishery in the English Channel are: the queen scallop, *A. opercularis*; commercially fished crustaceans, the spiny spider crab, *M. squinado* and brown crab, *C. pagarus*; three species of starfish; monkfish, *L. piscatorius*; and the common cuttlefish, *S. oficinalis*. Due to the low biomass of individual species there is likely to be little impact on populations of commercially important species. However, management plans for the scallop fishery should give consideration to populations of monkfish, brown crab and turbot, *S. maximus* in Falmouth Bay, and the thornback ray, *Raja clavata*, and cuttlefish in the Baie de Seine. The discard rate of finfish and commercial shellfish in the English Channel is between 18-100 %, depending on location. Levels of discarding are higher in the eastern English Channel than the western English Channel. Impacts on species that come into contact with, but are not retained by the dredge are discussed along with potential methods for mitigation of the impacts of the scallop dredge fishery on bycatch organisms.

5.1 Introduction

Bycatch (the total catch of unwanted or non-target species) and discards (the proportion of living organisms from a catch returned to the sea) are two of the most prominent issues currently under scrutiny in global fisheries management (Hall *et al.*, 2000; Kelleher, 2005). Most fishing gears are not completely selective to the target species. Therefore, non-target species are either retained as bycatch or returned to the sea as discards. Discarding occurs for a number of reasons: lack of commercial value; high-grading (retaining individuals of higher value only e.g. larger individuals of a species); practical reasons (e.g. lack of space or suitable facilities for storage of the catch on board, or availability of processing facilities at the landing port); lack of quota or the correct licence required to land the species. Individuals of the target species that are below the minimum legal landing size must also be discarded. For these reasons fish that are fit for human consumption are often discarded and it is this wasteful practice that has brought the issue of bycatch and discards into the political and media arena in Europe in recent years (Hall *et al.*, 2000; Davies *et al.*, 2009; Heath *et al.*, 2014). Fisheries have the potential to impact not only the target species population but the populations of any other species that are retained by the fishing gear (Suuronen, 2005). This may have implications for ecosystem functioning (e.g. the removal of apex predators such as elasmobranchs or organisms that play a key role in benthic nutrient cycling), or may impact other commercial fisheries (Craven *et al.*, 2013). Slow growing and low productivity species are considered at greater risk (Southall *et al.*, 2011). In the 1990's, estimates of global bycatch were 35 % of the total catch weight (Alverson *et al.*, 1994). This has declined over recent years, due to improved gear design and increased management of fisheries. In relation to global discards, a recent weighted estimate was 8 % of total catch weight (Kelleher 2005), although some fisheries produce significantly higher discards than others (up to 62 % in some shrimp trawl fisheries). There is further incentive for fishermen in the European Union to reduce bycatch through the staged introduction of the landing obligation (discard ban) under the reformed CFP (Common Fisheries Policy) that commenced in January 2015. This is intended to make fishing more sustainable through reducing the capture of low-value species and encouraging the utilisation of retained biomass that would normally be discarded (Mangi & Catchpole, 2013).

Eco-labelling bodies such as the Marine Stewardship Council (MSC) specify requirements for the management of a fishery that consider and reduce impacts on bycatch species, whether retained or discarded. In a pre-assessment for the MSC, the English Channel king scallop dredge fishery was '*considered to have relatively low levels of incidental bycatch of other marine organisms*' (Southall *et al.*, 2011). There is considerable variation in bycatch among dredge fisheries globally. Bycatch from scallop dredges in Shetland is fairly high, at 37 % of catch biomass (Shelmerdine, 2010). Conversely, in the north western Atlantic, bycatch of the morphologically similar *Placopecten magellanicus* (sea scallop) dredge fishery on Georges Bank (fished using the larger 'New Bedford' dredge design) is estimated at just 6 % (DFO, 2007; DFO, 2008). Quantification of bycatch is fundamental to the implementation of EBFM (Ecosystem Based Fisheries Management) (Link, 2002), also known as EAFM (Ecosystem Approach to Fisheries Management). This approach has the goal of maintaining the entire ecosystem in a healthy and productive state such that eco-system over-fishing does not occur and thereby preserves trophic interactions (Hilborn, 2011).

In ICES area VII (encompassing the English Channel, Celtic Sea, Irish Sea and Western Approaches), discards of fish and cephalopods from commercial fishing in the early 2000s were estimated at 63 % by number of those caught (35 % by weight), with 90 % of those discards attributable to otter and beam trawlers (Enever *et al.*, 2007). Improvements in gear selectivity in recent years (e.g. Campbell *et al.*, 2010; Ingólfsson 2011; Kynoch *et al.*, 2011; Gilman 2011) mean that the proportion of discards is likely to have decreased. Government initiatives such as Project 50% (Nelson, 2009) have also helped to identify fishers motivations for discarding and potential barriers and disincentives to altering fishing behaviour or adopting gear improvements. While the findings indicate that scallop dredging is associated with relatively few discards in this location, this supposition remains unsubstantiated.

Impacts of the dredge fishery on bycatch species

Organisms that are returned to the sea alive following retention in fishing gear may die from physical injuries obtained during the capture process, stress related symptoms or increased vulnerability to predation post release (van Beek *et al.,* 1990; Chopin & Arimoto, 1995; Jenkins *et al.,* 2001; Veale *et al.,* 2001; Depestele *et al.,* 2014). Susceptibility to capture and the survivability of bycatch species varies depending on their morphological and physiological traits. In the case of scallop dredges, damage can occur on contact with the dredge on the seabed and when inside the dredge bag due to abrasion from other organisms or debris. Stress or physiological impacts caused by emersion and sorting on deck can also prove fatal. Different species are susceptible to different parts of the capture process. For example, edible crabs sustain most damage from the dredge teeth, while starfish incur most damage inside the dredge bag (Jenkins *et al.,* 2001). The catch efficiency of dredges for bycatch species is low and thus damaged individuals can remain on the seabed (Gaspar *et al.*, 2003). The resulting carrion and fishery discards provide a food resource and can alter the prevalence of scavengers in an area (Kaiser & Spencer, 1994; Ramsay *et al.*, 1996, 1998; Link & Almeida, 2002).

Scallop fisheries around the UK

In the UK, the main king scallop fisheries occur across the English Channel, in Cardigan Bay, (Wales), around the Isle of Man, around the Channel Islands, the west and east coasts of Scotland and offshore from Scarborough in the North Sea. In the territorial waters around the Isle of Man, king and queen scallops are the two most valuable commercially fished species, and contribute the majority of total fisheries landings value into the Isle of Man (60 % value attributable to king scallops; 20 % attributable to queen scallops, data from the Department of Environment, Food and Aquaculture, Isle of Man Government). In the UK, king scallops are fished using Newhaven or N-Viro™ dredges and queen scallops are fished with queen scallop dredges or otter trawls. Bycatch composition (invertebrates, fish and elasmobranchs) from the queen scallop otter trawl fishery is low (on average 7 % of total catch weight) and species composition varies with geographic location around the Isle of Man; although species diversity of bycatch is similar. South of the island, bycatch is dominated by fish (45 %), followed by elasmobranchs and invertebrates (Boyle & Thompson 2012). In contrast, in other areas bycatch biomass in the queen scallop fishery is dominated by invertebrates, followed by elasmobranchs and fish. Significantly higher bycatch biomass occurs on the east coast compared to the west coast (Boyle, 2012). In Wales, scallops are currently the second most valuable commercially fished species (Lambert *et al.*, 2014a). Fishing effort is controlled through stringent restrictions on the total number of dredges permitted per vessel: up to 6 within 3 NM (nautical miles) of the coast; up to 8 dredges from 3-6 NM; and 14 dredges up to 12 NM. The main king scallop fishery is located in Cardigan Bay. King scallops also occur in small numbers north of the Llyn Peninsula and in Liverpool Bay, to the north east of Anglesey; however seldom in quantities that produce a viable fishery (Mark Roberts, *FV* Harmoni, pers. comm). Queen scallops are highly abundant in Liverpool Bay (over 20 individuals per 100m^2) but are low in abundance in Cardigan Bay and the Llyn Peninsula (<3 individuals per 100m^2) (Lambert *et al.*, 2014b,c).

English Channel king scallop fishery

The English Channel scallop fleet can be defined by four broad categories: a) <15 m vessels that fish for scallops within 6 NM of the coastline on a year-round basis; b) <15 m vessels that fish for scallops within 6 NM of the coastline on a seasonal or part-time basis, targeting other species at different times of the year; c) >15 m vessels that fish for scallops outside 6 NM on a year-round basis; d) beam-trawlers that convert to scallop dredge gear at certain times of year in order to preserve quota, or if scallop landings are considered to be more profitable (Richard Caslake, Seafish, pers. comm). Therefore, many vessels that target scallops do so only at certain times of the year (targeting other species when it becomes more profitable to do so), and in certain areas, depending on the seasonal variation in condition (reproductive state) of king scallops. For example, there is little inshore scallop fishing activity in the eastern English Channel between March and November (see Chapter 2), but offshore grounds in the eastern English Channel have been fished year-round in recent years. In contrast, in the western English Channel, the inshore fleet operates on a year-round basis with localised seasonal patterns of movement (see Chapter 2). The location of vessels is also dictated by management regulations and voluntary agreements. For example in 2013 and 2014, UK fishermen agreed to a temporary closure of ICES sub-area VIId to vessels >15 m LOA (Length overall) in return for a reallocation of Western Waters scalloping effort (kW days) from France. Seasonal closures also occur on inshore (<6 NM) grounds, implemented by local Inshore Fisheries and Conservation Authorities (IFCAs). These spatial and temporal patterns in scallop fishing have logistical implications for a programme of bycatch sampling and overall impacts on bycatch species populations.

On some scallop vessels, scallops are the only species retained; however high value species such as Dover sole (*Solea solea*), monkfish (*Lophius piscatorius*) and other flatfishes may also be retained (personal observation). Vessels are entitled to land bycatch species up to a limit of 5 % of the total catch weight of scallops. Any further landings are subject to the vessel holding the relevant quota. Total fishing mortality of these commercially important species is therefore a combination of the effects of the target fisheries for these species as well as bycatch from the scallop fishery and unobserved mortality from contact with the gear on the seabed. The configuration of scallop fishing gear can be optimised for the seabed and conditions in which fishing occurs. Increasing tension on the sprung dredge teeth can reduce the capture of heavy rocks and increasing belly ring size reduces the likelihood of catching flatfish in habitats where these occur (M. Rogers, *FV Amy R*, pers. comm.). Increasing belly ring size should also mean that only the largest scallops are retained, which are more valuable if sold at auction.

MSC requirements

Under the framework of the MSC assessment, any species other than the target species present in the catches of the fishery/gear combination in question is considered to be either a ‘main’ or ‘minor’ bycatch species. A species is considered a ‘main’ species if it comprises 5 % or more by weight of the total catch of all species (MSC, 2014). For species classified as ‘less resilient’ this threshold is lowered to 2 % of the total catch. Species are not considered a ‘main’ species if they are returned to the sea alive; however good evidence of post-capture survival is required. Endangered, threatened or protected (ETP) species must also be considered if they are present in bycatches. For the UK, such species are listed under the 'UK Post-2010 Biodiversity Framework' (JNCC & DEFRA, 2012), or under CITES (the Convention on International Trade in Endangered Species of Wild Fauna and Flora). Thus, the MSC framework provides a useful basis by which to define which parameters are most important to quantify, such that the sustainability of a fishery may be assessed.

Specific objectives were to:

a) Identify all bycatch species including ‘main’ bycatch species (those contributing > 5 % of catch weight) and ETP species that occur in the English Channel king scallop fishery.
b) Compare scallop dredge bycatch in the English Channel with bycatch from scallop fisheries across ICES area VII, in the Irish and Celtic Seas.

Given the spatial and temporal variability of the scallop fleet imposed on the sampling design in this study, the main focus is to report bycatch at broad geographic scales that encompasses data from a number of different areas across the English Channel and from different years. This approach also enables a comparison of the English Channel data with other important king scallop fisheries in the UK.

5.2 Methods

5.2.1 Sampling

Sampling occurred between June 2012 and June 2013. Ten sampling trips were conducted on board eight commercial scallop vessels. The aim was to sample a range of fishing grounds across the English Channel to encompass the environmental variation described in Chapter 4, however exact sampling locations were dictated by where skippers were fishing when sampling occurred. For example, no scallop fishing occurs in the inshore eastern English Channel between March and December and weather has a large influence on choice of fishing grounds (see Chapter 2). The total number of dredges used on the vessels varied between 10 and 34, depending on the size of the vessel. Samples were taken during normal scallop dredging activity. Trips were arranged to include the main scallop fishing grounds that were identified from Vessel Monitoring System (VMS) data and semi-structured questionnaires undertaken with skippers of scallop vessels. The following information was recorded for each haul sampled: co-ordinates at the start of each tow (the moment the gear made contact with the seabed following deployment) taken from the vessel GPS system, water depth (m) at the start of the tow, average speed of tow (knots), duration of tow (minutes) and co-ordinates at the time of gear retrieval (when the skipper began to winch the gear from the seabed). For each haul that came on deck the full contents of one or two dredges were retained for detailed sampling. For each haul, different dredges were selected for sampling (e.g. alternating between port and starboard dredges, and from bow to stern). On larger vessels this was not possible due to safety and logistical reasons. On such vessels, the crew separated the contents of the first dredge (stern end) from the conveyor belt and moved this to one side before sorting of the sample commenced. The dredge contents were recorded as follows: volume of large rocks and broken shell (measured as the proportion of the volume of a 5 stone fish basket, using a calibrated measuring stick). All scallops from the sampled dredge(s) were counted and their shell width (distance from anterior to posterior shell margin) measured to the nearest mm. Shell width was measured as opposed to shell height (distance from umbo to ventral shell margin) as this is how the crew sort the catch, which meant that the sorted scallops could be added to the retained catch after measurement. All remaining organisms from the dredge sample (e.g. sea urchins, crustaceans, starfish and non-commercial fish species) were identified and body length and/or count of each individual/species was recorded. For species where body length measurements were impractical (e.g. starfish, dead men's fingers) only count data were recorded. In total 99 hauls were sampled. All commercial fish, molluscs and crustacean species from each haul (all dredges) were counted and measured. It was also noted whether these species were retained or discarded.

To supplement the sampling data, additional data were obtained from the Centre for Environment, Fisheries and Aquaculture Science (CEFAS) for scallop observer trips that occurred between September 2011 and October 2012. Species for which length measurements were recorded during CEFAS observer trips were commercial finfish species and non-quota commercial shellfish species such as king scallops, lobster and whelks (see Appendix 5.1 for a complete list). During observer trips, a sub-sample (e.g. a single dredge) is taken from a haul and the data raised to the total number of dredges. Smaller benthic species (such as sea-urchins, starfish and small crustaceans) and non-commercial fish species were combined with any substrate (rock, broken shell, sand etc.) from the dredge sample and a total volume recorded as 'benthos'. However, 'benthos' is not consistently recorded across all observer trips and therefore the total catch weight could not be estimated. Therefore, all records of 'benthos' were removed from the CEFAS dataset and the CEFAS data were only in the analysis of bycatches of finfish and commercial shellfish species. Hauls that included records of species with no quantification (recorded only as 'observed') were also removed from the dataset. In total, data recorded from 308 hauls from 24 separate observer trips was retained for analysis. For a limited number of tows where no speed was recorded, a speed of 3 knots was assigned as this was the mean fishing speed observed across all sampling trips (see also Lee *et al*. 2010). The locations of all sampling trips are shown in Figure 5.1.

5.2.2 Data Analysis

Published data on standard length/weight relationships (Appendix 5.3) was used to calculate the total biomass of each species for which a length measurement was taken. Tow length was calculated by multiplying the duration of the tow by the average speed recorded for the tow. Area swept was calculated as the total width of the dredges multiplied by tow length. The total biomass of each species per tow was then calculated, by raising the biomass recorded to the total number of dredges (if from a sub-sample) and all values were standardised to kg km^{-2}. For the species for which only the count data were available, the mean weight of an individual was calculated from data collected during scientific surveys (see chapter 4). Total biomass per tow was estimated using these values (Appendix 5.4), and then standardised to kg km^{-2}. As each trip occurred in one localised area of the seabed, the mean biomass of each species retained per trip was calculated by pooling the data from all hauls per trip. These values were used to ascertain the proportion of the catch weight that was contributed by each species.

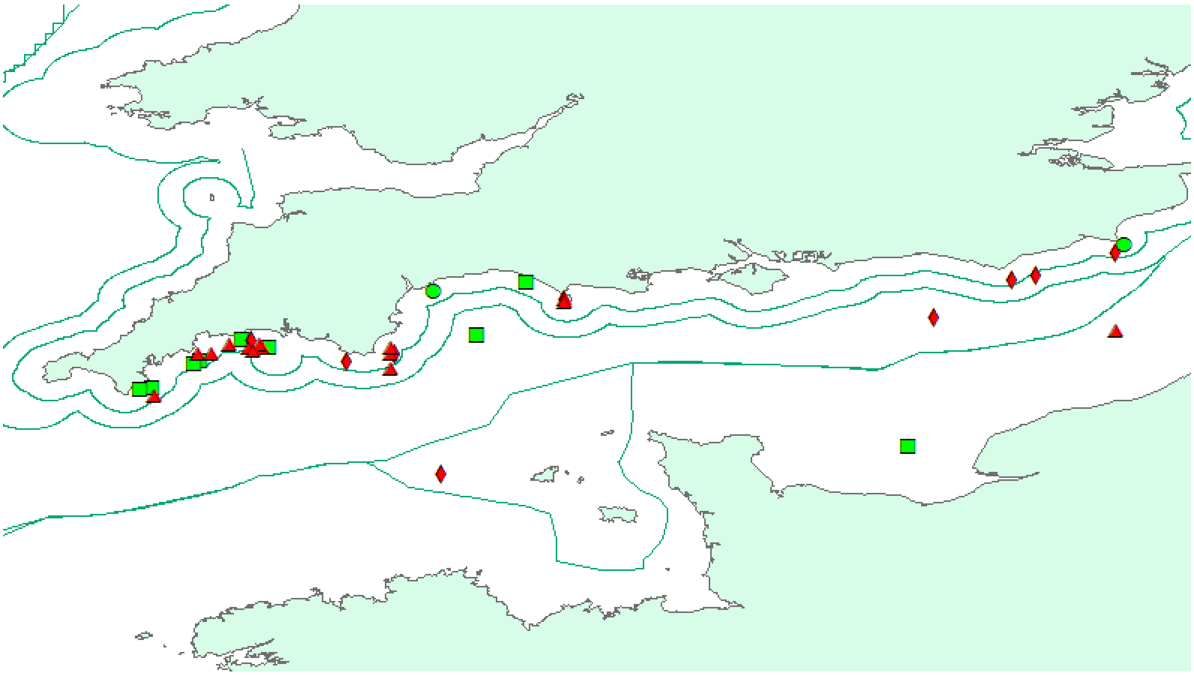

Figure 5.1: Location of the 34 sites sampled for bycatch in the English Channel. Sites sampled by the author are indicated by green squares (spring/summer) and green circles (autumn/winter). Sites from CEFAS observer trips are indicated by red triangles (spring/summer) and red diamonds (autumn/winter). The 6 and 12 NM limits are shown along the English coast, as well as the boundary between English/French territorial waters.

CEFAS data included measurements of *P. maximus* shell height whereas *P. maximus* shell width was recorded during sampling trips. In order to calculate size-weight (total wet weight) relationships for *P. maximus* the exponential relationship of weight with shell height and shell width was determined using data on king scallop size-weight relationships gathered during scientific surveys in the English Channel (see Chapter 4). Geographic differences in growth rates and allometry occur for *P. maximus* (Chauvaud *et al.*, 2012; G. Campbell, unpubl. data) Therefore, separate equations were determined for ICES sub-areas VIId and VIIe. The relationship between *P. maximus* shell width and total wet weight in sub-area VIIe (western English Channel) is described by the equation: y = 0.0003 $L^{2.8178}$ (R^2 = 0.96) and in sub-area VIId (eastern English Channel) by the equation y = 0.0006 $L^{2.6183}$ (R^2=0.88). The relationship between *P. maximus* shell width and total wet weight in area VIIe is described by the equation: y = 0.0002 $L^{2.9676}$ (R^2 = 0.95) (n=411) and in area VIId by the equation y = 0.0004 $L^{2.7724}$ (R^2 = 0.89) (n=502).

Environmental variables

Non-parametric multivariate analyses of the environmental and community assemblage data were performed in PRIMER v.6 (Clarke & Gorley, 2006). Results from elsewhere (Chapter 4) indicated that the variables tidal bed shear stress, depth, mean sea bed temperature (T_{mean}) and interannual temperature range (T_{range}) explained most of the environmental variation between the scallop fishing grounds in the English Channel. Values for these four parameters were extracted for the location of each sampling trip, including the CEFAS data (see Chapter 4 for data sources) and a draftsman plot was used to identify significant correlations between each pair of variables. T_{mean} was correlated with T_{range} (ρ=0.86), but not at the level (ρ=0.95), hence both variables were retained in the analysis (Clarke & Warwick, 2001). The dataset was normalised and a resemblance matrix was produced using Euclidean distance. To identify environmentally distinct regions a SIMPROF analysis identified significant groupings of sites (all samples from the same trip were grouped as a site) based on similarity of their environmental variables, at a significance level of 5 %. A Principal Component Analysis (PCA) was performed to establish which of the environmental variables explained the greatest variation among sites. The BIOENV procedure was used to investigate which environmental variables gave the highest correlation with bycatch species composition.

Catch composition

The bycatch species dataset was aggregated to genus level and square-root transformed to down-weight the influence of highly abundant or rare taxa. A resemblance matrix was created and used to generate a MDS (multi-dimensional scaling) plot to visualise clusters of sample sites based on their similarity in bycatch species composition. ANOSIM tests were used to ascertain whether samples grouped by the similarity in environmental parameters, season ('winter': October to March, or 'summer': April to September), or sample trip, had significantly different species composition. A SIMPER analysis was used to identify indicator species for the separate groups of sites identified form the analysis of environmental variables.

Finfish and commercial shellfish species

The total number of species in the CEFAS observer data was 45 (restricted to finfish and commercially important shellfish species). To enable a comparison of CEFAS observer data with sample data collected in the present study, the latter was constrained to the species recorded in the CEFAS data. Two additional species (considered to be commercial finfish) occurred in the survey dataset but not the CEFAS data were the red gurnard, *Chelidonichthys cuculus* and the Norway pout, *Trisopterus esmarkii*. The mean biomass of each species per trip (kg km^{-2}) was used to compare the species composition across the pooled data set, by sampling trip. An MDS plot was used to visualise groupings of sites based on their similarity in bycatch species composition and ANOSIM was used to test for significant differences in the species composition of finfish and commercial shellfish between groupings of sites based on similarity in their environmental parameters.

Spatial variation in bycatch

The observed patterns in bycatch were compared with bycatches in scallop fisheries in Wales and the Isle of Man. King scallop dredge bycatch data from Cardigan Bay, Wales and Manx (Isle of Man) territorial waters were obtained from the Fisheries and Conservation Science Group, Bangor University. The dataset encompassed 20 survey sites across the main king scallop fishing grounds in the Isle of Man that were identified from a high frequency of VMS records (Shepperson *et al.*, 2014). In Cardigan Bay, face-to-face consultation with local fishermen (M. Roberts, *FV Harmoni*, pers. comm) identified important king scallop fishing grounds and data from 57 sample sites were used in analysis. Data from the Isle of Man was for the period May 2012 to February 2013; and data from Cardigan Bay was collected between June 2012 and August 2014. If the same site was sampled in >1 year, the mean biomass values for that site were used. Information on tow length and area swept (total width of the dredges used) was used to calculate biomass of king scallops and bycatch species, standardised to kg km^{-2}. The methods described above were used to investigate species composition in PRIMER. Only one tow was conducted at each site, therefore a single value of biomass was used for each site, as opposed to mean values calculated for sites in the English Channel, where data from multiple tows were available for each site. The number of sample sites, sampling approach and analyses performed in each area are summarised in table 5.1.

Table 5.1: Summary of the number of sites sampled, sampling approach and the data analyses performed for each area included in the study.

Location	number of sites sampled	Sampling approach	Data analyses
English Channel (author sampling)	10	Sub-sample of entire catch	Species diversity & composition, correlation with environmental variables, discards
English Channel (CEFAS data)	14	Sub-sample of finfish and commercial species only	Species diversity & composition, correlation with environmental variables, discards
Cardigan Bay, Wales	57	Sub-sample of entire catch	Species diversity & species composition
Isle of Man	20	Sub-sample of entire catch	Species diversity & species composition

5.4 Results

5.4.1. Bycatch in the English Channel

English Channel environmental variables

The PCA analysis indicated that PCA1 explained 64 % of the environmental variation between sample sites in the English Channel and PC2 a further 26 %. PC1 is composed of a similarly weighted combination of T_{range} and T_{mean} in one direction and depth in the opposite direction. PC2 was mainly influenced by BSS. A SIMPROF test revealed three environmentally distinct groups of sample sites at the p=0.05 level. The first group (referred to as 'Shallow') included the four shallowest sites (two in Lyme Bay and two in the eastern English Channel), the second group (referred to as 'Far west') the two most westerly sites and the third group (referred to as 'West') the remaining four sites in the western English Channel (Table 5.2, Figure 5.2).

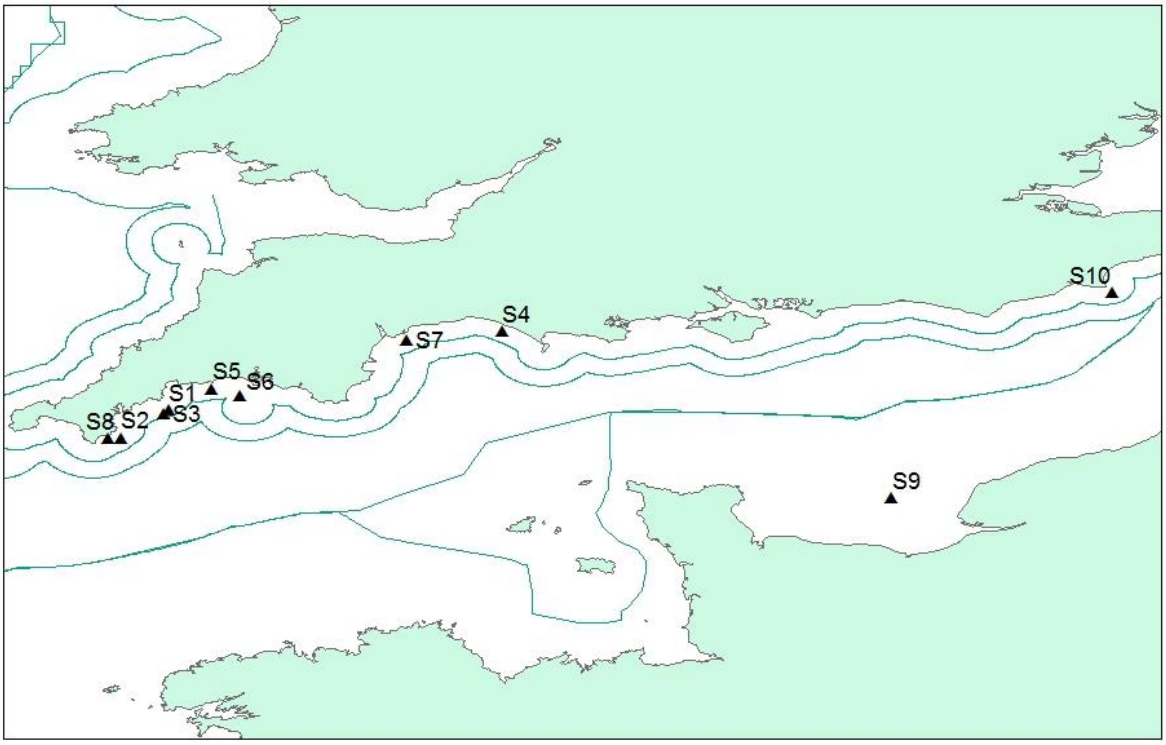

Figure 5.2: Location of the 10 sites sampled for scallop dredge bycatch by the author in the English Channel. Sampling took place on board commercial scallop vessels between June 2012 and June 2013.

Table 5.2: Groups of sample sites in the English Channel based on their similarity in four environmental parameters, identified by a SIMPROF analysis. Groups are significantly different at the p=0.05 level.

Site	SIMPROF group	ICES sub-area	bed shear stress (N m^{-2})	Mean seabed temperature (ºC)	mean temperature range (ºC)	depth (m)
S4	Shallow	VIIe	0.49	12.06	10.37	25
S7	Shallow	VIIe	0.13	12.04	10.36	16
S9	Shallow	VIId	0.92	12.30	10.76	26
S10	Shallow	VIId	0.42	11.83	11.64	29
S2	Far west	VIIe	0.62	10.69	7.79	70
S8	Far west	VIIe	0.80	10.69	7.79	32
S1	West	VIIe	0.11	11.34	8.28	58
S3	West	VIIe	0.12	11.24	8.27	60
S5	West	VIIe	0.08	11.50	8.67	45
S6	West	VIIe	0.12	11.50	8.80	49

English Channel catch composition

Inert material (broken shells, rock, sand, gravel) dominated the weight of catches, with a mean proportion of 75-92 % of the total weight (see appendix 5.5 for individual site values). *P. maximus* contributed 6-20 % of the total catch weight and bycatch varied from <1 % to 8 % of the total weight of the catch. Of the living biomass retained by the dredges, bycatch species contributed between 8 and 37 % to the catch weight, depending on the location, with a mean of 19 % across all trips. The highest proportion of bycatch occurred in the east of Lyme Bay. The proportion of bycatch between the three habitat groups was similar (ANOVA: $F_{2, 7}$=0.237 p=0.80) (Figure 5.3). The mean number of species retained per tow across all trips was 10.1 (±3.8) (Table 5.3).

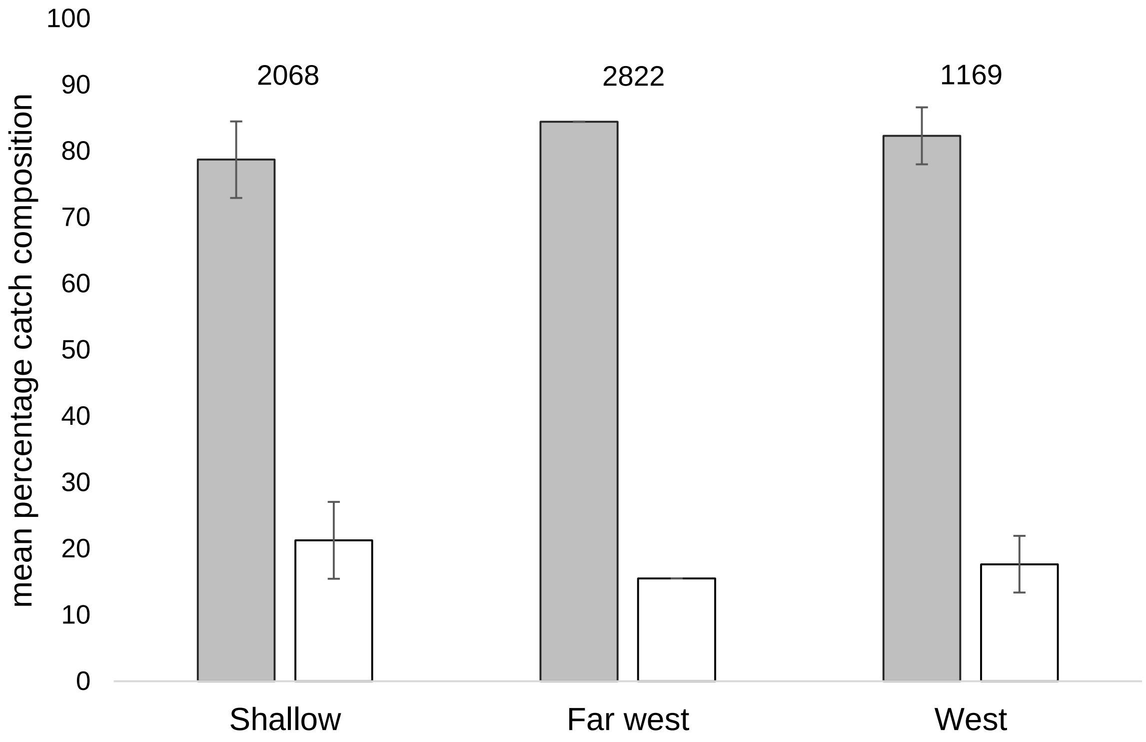

Figure 5.3: The percentage composition (biomass) (±S.E.) of *P. maximus* (grey bars) and bycatch species (white bars) in scallop dredge catches from three groups of sample sites in the English Channel (Shallow; Far west; West). The Far west group contains only two sites; therefore calculation of standard error was not possible. Numbers above the bars represent the mean total biomass of catches in each group (kg km^{-2}).

Table 5.3: Mean total number of species and total biomass (of *P. maximus* and all bycatch species) from each survey trip. Mean and standard error values (S.E.) are given.

		No. of species		Total biomass (kg m^{-2})	
Group	**Trip**	**Mean**	**S.E.**	**Mean**	**S.E.**
Shallow	S4	10.1	0.8	2818.4	253.1
Shallow	S7	7.9	0.7	1828.6	194.4
Shallow	S9	13.8	1.1	1585.2	195.4
Shallow	S10	9.2	0.8	1521.3	74.4
Far west	S2	4.2	0.8	1973.2	221.2
Far west	S8	12.8	0.9	3617.2	173.9
West	S1	7.3	1.1	1255.2	92.8
West	S3	7.3	0.8	1534.9	142.4
West	S5	17.2	0.6	1352.0	75.0
West	S6	11.0	1.8	461.1	32.7

Of the 74 taxa (65 at species level and 9 at genus level, Appendix 5.6) identified across all sampling trips, *P. maximus* was the dominant species and accounted for a mean of 81 % of total catch weight, while a further 16 species contributed to the top 99 % of the total mean biomass across sites (Table 5.4, Appendix 5.6). The queen scallop, *A. opercularis* occurred with the second highest mean biomass, and was the only species that constituted on average >5 % of the total catch weight across all sampling trips (Table 5.5). Other species that contributed >5 % to the total catch weight at one site or more are listed in Table 5.5. The commercial species brown crab, monkfish, cuttlefish and turbot each contributed > 5 % of the catch biomass at one sample site.

Table 5.4: Mean biomass of the species that contributed to the top 99 % of biomass caught across all trips. Species of commercial importance in the English Channel are indicated by bold text. Cum.% = cumulative percentage of bycatch. No. sites = number of sites at which this species occurred.

species	**common name**	**no. sites**	**mean biomass (kg km^{-2})**	**mean % of catch**	**cum. %**
Pecten maximus	**king scallop**	10	1476.3	81.0	81.0
Aequipecten opercularis	**queen scallop**	8	130.2	6.1	87.1
Marthasterias glacialis	spiny starfish	7	83.0	3.5	90.6
Maja squinado	**spiny spider crab**	8	27.0	1.4	92.0
Sepia officinalis	**cuttlefish**	5	26.3	1.3	93.3
Cancer pagurus	**brown crab**	10	16.0	1.1	94.4
Lophius piscatorius	**monkfish**	7	15.8	1.0	95.4
Asterias rubens	common starfish	6	20.7	1.0	96.4
Luidia ciliaris	seven-armed starfish	7	13.7	0.8	97.3
Buccinum undatum	**common whelk**	6	6.7	0.3	97.6
Ostrea edulis	**common flat oyster**	1	5.4	0.3	97.9
Raja clavata	**thornback ray**	4	5.0	0.2	98.1
Solea solea	**Dover sole**	8	3.2	0.2	98.3
Scyliorhinus canicula	small spotted catshark	7	3.5	0.2	98.5
Scophthalmus maximus	**turbot**	2	2.7	0.2	98.7
Pleuronectes platessa	**plaice**	6	2.4	0.2	98.8
Echinus esculentus	common sea urchin	6	1.8	0.1	99.0

Table 5.5: Species for which mean biomass contributed >5 % of the total living biomass in scallop dredge catches during at least one sample trip, from 10 sample trips in the eastern and western English Channel (S1-S10). Numbers represent the percentage contribution to the overall catch biomass and those >5 % are highlighted in bold. Species of commercial importance in the English Channel are indicated by bold text. S.E.=standard error.

	Shallow				**Far west**		**West**					
Common name	**S4**	**S7**	**S9**	**S10**	**S2**	**S8**	**S1**	**S3**	**S5**	**S6**	**Mean**	**S.E.**
P. maximus	**55.0**	**79.2**	**70.3**	**82.0**	**72.6**	**72.3**	**76.4**	**83.4**	**66.4**	**73.6**	**73.1**	2.6
A. opercularis	**28.4**	0.0	2.6	1.4	0.3	0.0	3.4	2.2	**19.3**	2.5	**6.0**	3.1
M. glacialis	0.0	0.3	0.0	0.0	2.8	**17.0**	**5.8**	**6.0**	2.6	**7.2**	4.2	1.7
C. pagurus	0.5	1.3	0.1	0.9	**8.8**	0.8	4.8	0.5	1.8	3.2	2.3	0.9
L. piscatorius	0.3	0.0	0.0	0.0	**7.8**	0.5	2.2	2.3	0.6	4.2	1.8	0.8
S. officinalis	0.0	0.0	0.4	3.9	0.0	**7.8**	0.0	0.0	0.6	0.4	1.3	0.8
A. rubens	3.0	0.7	**5.2**	0.9	0.4	0.0	0.0	0.0	0.3	0.0	1.0	0.5
S. maximus	0.0	0.0	0.0	0.7	**6.6**	0.0	0.0	0.0	0.0	0.0	0.7	0.7

There was a significant difference in species composition between the three groups of sample sites (Shallow, Far West, West) identified in the SIMPROF analysis as having significant variation in environmental parameters (ANOSIM: R=0.632, p=0.001, Table 5.6, 5.7, Figure 5.4a). There was also a significant difference in species composition between sample sites (ANOSIM: R=0.883, p=0.001) and samples from the same site were generally clustered together on the MDS plot, with some overlap between sites occurring (Figure 5.4b). Species composition was significantly different between all pairs of sites except for the two sites with the closest proximity to each other (S1 and S3). There was no significant difference in bycatch species composition between season (R=0.016, p=0.38); however the lack of temporally repeated samples and the inherent variability between all sites mean this result should be interpreted with caution. A SIMPER analysis of the three groups identified as having significantly different environmental variables revealed that the within group similarity in bycatch species composition was 67, 64 and 64 % for the groups Shallow, Far west and West, respectively.

Bycatch species contributing to the top 95 % of biomass in the Shallow group were *A. opercularis*, *A. rubens*, *M. squinado*, *S. officinalis*, *C. fornicata* and *B. undatum* (Table 5.8). In the Far west group, species contributing to the top 95 % of biomass were *M. glacialis* and *L. piscatorius* (Table 5.8), although the Similarity/Standard Deviation (Sim/S.D.) values were <0.5 therefore the biomass of these species was not consistent across sites within the group. In the West group, *A. opercularis*, *M. glacialis*, *L. ciliaris*, *L. piscatorius*, *C. pagarus* and *M. squinado* dominated the top 95 % of biomass (Table 5.8). The Sim/SD values for all these species were <1.3 meaning that the variation in biomass of the species between sites within the group was high. A full table of the SIMPER output is in Appendix 5.7.

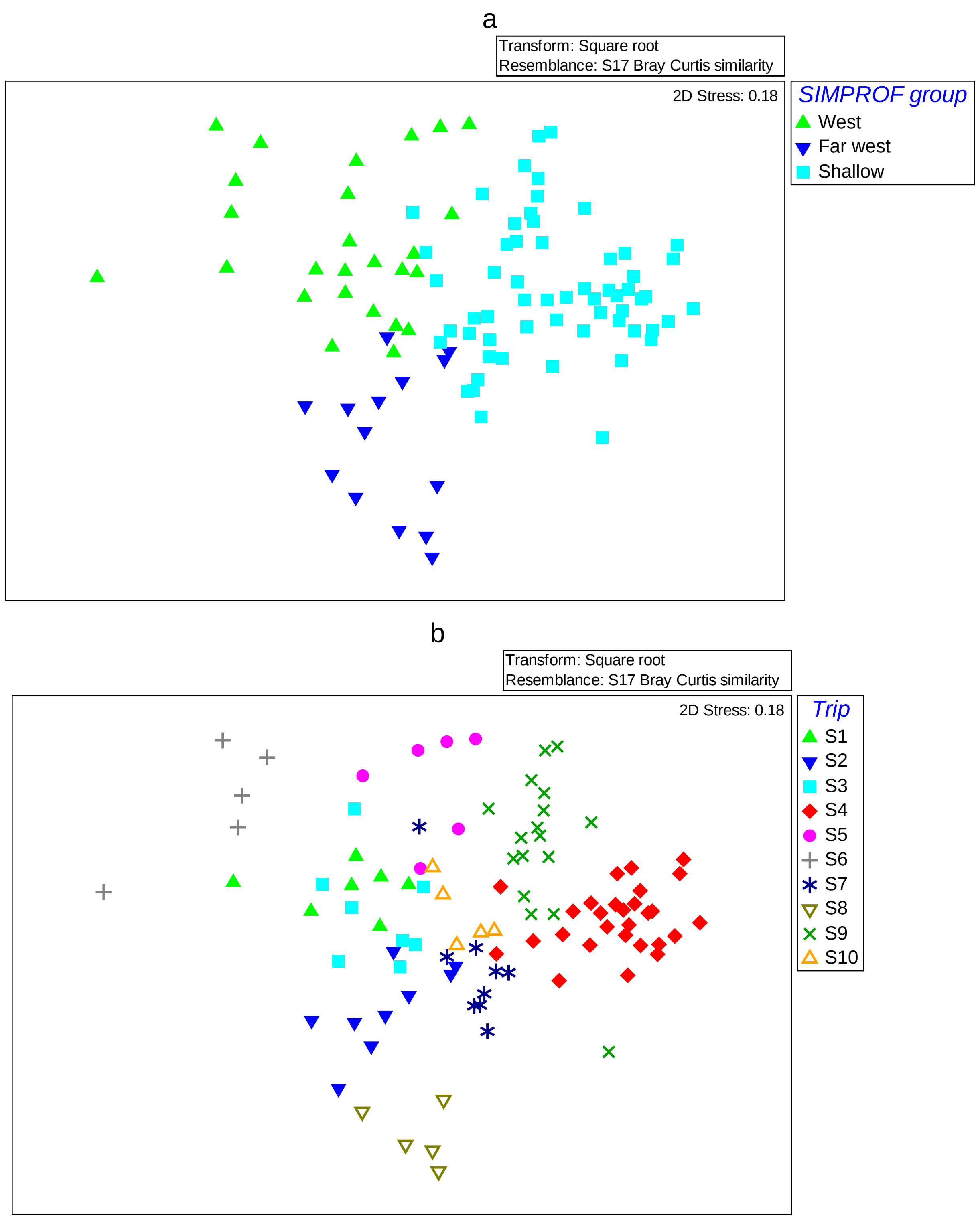

Figure 5.4: Multi-dimensional scaling plots of the similarity among sample sites based on bycatch species biomass (square-root transformed data) in scallop dredge catches in the English Channel. Each symbol represents a sampled haul from a single tow. a) symbols represent the three groups identified by SIMPROF analysis based on variation in environmental parameters at the sample sites; b) symbols represent sample trips (sites).

Table 5.6: The three groups of sites identified by significantly different environmental parameters in a SIMPROF analysis, and the location of the sites in the English Channel.

Group	site	Location
West	S4, S7, S9, S10	Eastern English Channel & Lyme Bay
Far west	S2, S8	Falmouth Bay (west)
Shallow	S1, S3, S5, S6	Falmouth Bay (east)

Table 5.7: Results from SIMPER and ANOSIM analysis for the dissimilarity in bycatch species composition between environmentally distinct groups of sites (Shallow; Far west; West).

Groups	Dissimilarity (%)	R statistic	p-value
West, Far west	47	0.668	0.001
West, shallow	47	0.649	0.001
Far west, Shallow	45	0.452	0.001

Table 5.8: Mean abundance (kg km^{-2}) of the species contributing to the top 95 % of bycatch biomass within the groups Shallow, West and Far West. Average within group similarities are shown in parentheses. Species of commercial importance are in bold text.

Shallow (66 % similarity)	mean biomass (kg per km-2)
A. opercularis	224.7
A. rubens	54.0
M. squinado	8.8
S. officinalis	5.3
B. undatum	5.2
C. fornicata	2.6
West (64 % similarity)	
M. glacialis	104.2
L. piscatorius	24.3
Far west (64 % similarity)	
A. opercularis	45.2
M. glacialis	22.5
L. ciliaris	12.3
L. piscatorius	5.8
C. pagurus	2.8
M. squinado	1.2

Endangered, Threatened or Protected species

There are 78 marine species classified under the 'UK Post-2010 Biodiversity Framework' (JNCC & DEFRA, 2012). Of these, four were recorded as bycatch in the present study (Table 5.9).

Table 5.9: Marine species classified under the 'UK Post-2010 Biodiversity Framework' (JNCC, 2014) or the ICUN red list that occurred in bycatch of the English Channel king scallop dredge fishery.

Species	Common name	Information
Lophius piscatorius	monkfish	Advice from ICES is based on the approach for data limited stocks. Evidence of medium recruitment from 2008-2012 and a recent increase in stock biomass. Uncertainties in growth estimates and ageing prevent accurate stock assessment and specification of stock reference points (Seafish, 2013)
Ostrea edulis	native oyster	Populations under threat and/or in decline in OSPAR region II (includes the English Channel). Has global and regional importance as a keystone species (OSPAR Commission, 2009)
Pleuronectes platessa	plaice	Stocks in the western English Channel are within biologically safe limits. Stocks in the eastern English Channel are in a period of recovery following depletion of stocks between 2003 and 2008. ICES recommend a reduction in discarding in this area (Seafish, 2013b).
Solea solea	Dover sole	Stocks are considered to be above safe biological limits for the English Channel. The Hastings sole trammel net, Dutch sole gill net and otter-trawl fisheries in the eastern English Channel are MSC certified (Seafish, 2013c).
Raja clavata	thornback ray	Stock abundance increased by an estimated 60 % between 2007 and 2013 (based on landings data), however the species is highly vulnerable to capture in demersal fisheries, matures at a late age and has low fecundity. Mortality from discards is unknown. Inadequate information available for stock assessment (McCully, 2013; Lart, 2014).

Monkfish were present in the bycatch at 7 of the 10 sample locations and contributed >5 % of total catch weight at site S2 (Falmouth Bay, western English Channel); however monkfish are only classified under the 'UK Post-2010 Biodiversity Framework' in the eastern English Channel (JNCC, 2014). None of the species caught as bycatch are classified under CITES (Convention on International Trade in Endangered Species of Wild Fauna and Flora). *Raja clavata* is classified as ‘near threatened’ on the International Union for Conservation of Nature (ICUN) red list (ICUN 2014). Eight of the recorded bycatch species are subject to TACs in the English Channel (Council of the EU, 2014). These are: thornback, undulate and cuckoo rays; Dover sole, megrim, plaice, turbot and monkfish. Of these species, turbot and monkfish were the only two species that accounted for >5 % of catches, at a single site respectively.

Finfish and commercial shellfish

The PCA analysis indicated that PC1 explained 70 % of the environmental variation between the combined author and CEFAS sample sites, and PC2 a further 21 %. PC1 is composed of a similarly weighted combination of T_{range} and T_{mean} in one direction and depth in the opposite direction. PC2 was mainly influenced by BSS. A CLUSTER analysis combined with SIMPROF testing revealed six groupings of sites based on significant differences in their environmental parameters at a significance level of 5 % (Table 5.10).

Table 5.10: Results from a CLUSTER analysis of the similarity in environmental parameters at all author and CEFAS sampling sites

Group	Sites	Location
FB_mid	C28, S1, S3	Falmouth Bay (mid)
FB_east	C10, C11, C12, C19, C25, C27, C7, S5, S6	Falmouth Bay (east)
SB_EC	C16, C17, C18, C20, C3, C22	Start Bay and mid-eastern English Channel
FB_west_WC	C23, C24, C4, S2, S8	Falmouth Bay (west), mid-western English Channel, Start Bay
LB_Portland	C13, C26, C14	Lyme Bay (Portland)
LB_EC	C1, C6, S10, S4, S7, S9, C2, C5	Lyme Bay and eastern English Channel

When considering finfish and commercial shellfish bycatch species only, using the combined survey and CEFAS datasets, the five bycatch species with the highest mean biomass across all sample sites were *A. opercularis*, *M. squinado*, *Lophius* sp., *S. officinalis* and *C. pagarus* (Table 5.11).

Table 5.11: The 20 finfish and commercial shellfish species with the highest mean biomass, from all sites sampled by the author and CEFAS data. Number of sites = number of sites sampled at which the species occurred.

Species	**mean catch biomass ($kg\ km^{-2}$)**	**number of sites**
Pecten maximus	1262.7	33
Aequipecten opercularis	40.3	10
Maja squinado	31.5	25
Lophius sp.	19.1	24
Sepia officinalis	17.4	10
Cancer pagurus	14.0	31
Pleuronectes platessa	12.0	22
Solea solea	4.3	26
Raja clavata	3.8	11
Microstomus kitt	2.7	23
Buccinum undatum	2.0	7
Ostrea edulis	1.8	3
Scophthalmus maximus	1.7	9
Raja brachyura	1.7	7
Scyliorhinus canicula	1.6	10
Trisopterus luscus	0.8	6
Scophthalmus rhombus	0.6	6
Raja montagui	0.6	3
Homarus gammarus	0.5	4
Lepidorhombus whiffiagonis	0.3	3

An ANOSIM revealed significant differences in the composition of finfish and commercial shellfish species between environmentally distinct groups (R=0.403, p=0.001). An MDS plot indicated that sites in the middle and eastern parts of Falmouth Bay (FB_mid; FB_east) had more similar species composition than other sites, and sites from Lyme Bay were clustered togther (LB_Portland; LB_EC) (Figure 5.5). The percentage dissimilarity in species composition between groups ranged between 3 and 48 % and species composition was significantly different between six pairs of environmentally distinct groups (Table 5.12). The BIOENV procedure identified that a combination of all four environmental variables best explained the variation in species composition between sites (ρ=0.368, p=0.001).

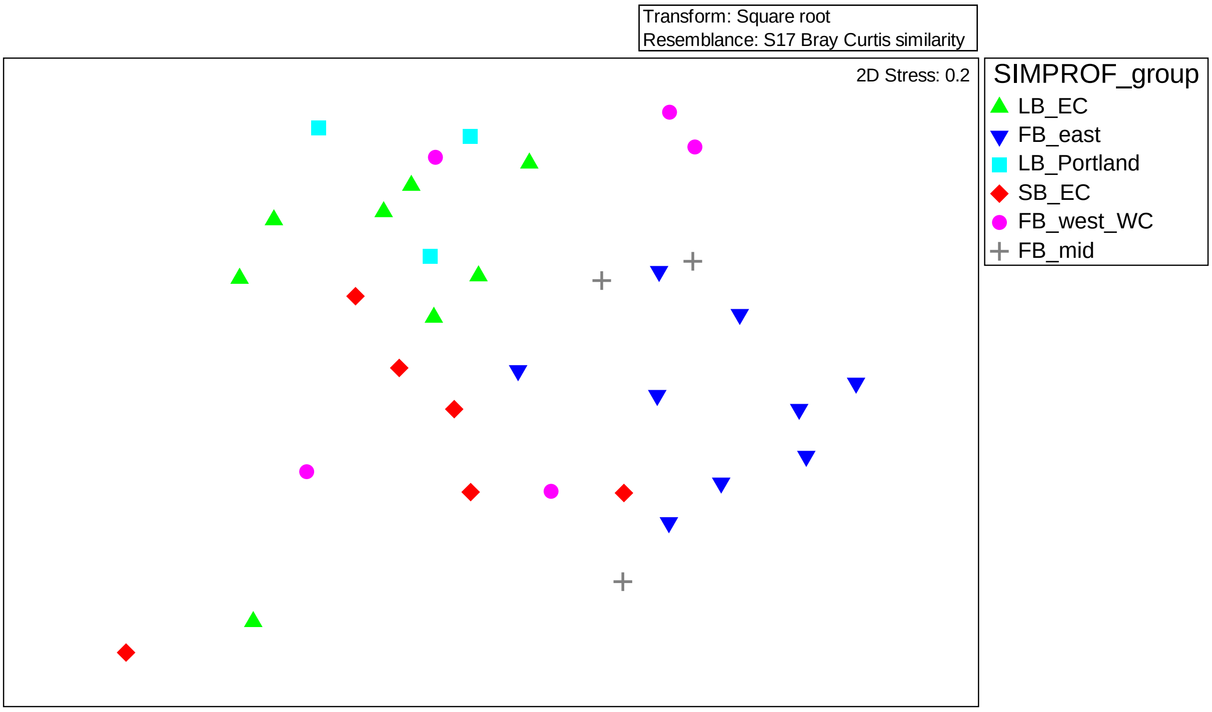

Figure 5.5: Multi-dimensional scaling plots of relative similarity in biomass of finfish and commercial shellfish species (square-root transformed data) in scallop dredge catches in the English Channel. Each symbol represents pooled data from one sample site. Symbols represent the groups of sample sites identified by a SIMPROF analysis based on the similarity in environmental parameters.

Table 5.12: Percentage dissimilarity in species composition of finfish and commercial shellfish bycatch species between groups A-F (groupings based on differences in environmental parameters at sites) and the R statistic for pairwise comparisons. Pairings for which an ANOSIM analysis revealed significant differences in species composition are highlighted in bold. ns = non-significant result.

Groups	Dissimilarity %	R statistic	p-value	
FB_EAST, LB_PORTLAND	**48**	**0.821**	**0.005**	
LB_EC, FB_EAST	**47**	**0.721**	**0.001**	
LB_PORTLAND, FB_MID	43	0.704	0.100	ns
FB_EAST, SB_EC	**45**	**0.553**	**0.005**	
FB_EAST, FB_WEST_WC	**43**	**0.509**	**0.008**	
LB_EC, FB_MID	45	0.460	0.055	ns
LB_PORTLAND, SB_EC	42	0.309	0.083	ns
LB_EC, FB_WEST_WC	**43**	**0.270**	**0.033**	
LB_EC, SB_EC	**41**	**0.264**	**0.027**	
SB_EC, FB_MID	41	0.179	0.202	ns
LB_EC, LB_PORTLAND	37	0.105	0.285	ns
FB_EAST, FB_MID	34	0.079	0.264	ns
SB_EC, FB_WEST_WC	41	0.016	0.413	ns
LB_PORTLAND, FB_WEST_WC	39	-0.015	0.482	ns
FB_WEST_WC, FB_MID	39	-0.190	0.821	ns

Discards

Based on data from the present study and CEFAS trip data, the mean biomass of discarded scallops below the minimum landing size (110 mm in sub-area VIId and 100 mm in sub-area VIIe) ranged from 1.5 – 52.9 % per trip. The mean proportion discarded was 20 % in VIId (eastern English Channel) and 27 % in VIIe (western English Channel) respectively (Figure 5.6). The lowest amount of king scallop discards occurred at a site in eastern Lyme Bay (Start Point).

In total, twenty different bycatch species were retained. Individuals of commercial species that were below the minimum landing size for the species were discarded and all other (non-commercial) species were discarded. The mean proportion of commercial finfish and shellfish biomass (not including king scallops) discarded during a trip ranged from 18-100 %. The mean biomass of finfish and commercial shellfish (excluding king scallops) retained per haul across all trips was 36 kg km^{-2} (Figure 5.7). The mean biomass discarded per trip was significantly higher in the eastern English Channel (sub-area VIId, 135 kg km^{-2}) than the western English Channel (sub-area VIIe, 66 kg km^{-2}), ($t=2.0523$, $d.f=32$, $p=0.048$). However, there were fewer samples from the eastern English Channel and there was a large degree of variation in discarded biomass between samples in the eastern English Channel, therefore the statistical significance of the latter result should be interpreted with caution. The higher discards in sub-area VIId were largely attributed to the species *P. platessa*, *S. officinalis* and *M. squinado*.

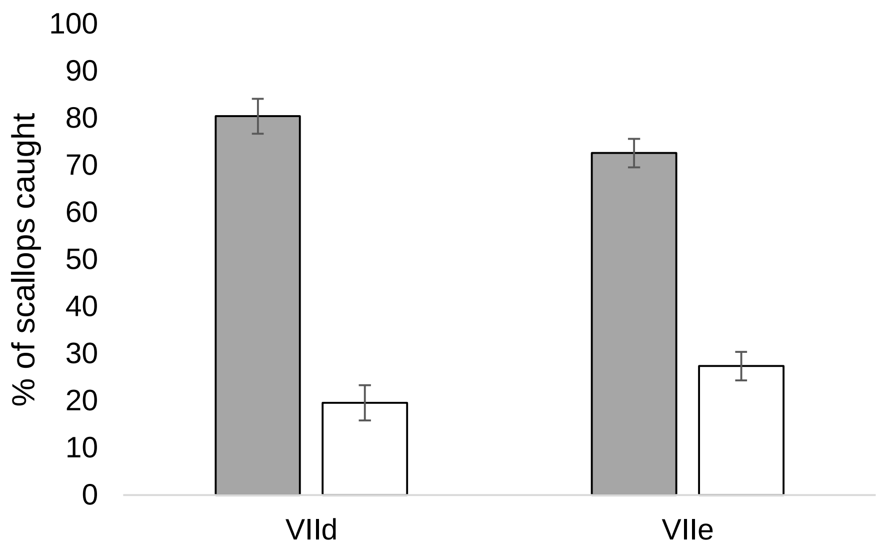

Figure 5.6: Mean proportion (±S.E.) of *P. maximus* in dredge catches that were retained (grey bars) or discarded (white bars) in scallop dredge catches in the eastern (ICES sub-area VIId) and western (ICES sub-area VIIe) English Channel. Data from the present study and CEFAS sampling trips.

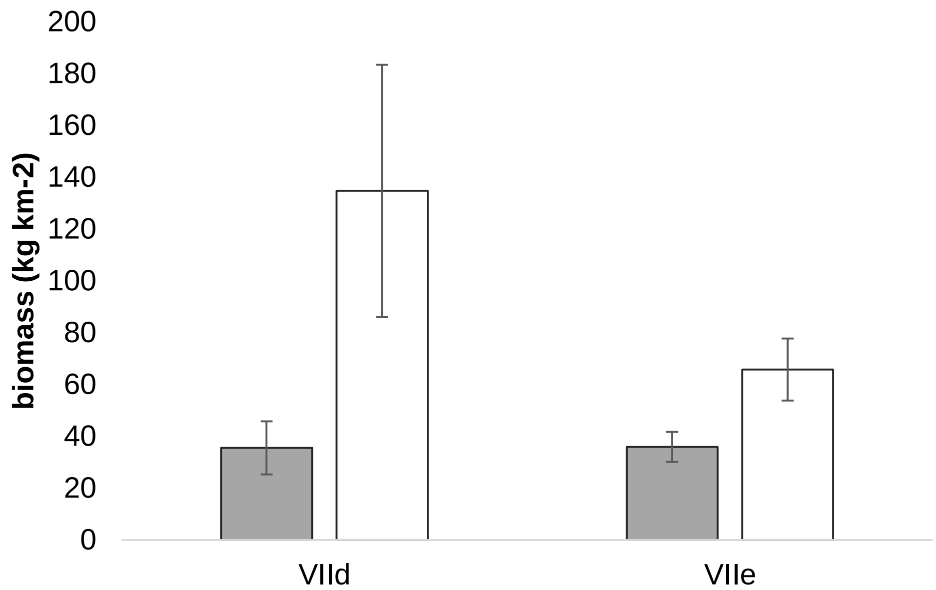

Figure 5.7: Mean biomass (kg km^{-2}) (±S.E.) of finfish and commercial shellfish (excluding king scallops) retained (grey bars) and discarded (white bars) in scallop dredge catches in the eastern (ICES sub-area VIId) and western (ICES sub-area VIIe) English Channel. Data from the present study and CEFAS sampling trips.

5.4.2. Spatial variation in bycatch between the English Channel, Wales and the Isle of Man

There was no significant difference in scallop dredge bycatch species composition from three geographically distinct areas around the Isle of Man; the south, east, and west (ANOSIM: r=0.149, p=0.054), therefore all Isle of Man data was pooled for comparison with catches from Cardigan Bay and the English Channel. The mean biomass of dredge catches was significantly different between all five areas (ANOVA: $F=11.29_{4, 82}$, $p<0.001$). Total catch biomass was greatest in Cardigan Bay (Figure 5.8a), although the highest species diversity occurred in catches around the Isle of Man (Figure 5.8b). Lowest catch biomass occurred in the English Channel bycatch assemblage 'West' (see previous results). Species composition was significantly different between all five areas (ANOSIM: R=0.58, p=0.001, pairwise comparisons in table 5.13). A low R-value (<0.3) between 'EC_Far West' and CB indicates significant overlap in the bycatch species composition of these two areas. Within group similarity ranged from 37 % in the 'EC_West' group to 51 % in Cardigan Bay. Dissimilarity between groups ranged from 61 % (CB/IM) to 88 % (CB/EC_Far West).

P. maximus contributed the highest biomass to catches in all areas of the English Channel, Cardigan Bay and the Isle of Man. Sim/SD values for *P. maximus* were >1.3 in all areas meaning that biomass was consistent between samples within areas. Cardigan Bay was characterised by a higher abundance of king scallops (85 % of catch biomass) than all areas of the English Channel and the Isle of Man, with just three further species contributing to the top 90 % of biomass; *M. squinado* and *Asterias rubens* accounted on average for 4 % and 3 % of catch biomass respectively and *C. pagurus* contributed 1.5 % of catch biomass. In the Isle of Man, *P. maximus* accounted for an average of 47 % of catch biomass. Five species that contributed to the top 80 % of bycatch biomass around the Isle of Man include *A. opercularis* (13 %) and *A. rubens* (11 %), with *Raja naevus*, *E. esculentus* and *Eledone cirrhosa* contributing 4, 4, and 3 % on average. Although a number of commercial finfish and shellfish species were present in both Cardigan Bay and the Isle of Man, catches were low, with no single species contributing >2 % to catch biomass. *A. rubens* contributed consistent catch biomass in all areas of the Isle of Man, but not in Cardigan Bay. Eleven species were responsible for the top 80 % similarity within groups, across all sample areas, of which six are commercially fished (Table 5.14). Typical species for each of the five areas are in table 5.15.

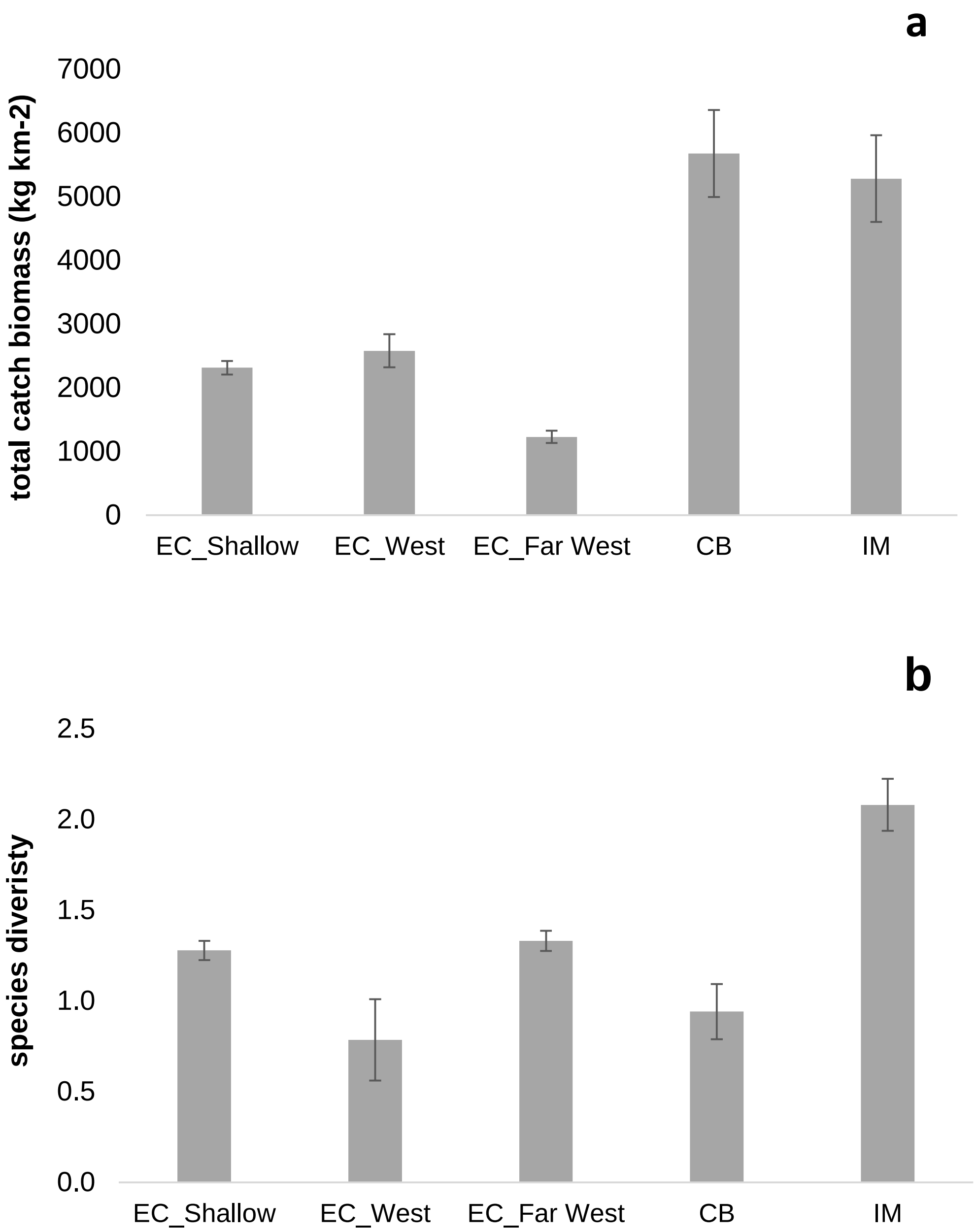

Figure 5.8: a) mean total catch biomass; b) mean species diversity, in scallop dredge catches from five areas: the English Channel (EC_Shallow, EC_Far West, EC_West), Wales (CB) and the Isle of Man (IM). Error bars represent one standard error of the mean.

Table 5.13: Output from a pairwise ANOSIM analysis of the difference in scallop dredge catch species composition between five areas: English Channel (EC_Shallow, EC_Far West, EC_West); Cardigan Bay (CB); Isle of Man (IM).

Groups	R statistic	p-value
EC_Shallow, IM	0.877	0.001
EC_West, IM	0.861	0.001
EC_Far West, EC_Shallow	0.717	0.001
EC_Far West, IM	0.712	0.001
EC_West, EC_Shallow	0.692	0.001
EC_West, CB	0.560	0.001
EC_Shallow, CB	0.547	0.001
CB, IM	0.528	0.001
EC_West, EC_Far West	0.432	0.001
EC_Far West, CB	0.216	0.002

Table 5.14: Species contributing to the top 80% similarity within groups in scallop dredge bycatch at sites in the English Channel, Cardigan Bay and the Isle of Man. Species of commercial importance are in bold text.

English Channel	Cardigan Bay	Isle of Man
Pecten maximus	***Pecten maximus***	***Pecten maximus***
Cancer pagurus	*Asterias rubens*	*Asterias rubens*
Aequipecten opercularis		***Aequipecten opercularis***
Marthasterias glacialis		*Alcyonium digitatum*
Maja squinado		*Luidia ciliaris*
Luidia ciliaris		
Solea solea		
Lophius piscatorius		
Microstomus kitt		

Table 5.15: Typical species found in scallop dredge bycatch in each of the five areas. Species with notably high abundance (a) or those unique to an area (*) are listed.

	English Channel (Shallow)	English Channel (Far West)	English Channel (West)	Cardigan Bay	Isle of Man
Ascidians					*Ascidia conchilega**
Bivalves	*Ostrea edulis**			*Glycymeris glycymeris*[a]	*Arctica islandica** *Modiolus modiolus*[a] *Anomia* sp.[a]
Bryozoans				*Flustra foliacea*[a] *Bugula flabellate** *Chartella* sp. * *Alcyonidium diaphanum*[a]	
Cephalopods	*Sepia officinalis**				*Loligo vulgaris*[a] *Eledone cirrhosa*[a]
Crustaceans				*Maja squinado*[a] *Necora puber** *Homarus gammarus*[a]	
Echinoderms					*Echinus esculentus*[a] *Solaster endeca** *Crossaster papposus*[a] *Stichastrella rosea** *Echinocardium cordatum*[a]
Fish/Sharks/Rays	*Arnoglossus laterna** *Scophthalmus maximus** *Scophthalmus rhombus**	*Lepidorhombus whiffiagonis** *Leucoraja naevus**	*Arnoglossus imperialis**	*Ammodytes* sp.[a] *Blennius ocellaris** *Chelidonichthys lucerna**	*Raja* sp.[a] *Taurulus bubalis**
Gastropods				*Capulus ungaricus**	*Neptunea antiqua*[a] *Colus* sp.[a]
Hydroids				*Abietinaria abietina*[a] *Hydrallmania* sp.[a]	
Soft corals					*Alcyonium digitatum*[a]
Sponges				*Halichondria* sp.*	*Haliclona* sp.[a]

5.5 Discussion

Understanding the amount of bycatch and discarding associated with a fishery is an important step in understanding the sustainability of that fishery. This understanding also helps to identify issues and can drive initiatives that might reduce bycatch if levels are considered unsustainable. The results of the current study provided an estimate of bycatch biomass and composition typically associated with the king scallop dredge fishery across the English Channel. We were able to evaluate bycatches that occur in the English Channel against other major UK king scallop fisheries, which indicated that there is considerable variation in the amount of bycatch associated with king scallop fisheries that occur in different localities.

Bycatches in the English Channel scallop fishery

Overall, 19 % of the wet weight of scallop dredge catches in the English Channel were comprised of bycatch. The proportion of bycatch (as a proportion of the total catch) was similar across all areas sampled in the English Channel. Discards of finfish and commercial shellfish as a proportion of total bycatch were highest in the eastern English Channel. This was mainly due to the high biomass of discarded cuttlefish, plaice and spider crabs that were predominant in the bycatch there. The selectivity of the dredge gear allows small benthic organisms to be riddled out of the bottom of the dredge bag, through interconnecting steel rings, therefore bycatches were dominated by larger benthic species, such as starfish, brown and spider crabs and larger demersal fish species. Individual bycatch species were present at consistently low levels (<9 % of total catch biomass at all sites sampled). There were three exceptions to this; at two sites queen scallops contributed 19 and 28 % of the catch biomass, and at one site the starfish, *M. glacialis,* contributed 17 % of the total catch biomass.

Low catches of commercial fish species may relate to low local abundance of those species at a particular site (Craven *et al.*, 2013), however in the present study, the boundaries of scallop and demersal beam trawl fisheries overlap, particularly in the western English Channel. This overlap suggests a low catch susceptibility of ground fish and commercially important shellfish species to the Newhaven scallop dredge that is used by vessels in the scallop fishery. Incorporating the additional samples from the CEFAS data highlighted that in the English Channel, the biomass of bycatch in scallop dredges is dominated by commercially important, rather than non-commercial species (with the exception of the spiny starfish, *Marthasterias glacialis*). The latter species was particularly prevalent in the bycatch at sites in Falmouth

Bay, with only one further record outside of Falmouth Bay. The commercially important species that dominated the scallop dredge bycatches in the English Channel (including samples from the present study and CEFAS data) were the spiny spider crab, monkfish, queen scallop, brown crab and cuttlefish. Non-commercial species that were also prevalent in catches were the seven-armed starfish (*Luidia ciliaris*) and the common starfish (*Asterias rubens*), the sea urchin, *Echinus esculentus* and the small spotted catshark, *Scyliorhinus canicula*. Other types of species that were retained by the dredge were a number of other flatfish and finfish species, starfish, echinoderms, small crustaceans, bivalves, hydroids and bryozoans. The individual proportion of each of these species in catches was low (on average <0.5 % of catch biomass). Although a low proportion of bycatch is a positive aspect of any fishery, there are secondary effects of dredging that may have significant impacts on other species or community structure (Bianchi *et al.,* 2000; Bradshaw *et al.,* 2002). Such effects include damage to organisms that are not retained by the dredge (Veale *et al.,* 2001), scattering of benthic organisms (Collie *et al.*, 1997), increased susceptibility to predation (Ramsay & Kaiser, 1998) and changes in feeding patterns (Ramsay *et al.*, 1996).

Broad-scale variation in bycatch in scallop fisheries

Environmental and physical conditions at the seabed vary across a variety of spatial scales which affects the related species assemblage composition. Bycatch assemblages in some fisheries are known to vary with depth, season and other abiotic factors (Probert *et al.,* 1997; Bergmann *et al.,* 2002; Rodrigues-Filho *et al.,* 2013). There was moderate correlation between the species resemblance matrix and depth and mean seabed temperature. Bycatch composition was significantly different between all but two of the author sample sites, the latter were located closest to each other. In the English Channel, three distinct bycatch assemblages were identified that related to the environmental parameters at the associated sample sites. At a broader geographic scale, in ICES area VII, significant differences in bycatch assemblage composition were found between scallop fishing grounds in Cardigan Bay and around the Isle of Man. Many of the species contributing to the majority of dissimilarity in bycatch assemblage between the English Channel, Cardigan Bay and the Isle of Man, were present in all areas but were not consistently abundant between samples in all areas (Sim/SD values <1.3), indicating that there is high variation in bycatch abundances at localised, as well as larger spatial scales across the extent of the fishery. Small scale differences in the bycatch composition in Cardigan Bay are attributed to geographic variation rather than management area (Lambert *et al.*, 2014b).

Temporal and spatial variation

Spatial and temporal variation is inherent in bycatch data (Allen *et al.,* 2002; Borges *et al.,* 2004; Craven *et al.*, 2013). Therefore, even with many samples covering a broad temporal and spatial scale, bias may hide patterns in the data and stratification of sampling effort cannot guarantee reliable samples (Rochet & Trenkel 2005). Bycatch from scallop fisheries can vary with location, vessel, gear configuration, season, environmental and weather conditions, tow duration, trip and haul. Seasonal variations in fish and invertebrate abundance and behaviour are also likely to influence the prevalence of certain species in catches (Wilberg *et al.*, 2010). Identifying hotspots or certain times or year when bycatch species are more prevalent, or more susceptible to capture, can help inform management measures that could reduce these bycatches, such as the use of temporary closed areas or particular fishing gears. In the present study there was a particularly high biomass of the common cuttlefish, *S. officinalis* in catches in the Baie de Seine. Cuttlefish are a commercially important cephalopod species in the north-east Atlantic and the main fishing grounds are in the English Channel. The species is short-lived and recruitment to the fishery peaks in autumn when juveniles migrate to offshore wintering grounds (Royer *et al.,* 2006). Sampling at the site in the Baie de Seine occurred at the end of September, which coincided with the high abundance of cuttlefish observed in these samples. If the catch quantity of this species was of concern, management could restrict fishing activity to times of the year when catchability is lower. Due to the lack of seasonal resolution, and the lack of samples from larger vessels in the current dataset, it is not possible to raise the data to the annual landings of the scallop fleet in the English Channel. However, the mean contribution of cuttlefish to the overall catch in the Baie de Seine was 7.8 %, therefore it is unlikely that the mean proportion of cuttlefish bycatch throughout the year would exceed 5 % of the total catch.

Sampling in the present study is weighted towards summer sampling in the western English Channel, with fewer samples during winter and from the eastern English Channel. This is largely due to the difference in the total number of vessels that target scallops in each area and the seasonality of the scallop fishing in the inshore eastern English Channel, which provided few sampling opportunities. The data does however indicate that overall, bycatch of commercially important and sensitive species is low compared to bycatch in other fisheries (Kelleher 2005).

Discards

In terms of biomass, discards of bycatch are higher in the eastern English Channel. On average 73 % of biomass from scallop dredge catches in the English Channel is discarded, although non-commercial species account for the majority of the discarded biomass. Between 18 and 100 % of commercial bycatch species biomass from scallop dredge catches in the English Channel is discarded. This level of discarding is higher than in other UK fisheries (Table 5.16), potentially due to a lack of quota or lack of desire to retain any bycatch species (personal observations). In the UK, vessels fishing with a scallop licence are currently allowed to retain quantities of commercial bycatch species that amount to up to 5 % of the total catch weight of scallops. To retain further bycatch, the relevant quota must be obtained otherwise the bycatch must be discarded at sea. However, a ban on discarding pelagic (e.g. mackerel and herring) and quota species (such as cod, haddock and whiting) is currently being phased in under the new CFP regulations (European Commission 2013), meaning that by 2019 all commercial bycatch species may have to be landed. Although this will result in a significant increase in landed biomass for some fisheries (Catchpole *et al.,* 2008; Poos *et al.,* 2010), commercial species account for <7 % of total catch biomass in the English Channel king scallop fishery. Therefore, although the impacts of the new legislation will be less significant than for other fisheries, there are likely to be financial implications and logistical issues associated with the retention of discards (Mangi & Catchpole, 2013).

Table 5.16: Estimates of discard rates for a selection of different vessel segments in the U.K. fishery showing the average vessel length, the annual landings per vessel, the percentage of total discards and quota species discards. Source: Mangi & Catchpole 2013.

Vessel segment	Average vessel length (m)	Annual landings per vessel (t)	% of total discards	% of quota discards
<10 m drift/fixed nets	8	21.6	16.7	6
Gill netters	18	146	6.8	2.5
<10 m demersal trawl/seine	10	27	16.7	5.9
Area VIIb-k trawlers 10-24 m	13	74.8	16.7	6
North Sea beam trawlers <300 kW	14	74.9	15.6	5.7
North Sea *Nephrops* < 300 kW	14	90.9	22.6	8.1
South west beam trawlers <250 kW	20	129.2	16.6	6
South west beam trawlers >250 kW	27	252	16.7	6
South west beam trawlers >250 kW	21	234	22.5	8.2

Discards of undersized scallops are likely to be higher in areas that are fished heavily and/or have recently been harvested as the majority of scallops over MLS will have been removed from the area. The present study revealed that discarding of undersized scallops is more frequent in the western English Channel than in the eastern English Channel. Fatal damage to *P. maximus* can occur during dredging and varies between 2 and 20 %, largely due to spatial variation in shell thickness (Beukers-Stewart & Beukers-Stewart, 2009). Intermediate damage may not be immediately fatal but could lead to an increased likelihood of predation (Caddy, 1973; Jenkins & Brand, 2001). Damage to the mantle, similar to that caused during dredging, increases the likelihood of death within 30 days post-dredging (Gruffyd, 1972). The majority of damage that occurs is in the form of small chips on the perimeter of the shell that, although unlikely to cause immediate problems, can result in the redirection of energy from reproduction to repair leading to lower reproductive output (Kaiser *et al.,* 2007). Mortality following dredging is greater in younger scallops as their smaller size means they are more likely to be caught up in the mesh of the steel belly and they may be more susceptible to the effects of stress (Gruffyd, 1972; Maguire *et al.,* 2002). Due to the greater incidence of undersized discards in the western English Channel, improving gear efficiency to reduce the amount of undersized scallops retained by the dredge (Lart *et al.,* 2003) would provide benefits to the stock.

Impacts of dredging on benthic organisms

In the present study, the majority of species caught by scallop dredges in the English Channel composed <5 % of the overall biomass in the catches. The most prevalent species in the bycatch was the queen scallop, which contributed on average 6 % of overall catch biomass. Queen scallops can experience minor shell damage when caught by dredge gear (Montgomery, 2008). Although this damage alone is unlikely to prove fatal, mortality increases with length of emersion time and the damaged shell may cause an increased risk of predation or disease once returned to the seabed. Hence, the likelihood of such factors should be accounted for when assessing the vulnerability of discarded species to the impacts of dredging (Pranovi *et al.,* 2001). Estimates of discard survival in many studies are likely to be overestimated due to the limited time period of experiments and the lack of accountability of indirect effects such as secondary predation (Policarpo, 2012). The indirect effects of scallop dredging can have greater effects on benthic species than just the capture process alone (Eleftheriou & Robertson, 1992; Jenkins *et al.,* 2001). In addition, interacting physical and biological factors (often species-specific) can increase mortality rates (Davis, 2002; Benoit *et al.*, 2010, 2013).

'Unobserved fishing mortality' (definition in Alverson *et al.*, 1994) is the mortality that occurs as a result of direct contact by the fishing gear that does not result in capture. Species that are more susceptible to damage (soft bodied, fragile organisms) show higher post-capture mortality (Depestele *et al.*, 2014). The damage sustained within the dredge belly is a function of the number of stones retained and the dredge fullness (Veale *et al.*, 2001). A higher proportion of rocks, or large, heavy rocks in the dredge can caused greater damage to bycatch depending on species morphology. However, other methods of capture, such as trawling can have a greater impact on certain species than scallop dredging (Kaiser *et al.*, 1996; Royer *et al.*, 2006). Inter-specific damage levels may vary throughout the year due to seasonal changes in morphology, such as in ovigerous brown crabs in which the abdomen becomes distended, or in the common starfish that becomes large and swollen after the warm summer period and therefore more susceptible to damage (Veale *et al.*, 2001). Table 5.17 broadly summarises information currently available for damage levels sustained from bottom towed fishing gear, or post-capture survival rates for the main commercial species found in scallop dredge catches in the English Channel. As a high proportion of rocks was recorded in dredge catches in the English Channel (75-92 % of the total catch weight), it might be anticipated that damage rates to bycatch would be high and hence survivorship would be low for many species depending on their morphology as indicated in Table 5.17.

Monkfish dominate the bycatch assemblage of scallop dredges in the Isle of Man fishery (Duncan, 2009; Craven *et al.*, 2013). Monkfish caught in the Isle of Man king scallop fishery are generally juveniles, which might have implications for the viability of the wider stock if the level of mortality was significant (Craven *et al.*, 2013). Authors of the latter study noted a decrease in the abundance of monkfish in the Isle of Man scallop dredge fishery over a 13 year period. Monkfish are commercially valuable, slow growing and vulnerable to exploitation. The impact of the scallop dredge fishery on monkfish stocks therefore needs to be given further consideration in the context of monkfish stock assessment. This issue is particularly relevant in western Falmouth Bay, where monkfish contributed nearly 8 % to scallop dredge catch biomass. *C. pagurus* and *S. maximus* also contributed >5 % of catch biomass in this area and may warrant further consideration in fishery management plans. Some species experience limited impact from bottom fishing gears. For example, *Scyliorhinus canicula* have survival rates of up to 98 % from beam trawl catches (Kaiser and

Table 5.17: Damage level/post capture survival information for commercial species contributing on average >1 % of total weight from scallop dredge catches in the English Channel; as well as other taxa present in dredge catches. Studies relate to the following fishing gears: [a]Scallop dredge; [b]Beam trawl; [c]Rapido trawl.

Commercial species	Damage level/post-capture survival rate	Reference
P. maximus	low damage, high survival	Jenkins *et al.*, 2001[a]
A. opercularis	low damage, high survival	Veale *et al.*, 2001[a]
C. pagurus	moderate damage (increases when dredge contact occurs but the individual is not retained in the dredge bag)	Jenkins *et al.*, 2001[a]
M. squinado	high damage	Hall-Spencer *et al.*, 1999[c]
L. piscatorius	low post capture survival (predicted)	Craven *et al.*, 2013[a]
S. officinalis	low survival	Anon, 2012[b]
Other taxa		
Asteroidea	high survival	Depestele *et al.*, 2014[b]
Gastropoda	high survival	Depestele *et al.*, 2014[b]
Crustacea	variable survival (species specific)	Depestele *et al.*, 2014[b]
Rajiformes	high survival	Depestele *et al.*, 2014[b]
Pleuronectiformes	moderate survival	Depestele *et al.*, 2014[b]

Spencer, 1995; Revill *et al.,* 2005) and the abundance of this species increased over time in the Isle of Man scallop dredge fishery (Craven *et al.*, 2013).

As for *S. canicula*, Rajiformes tend to have high survival rates, whereas Pleuronectiformes and Gadiformes are more susceptible to damage and death than the other two taxa (Benoit *et al.*, 2013). Plaice, Dover sole and oysters are classified as species 'of principal importance for the purpose of conserving biodiversity' under the UK Biodiversity Action Plan. However, each of these species contributed <0.5 % of overall catch biomass from scallop fisheries in the English Channel, therefore the scallop fishery is unlikely to have potential population consequences for these species. Catches of native oysters could have potential negative effects on local isolated populations of this species and a system of voluntary declaration of such landings or voluntary no fishing zones to protect these species could prove beneficial. Oysters were found in the bycatch at a single site in the present study, contributing just 0.3 % to mean bycatch biomass, therefore population impacts from the dredge fishery are likely to be minimal and restricted to specific locations in the English Channel.

A bycatch species of particular concern is the thornback ray, *Raja clavata* due to a number of contributing factors. Slow growth, late maturity, low fecundity, and susceptibility to dredge capture have led to a recent precautionary 20 % reduction in the TAC (Lart *et al.,* 2014). However, this measure will not prevent the species being caught and as post-capture survival has not been empirically quantified, any catches have the potential to impact negatively on the population. This species accounted for an average of 0.2 % of catch biomass in the English Channel, therefore population impacts are likely to be minimal, however closure of nursery and/or breeding grounds may provide greater protection for this species (Lart *et al.,* 2014).

Some species were typical in scallop dredge bycatch in all the areas assessed in the present study (English Channel, Wales, Isle of Man), as well as in the Shetland dredge fishery (see Shelmerdine 2010). These species included *C. pagarus*, *A. rubens*, *E. esculentus*, *A. opercularis*, *L. piscatorius*, *L. ciliaris,* and *N. antiqua*. The prevalence of these species in dredge catches is likely to be a combination of their affinity with specific scallop habitats and their susceptibility to capture in dredges. The response of hermit crabs to dredged areas vary between species. While *P. bernhardus* actively migrate to freshly dredged ground to feed on damaged fauna, *P. prideaux* elicited no such response (Ramsay *et al.*, 1996). The latter two species have similar diets; therefore the differences in response were attributed to competition or niche separation. Populations of species such as *C. pagarus* may require consideration for monitoring programmes where removals from the scallop dredge fishery are significant in relation to the commercial *C. pagurus* fishery. This may occur in western Falmouth bay (site S2 in the present study) where this species contributed 9 % to total catch biomass.

Chronic or frequent fishing can lead to wide-spread depletion of benthic invertebrate prey species (Hiddink *et al*., 2006b; Hinz *et al*., 2009b), which may alter trophic interactions (Hilborn, 2011). However, fisheries also generate carrion, providing food subsidies to demersal fish and other scavengers (Garthe *et al.*, 1996). This in turn can affect the feeding response of scavengers. Predators and scavenging species such as starfish, hermit crabs, brittlestars and whelks feed on damaged organisms left in the path of the dredge, and prevalence of such species following dredging can therefore increase, or they may become the dominant fauna in areas subject to bottom trawling and dredging (Berghahn, 1990; Ramsay *et al.,* 1998; Collie *et al.*, 1997, 2000). Although whelks appear to suffer little or no physical damage following contact with dredge gear, they become more susceptible to predation after such events suggesting that fishing may indirectly increase whelk mortality (Ramsay & Kaiser 1998). Other scavengers such as starfish may experience population growth with increased dredge activity and therefore subsequent mortality from the fishery may be insignificant (Kaiser & Hiddink, 2007).

Reducing bycatch

In many fisheries, legal requirements and a need for better management of marine resources has driven the development of improved fishing gear, resulting in an increase in catch efficiency (Hinz *et al.*, 2009) and selectivity (Graham *et al.*, 2007; Lart *et al*., 2003). The ecosystem impacts of bottom-trawling and dredging are linked to habitat type, which is highly correlated with depth (Kaiser *et al.*, 2006). Some authors cite the steel teeth of the dredge as being the most harmful component of the dredge to benthic organisms (Shephard *et al.*, 2009) but it is likely that the weight of the belly bag dragging along the seabed is responsible for a large amount of damage, in particular to soft, erect or fragile organisms (Hinz *et al.*, 2012). Alternative dredge designs have addressed both of these issues, although not simultaneously. Alternatives to the Newhaven scallop dredge design have been developed with the aim of: reducing environmental impacts; reducing unwanted bycatch (organisms and inert material such as rocks and stones); and increasing fuel efficiency while maintaining or increasing catches of the target organism, the king scallop. There have been national and EU funded projects (Ecodredge FAIR CT98-4465; Lart *et al.*, 2003) to encourage development of new scallop dredge designs. The N-Virodredge™ is a modified Newhaven dredge that replaces the fixed metal teeth with individually sprung tines and has rollers underneath the metal collector bag. The tines aim to reduce seabed penetration and drag; to avoid larger cobbles or boulders becoming trapped by the teeth and being dragged along the seabed. The tines may reduce damage to organisms due the individual movement of the tines allowing greater escape gaps and significantly lower tension compared to the tooth bar of a Newhaven dredge, although this has not yet been empirically tested. Trials have indicated that fewer stones are retained and dragged over the seabed which reduces the weight of the belly (and therefore impact on benthic fauna) as well as reducing damage to the catch (Filippi, 2013). In the 'Oban' dredge design the steel belly bag is replaced with square-meshed rubber matting to reduce the overall weight of the bag. A further design known as the 'skid-dredge' lifts the belly bag off the seabed using metal skids. This allows better riddling of smaller organisms from the bottom of the bag and significantly reduces the area of seabed contacted as the dredge passes over. At the present time, empirical testing of these designs is too limited for satisfactory conclusions to be drawn regarding the reduction of impact on bycatch species.

Conclusions

Due to inherent variation in bycatch assemblages, coupled with seasonal variation in the abundance of certain species (e.g. Veale *et al.*, 2001), accurate estimates of bycatch can only be obtained through regular sampling, covering an appropriate spatial, temporal and seasonal scale. Distinct geographic areas, defined by physical and biological parameters should be incorporated into sampling plans. The results of this study indicate that overall bycatch in the English Channel king scallop fishery is low in relation to other fisheries and other scallop dredge fisheries that occur elsewhere in the UK. There are few commercial bycatch species that warrant concern or additional consideration in fishery management plans. The proportion of bycatch in Cardigan Bay, Wales is slightly less than that in the English Channel; however higher bycatch biomass occurs around the Isle of Man (on average 53 % of total catch weight). Bycatch species composition varies with localised and broad spatial scales, which is attributed to differences in physical and environmental conditions as well as seasonal variations in species abundances and catch susceptibility. Bycatch can be reduced by: using improved fishing gear that reduces bycatch and impacts on organisms that are not retained by the dredge; seasonal management restrictions to remove fishing impacts during times when certain species are more vulnerable to capture; and reduced overall fishing effort.

CHAPTER 6: CONNECTIVITY BETWEEN KING SCALLOP, *PECTEN MAXIMUS* L., STOCKS IN THE ENGLISH CHANNEL

MSC data requirements addressed:

P1	Data requirements
Stock Status	Principle 1 of the MSC assessment criteria states that the unit of certification is "*The fishery or fish stock (biologically distinct unit) combined with the fishing method/gear and practice (=vessel(s) and/or individuals pursuing the fish of that stock) and management framework*". In order to meet the requirements of Principle 1, biologically distinct populations, for which appropriate management and harvest strategy can be implements, must be identified.

The research in this was completed in collaboration with Dr Natalie Hold (Bangor University) and is due to be submitted as a journal paper, currently in preparation:

Genetic structure of the commercially important scallop *Pecten maximus* and implications for fisheries management.

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Abstract

Populations of commercially exploited bivalves are often managed as single stocks across large scale management units. Evidence for genetic structure among nine distinct scallop beds within the English Channel was investigated using previously developed microsatellite markers. A genetically distinct population was identified in the Baie de Seine on the French side of the English Channel. Scallop aggregations from Lyme Bay in the western English Channel to Sussex in the eastern English Channel showed evidence of connectivity, which was linked to the dominant oceanographic current regime. At a local scale, more complex processes occur within Falmouth Bay, with each of three sample sites within the bay significantly genetically differentiated from the others. Effective population sizes were >>1000 for all locations except for west Lyme Bay and one sub-population from Falmouth Bay. The results give an indication of the largest spatial scale at which the English Channel scallop fishery should be managed. Three major management units were identified: the Baie de Seine; the English side of the Channel from eastern Falmouth Bay to Sussex; Falmouth Bay and north of Cornwall in the western English Channel. The Baie de Seine population was not significantly differentiated from the north Cornwall population therefore further samples from the north coast of France and the Channel Islands will provide evidence of the mechanism of linkage between these populations and other populations around the British Isles.

6.1 Introduction

A biological stock can be defined as "*a group within a species population which has sufficient spatial and temporal integrity to warrant consideration as self-perpetuating units*" (Pawson, 1995). For fisheries management purposes this relates to the extent to which exploitation effects of a fishery are identifiable in a species population. In relation to fisheries management it is important to understand the level of connectivity between geographically distinct populations in order to set appropriate harvest rules and maintain genetically 'healthy' stocks.

P. maximus are patchy in distribution, which is reflected in the distribution of known commercial fishing activity (Stelzenmuller *et al.,* 2008). For many marine bivalves it has been suggested that larvae are capable of dispersing over wide geographical ranges through passive transport that is moderated by larval behaviour (Robins *et al.,* 2013). This dispersal can provide gene flow or connectivity between disparate fishing grounds (Brand, 1991). Connectivity can help to maintain biomass and genetic diversity, and reduces the risk of population decline in harvested populations. Oceanographic features such as gyres, together with larval behaviour can prevent movement of larvae out of an area leading to isolated, self-recruiting populations. Genetically distinct stocks represent the largest unit at which a species should be managed in order to meet fishery and/or conservation objectives (Fogarty & Botsford, 2007). Self-recruiting populations (or those with a small effective population size (N_e)) are more susceptible to over-harvesting. The effective population size is the number of individuals that would produce the same amount of genetic drift or inbreeding as the actual population. A general rule of thumb is that effective population size is 0.2-0.5 of the total population size (Mace & Lande, 1991). Others estimate that N_e should be at least 50 (Reed & Bryant, 2000) to 5000 (Lande, 1995) individuals. Problems from inbreeding or genetic drift may arise when the proportion or number is lower. In fisheries management N_e is a key factor when setting sustainable catch levels.

The Channel scallop fishery

There is no TAC (Total Allowable Catch) limit imposed on *P. maximus* in EU waters. Total UK scallop landings increased from 27,000 tonnes in 2008 to 53,300 tonnes in 2012, of which *P. maximus* is the main species and to a lesser extent the queen scallop, *Aequipecten opercularis*. Activity of scallop vessels > 15 m LOA is managed through the Western Waters

(ICES area VII) effort regime (Council Regulation (EC) No 1415/2004) in the English Channel, where landings have increased steadily over the last decade (MMO, 2012). Further effort control measures are implemented within 6 NM of the coastline, such as dredge limitations and curfews. These measures are not based on stock assessments but defined by administrative area rather than population dynamics. The lack of a full stock assessment means that little is known about the census size of scallop stocks in English waters or whether the current yield is sustainable (19,400 tonnes were landed into English ports in 2012). Such data would inform sustainable management but for that to occur there needs to be an understanding of stock structure.

Reproductive biology

Scallops are simultaneous hermaphrodites and reproduce by shedding gametes into the sea, where fertilisation and embryological development occur externally (Cragg & Crisp, 1991). *P. maximus* are considered *r*-strategists due to the vast number of eggs produced in a single spawning combined with prolonged and/or repeated periods of larval dispersal (Mackie & Ansell, 1993). The fertilised embryos disperse in the water progressing through the stages of trochophore, veliger and pediveliger (when the larvae become capable of crawling) before settling to the seabed prior to metamorphosis. The survival and growth rate of juvenile scallops has been attributed to behaviours exhibited in the early stages of development; the ability to delay metamorphosis and actively seek out a suitable substratum on which to settle prior to metamorphosis (Culliney, 1974). From three days old, swimming behaviour occurs as an alternating pattern of periods of spiralling upwards motion, followed by cessation of activity and sinking due to the density of the larval shell (Cragg & Crisp, 1991). Mean vertical velocity is 0.5-1.2 mm s^{-1} depending on the larval stage (Cragg, 1980). Food supply, temperature and depth can all influence the development of larvae and *P. maximus* larvae can take between 31 and 55 days to metamorphose (at 18°C and 12°C respectively) (Cragg, 2006). This reflects the period of time the larvae spend in the water column prior to settlement and the influence environmental factors can have on dispersal time and distance.

Hydrodynamics

Identification of bivalve larvae is difficult (Paugam *et al.*, 2003) therefore direct observation cannot be used to map the movement of larvae in the ocean. Analysis of connectivity between marine populations can be informed using oceanographic models and knowledge of pelagic larval duration (PLD) (Grantham *et al.,* 2003; Shanks, 2009). Such models demonstrate the net movement of water between spawning grounds to provide insight into the potential direction and magnitude of larval transport. Larval behaviour coupled with spatial variations in oceanographic features creates complex patterns of connectivity structure that contributes to self-recruitment or dispersal (Knights *et al.,* 2006; Cowen *et al.,* 2006; Woodson and McManus, 2007). Including larval behaviour in models can lead to new insights into the mechanisms that influence connectivity (Galindo *et al.*, 2010, Robins *et al.,* 2013). Although larval dispersal distance is correlated with genetic differentiation for some species, this does not reflect the general rule for marine organisms and for many species it is a poor predictor of genetic differentiation (Weersing & Toonen, 2009; Lee *et al.,* 2013). Additionally PLD is correlated with dispersal distance; however the relationship is bimodal with species with short PLD having both small (<1 km) to large (>100's km) dispersal distances (Shanks, 2009). Residual currents, although one to two orders of magnitude weaker than tidal currents, can be strong enough not to be disturbed by weather events. However, coastal morphology and the time of day or season of larval release in relation to tidal and residual currents also have strong influence on the fate of larvae (Robins *et al.,* 2013).

The main current flow in the English Channel runs from west to east, resulting in a net easterly transport of larvae in summer (Dare *et al.*, 1994). However the modelled mean influx to the North Sea at the eastern end of the Channel is relatively low at 0.06 Sverdrup (1 Sverdrup = 10^6 m^3 per second), and close to zero in the spring time (OSPAR Commission, 2000). Wind speed and direction also have a large impact on net volume of water flux in the English Channel (Gerritsen *et al.*, 2001), where strong tidal currents and residuals occur (averaging 5 cm s^{-1}; Salomon, 1991). Further non-linear effects of tidal motion occur as eddies and tidally induced gyres, particularly near Portland Bill, Baie de Seine, Cap de la Hague, Falmouth Bay and Lyme Bay (Salomon & Breton, 1993; Dare *et al.*, 1994). Residual circulations such as these are likely to restrict the dispersal of marine larvae and effects can be amplified by seasonally induced stratification. This occurs in the Irish Sea *Nephrops norvegicus* fishery (Hill *et al.,* 1997). Seasonally stratified waters occur offshore from Falmouth Bay with a frontal jet travelling from east to west just outside the bay. Inshore waters are characterised by a mixed water column with anticlockwise tidally induced eddies in the west of the bay and clockwise eddies in the east (Ferentinos & Collins, 1979; Figure 6.1). This provides a mechanism to prevent mixing of larvae across, and retention within, Falmouth Bay.

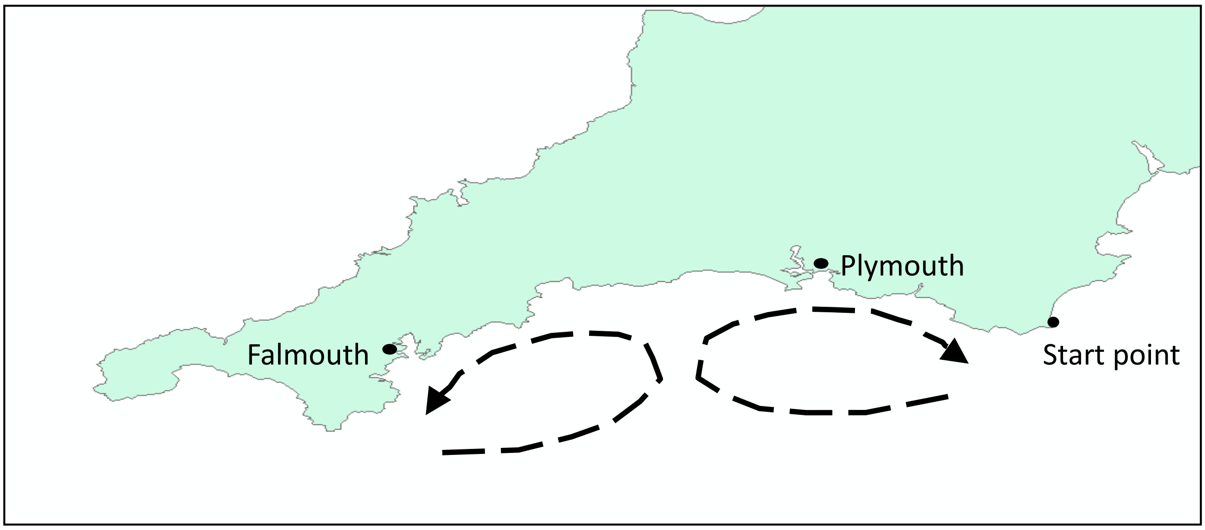


Figure 6.1: Residual circulation eddies in Falmouth Bay. Image reproduced from Ferentinos & Collins, 1979.

Mean bottom stress is much stronger in the eastern than the western Channel, with minimal bottom stress in Falmouth Bay (Pingree & Griffiths, 1979). Tidal bed shear stress ranges from 0.03 N m^{-2} in some inshore areas to 3 N m^{-2} in the centre of the English Channel, but scallops occur at commercial densities where bed shear stress is <2 N m^{-2} (see Chapter 4). This results in a virtual absence of scallops in the middle of the English Channel. The majority of the sea bed in the eastern Channel is < 50 m deep and in the western Channel the majority of the seabed is > 50 m deep. Seasonal stratification is therefore greater in the western Channel, and influences food availability and hence survival of benthic organisms such as scallops. Current evidence suggests that some retention of larvae occurs within Falmouth Bay, Lyme Bay and the Baie de Seine, with minimal connectivity between the French and English sides of the Channel and limited connectivity between the English Channel up in to the Irish and North Seas (Table 6.1, Figure 6.2).

Table 6.1: Summary of published evidence of the genetic and hydrodynamic connectivity (or isolation) between scallop beds in the English Channel.

Location	Evidence	Authors
Western Channel	• Minimal larval exchange between French and English scallop populations. • Low level of dispersal from western to eastern Channel. • Possible recruitment from the west in southern Cornwall. • Local retention in Lyme Bay. • Limited dispersal east from both Falmouth Bay and Lyme Bay (due to an anti-clockwise gyre).	Dare *et al.*, 1994
Lyme Bay Eastern Channel	• Dispersal from east to west • Potential recruitment from French populations to eastern English Channel, facilitated by anti-clockwise gyre west of Beachy Head. • Larvae travelling up into North Sea are lost (no reported populations south of Scarborough).	CEFAS 2012
Baie de Seine	• Majority of stock are recruited from within the Bay (variable and active larval swimming models combined with hydrodynamics) • Some larvae move west towards Baie de St Brieuc and north eastern populations in the bay provide a source for the eastern English Channel. • Wind induced currents alter the direction and extent of larval transport, in addition to the influence of residual tidal currents.	Nicolle *et al.*, 2013
North Cornwall/ Irish Sea	• Limited dispersal of cockle larvae from southern Irish Sea towards north Cornwall.	Robins *et al.*, 2013

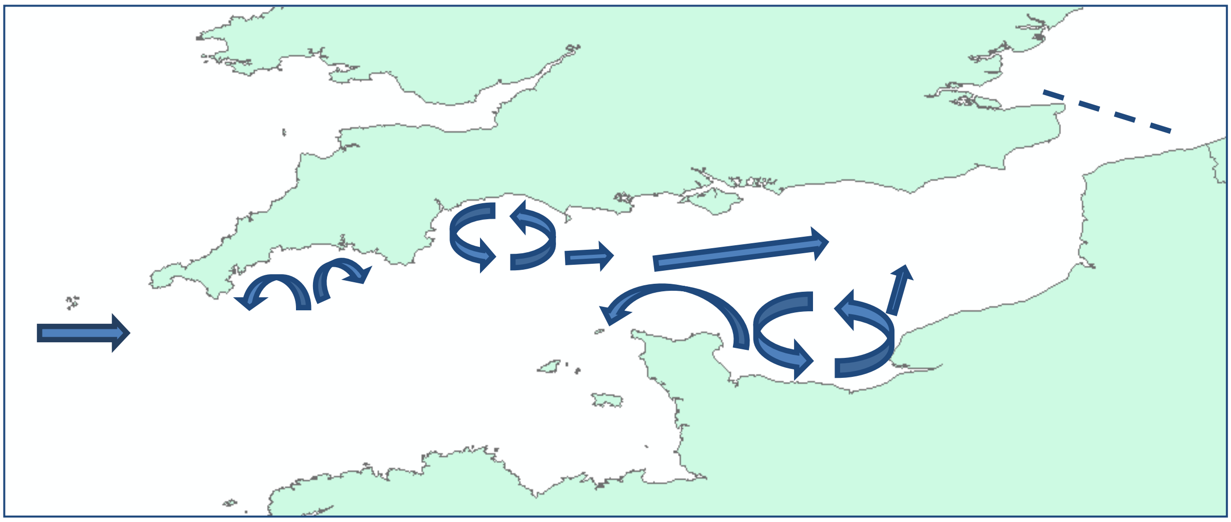


Figure 6.2: Expected connectivity of scallop aggregations in the English Channel based on the predicted dispersal of larvae from previous studies.

Genetic connectivity

If connectivity among populations is low and gene flow is restricted, genetic drift may lead to divergence in allele frequencies among populations. This divergence can be detected using molecular markers such as microsatellites. Differences in allele frequencies can be used to estimate the genetic connectivity between two populations. Genetic drift and mutation serve to increase differentiation between isolated populations (where natural selection is not a significant contributing factor), but very low levels of migration from outside can mask the variation caused by these processes (Avise, 1994). A complex array of interactions (including migration, mutation, population bottlenecks and gene flow) produces empirical genetic signals that are commonly used to assess population differentiation. For any population, the current genetic signal reflects long-term average patterns (Galindo *et al.*, 2006) and therefore it is important to combine genetic connectivity matrices with biological and oceanographic knowledge. The absence of significant genetic differentiation does not necessarily equate to high connectivity as homogenisation of allele frequencies can occur at very low levels of migration. However, significant genetic differentiation does infer low connectivity between populations e.g. one individual or less per generation (Hellberg *et al.*, 2002). This level of genetic differentiation is relevant for fisheries management and defines the largest scale at which a stock should be managed. Genetic analyses of the population structure of marine organisms are further complicated by numerous climatological and hydrodynamic processes, combined with temporal and spatial variations, which influence the movements of larvae. Modern connectivity can also be attributed to populations sharing a recent common ancestor.

The shelf-wide hydrodynamic system is likely to have existed since deglaciation c. 8000 years ago (Hill *et al.*, 2008). In unexploited grounds, scallops greater than 9 years of age dominate (Brand *et al.*, 1991) but the lifespan can exceed 20 years in extreme cases (Tang, 1941). This represents approximately 1600 or more generations over 8000 years, through which divergence could have occurred. In an evolutionary context this is a relatively short time scale.

Anthropogenic activity can also influence the genetic signal of populations. Such activities include harvesting, or seeding of commercial scallop beds with spat from other areas that can lead to homogenisation of allele frequencies being mistaken as connectivity. Transplantation has been documented in the Isle of Man, Portland Harbour and various locations in Ireland (Beaumont, 2000). The Channel Islands were seeded with scallops from Scotland and Ireland in 2006 (Jonathan Shrives, States of Jersey, pers. comm.), and scallops from northwest Scotland have been transplanted to the Baie of St. Brieuc (Mackie & Ansell, 1993). Particularly heavy fishing intensity over a short time period has the potential to cause a population bottleneck. This can lead to very low allelic diversity or conversely, high heterozygosity due to recent large populations (Beebee & Rowe, 2004). Factors such as these demonstrate that that genetic analysis is best interpreted in the context of other types of information where available.

To successfully identify genetic differences between populations, genetic markers that are neutral with regard to natural selection are used. Previously, the use of allozyme loci failed to identify differentiation between various European *P. maximus* populations (Beaumont *et al.*, 1993). Microsatellites are a type of co-dominant DNA marker useful in determining population structure (Carvalho & Hauser, 1998). They are generally non-coding, neutral in relation to natural selection and highly polymorphic. Each locus is identifiable by a particular sequence of nucleotide base pairs in the chromosome onto which primers can bind. Through polymerase chain reaction (PCR) amplification, repeat motifs can be replicated many times. Microsatellites are tandem repeats of 2-10 base pair motifs and have been used extensively in recent years in studies of population ecology due to their fast mutation rates (Zhang & Hewitt, 2003). The limitations of using microsatellites include, variations in mutation rates, allele size difference may not be directly related to divergence, and questionable neutrality of some microsatellite sequences (some sequences may be of functional importance) (Zhang & Hewitt, 2003). Null alleles are also common in microsatellites and arise when mutations in the primer binding cause amplification failure. This results in, deviation from Hardy-

Weinberg Equilibrium (HWE) due to an apparent excess of homozygotes, and a decrease in the fixation index (F_{ST}) values (Chapuis and Estoup, 2007). Null alleles are common in marine bivalves (Kenchington *et al.*, 2006; Lallias *et al.*, 2010).

To date, 42 microsatellite markers have been isolated for *P. maximus* (Watts *et al.*, 2005 [9]), Charrier *et al.*, 2012 [9]; Hold *et al.*, 2013 [12]; (Morvezen *et al.*, 2013 [12]). Using microsatellites, Hold (2012) identified connectivity between scallop populations in the Irish Sea, around Ireland, and Scotland; reinforced by residual patterns in ocean currents. The aim of the current study was to test for genetic connectivity at the scale of the commercial fishery in the English Channel. The results are discussed in the context of hydrodynamic, physiological and ecological processes and can be used to aid definition of appropriate spatial scales for the management of the fishery.

6.2 Methods

Sample collection

Pecten maximus tissue samples were collected from eight sites in the English Channel. A further sample from Falmouth Bay (FAL) from Hold (2012) was also included in the analysis (Figure 6.3). Sample collection was arranged by liaising with fishermen operating in the areas where samples were required, who recorded co-ordinates for the location of a marked bag of scallops. On landing the catch the live scallops were collected and transported to the processors in a refrigerated lorry, where they were shucked and the mantle and frill tissues from 100 scallops were placed in individual bags and flash frozen. The frozen samples were sent to the author by overnight courier. On receipt, the samples were allowed to defrost and a small piece of mantle tissue (approximately 5 x 5 mm) was cut and stored in 100 % ethanol in 1.5 ml microcentrifuge tubes.

DNA extraction and amplification

DNA was extracted using CTAB extraction buffer (Doyle & Doyle, 1987). Samples were homogenised in 40 µl CTAB buffer (40 µl of 2% CTAB buffer to 20 µl of mercaptoethanol). 200 µg of Proteinase K was added to each sample, which were then incubated overnight at 59°C. The following day phenol-chloroform-isoamyl alcohol was used to extract the DNA followed by two chloroform-isoamyl (24:1) washes. The DNA was precipitated with twice the volume of cold absolute ethanol (stored in the freezer) and 0.1 times the volume of sodium chloride (pH 8, 5M). After precipitation at -20°C for 1 hour the solution was centrifuged causing a pellet to form and the ethanol poured away. The pellet was allowed to air dry for 10 minutes. The DNA was re-eluted using 50 µl of TE buffer (pH 8) and stored at 4°C (Hold, 2012).

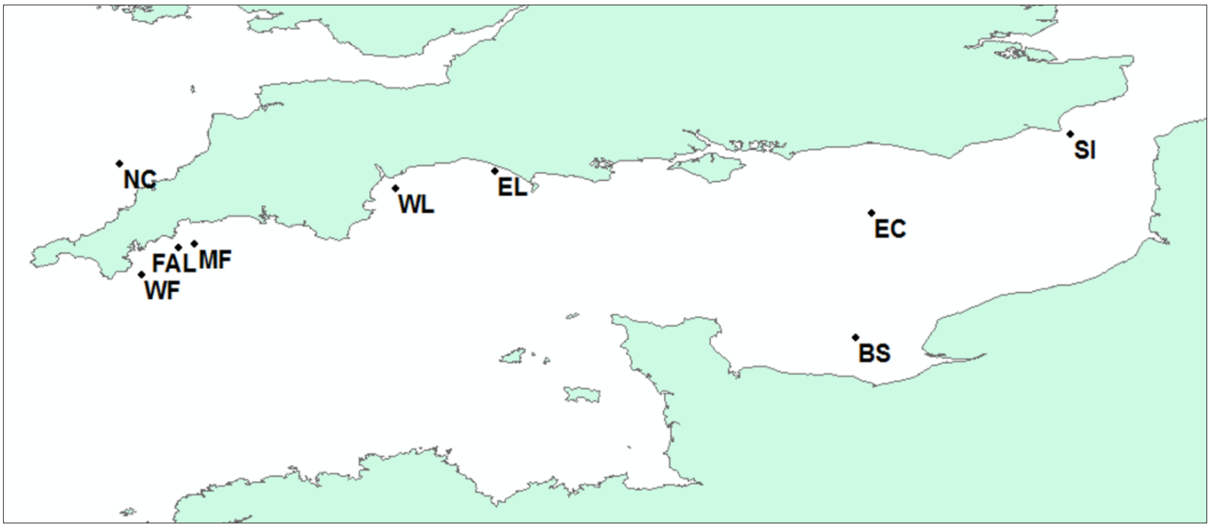


Figure 6.3: Location of the eight sites from which *P. maximus* tissue samples were obtained as well as the existing data from Falmouth Bay (FAL). NC – north Cornwall; WF – west Falmouth Bay; MF – mid Falmouth Bay; WL – west Lyme Bay; EL – east Lyme Bay; BS – Baie de Seine; EC – eastern Channel; SI – Sussex inshore.

12 microsatellites (see Hold *et al.*, 2013) were amplified using 3 multiplex polymerase chain reactions (PCR). 1 µl of DNA (c. 10 ng) was combined with 6 µl of Qiagen Type IT multiplex PCR mix, 2.5 µl dH_2O, 2 µl of Primer mix and 0.5 µl BSA (10mg ml^{-1}). The Primer mix contains the forward and reverse Primers for all loci in the multiplex as well as fluorescent tails (PET®, VIC®, NED®, FAM®). There were 3 PCR thermo-cycling profiles used (Profile one for multiplex A and C, profile two for multiplex B and profile three for multiplex D) (Table 2). Profile one used a touchdown PCR regime to increase stringency; 95 °C for 5 min, (95 °C 30 s, 64 °C [- 1 °C per cycle] 90 s, 72 °C 30 s) x 6 cycles, (95 °C 30 s, 58 °C 90 s, 72 °C 30 s) x 9 cycles, (95 °C 30 s, 50 °C 90 s, 72 °C 30 s) x 15 cycles with a 45 min extension at 72 °C. Profile two; 95 °C 5 min, (95 °C 30 s, 51 °C 90 s, 72 °C 30 s) x 16 cycles, (95 °C 30 s, 50 °C 90 s, 72 °C 30 s) x 19 cycles, final extension of 45 min at 72 °C. Profile 3 also used a touchdown regime; 95 °C 5 min, (95 °C 30 s, 60 °C [- 1 °C per cycle] 90 s, 72 °C 30 s) x 10 cycles, (95 °C 30 s, 50 °C 90 s, 72 °C 30 s) x 13 cycles, final extension of 45 min at 72 °C (Hold *et al.*, 2013). PCR products were identified on an ABI3130XL sequencer using the LIZ600 size standard. The Genemapper® software was used to size alleles.

6.3 Data Analysis

Data quality and loci characteristics

Three markers were removed from the final analysis due to failure of amplification or missing frequency data. Tests for null alleles, large allele dropout (both caused by failure of an allele to amplify to detectable levels during PCR) and stuttering (strand slippage during DNA synthesis) were conducted using Microchecker (Van Oosterhout *et al.*, 2004). Null allele frequencies were estimated using the software FreeNa which uses the EM algorithm method (Dempster *et al.*, 1977) to estimate null allele frequencies, preferable to other algorithms as it produces less biased estimates (Chapuis & Estoup, 2007)..

GenePop v 4.0 (Rousset, 2014) was used to investigate linkage disequilibrium for each pair of loci in each population (594 tests). For both tests the Markov Chain parameters used were a burn-in of 10000 followed by 100 batches of 5000 iterations each. P-values were corrected using the False Discovery Rate (FDR) method (Benjamini & Hochberg, 1995). This method controls for the expected proportion of false discoveries amongst the rejected hypotheses. The false discovery rate is a less stringent condition than the family-wise error rate and therefore more powerful.

GeneAlEx 6 (Peakall & Smouse, 2012) was used to estimate the observed and expected heterozygosity (Ho and He), the number of alleles (Na) and the effective number of alleles (Ne). Allelic richness is more sensitive to a decrease in population size or past bottlenecks than heterozygosity (Nei *et al.*, 1975). Allelic richness, adjusting for variation in sample size using the rarefaction method (Hurlbert, 1971), and the number of private alleles in each population, was calculated using HP-Rare (Kalinowski, 2004, 2005). Departure from Hardy-Weinberg equilibrium was calculated for each population/locus combination in GeneAlEx 6 and p-values were corrected for multiple testing using the FDR method.

Population structure

GenAlEx 6.5 (Peakall & Smouse, 2006, 2012) was used to calculate a pairwise genetic distance matrix by population using G''_{ST} (Hedrick's standardized G_{ST} further corrected for bias when the number of populations is small; Meirmans & Hedrick, 2011). G''_{ST} is used rather than F_{ST} due to the highly polymorphic nature of alleles in microsatellites. F_{ST} was designed for biallelic alleles, and for multi-allelic markers, the maximum possible value of between-population differentiation is not necessarily equal to one, but is instead determined by the amount of within-population diversity (Hedrick, 1999). For microsatellites with high heterozygosity the maximum F_{ST} or G_{ST} value is often 0.1-0.2. A principal coordinate analysis (PCoA) with standardisation was performed to identify the populations that contribute most to genetic differentiation.

Effective population size

Effective population size (N_e) was estimated using the Linkage Disequilibrium (LD) one-sample method in LDNe (Waples & Do, 2010). Random mating was used. Using rare alleles of low (0.001) frequency can fail to exclude any alleles, and upward bias of N_e rises sharply; however in many cases the effect is not too severe. Therefore, an allele frequency of ≥0.02 was used as the threshold for the sample sizes (n = 31-45) in this study, as suggested by Waples & Do (2010). The test is better at detecting small N_e than medium or large N_e. In populations with N_e <50 useful information can be gained from sample sizes as low as 25, but the method is likely to provide imprecise estimates if N_e is >1000. Therefore the method can be used to detect populations with Low N_e but moderate to large estimates may not be reliable. In the context of this study, it may be a useful indicator of populations that may require greater conservation management where low N_e is detected.

6.4 Results

Data quality and loci characteristics

The W5 allele had low amplification success and therefore was not used in analysis. Microchecker showed that markers P60 and W4 indicated no evidence of null alleles while all other markers indicated the presence of null alleles; however this was not consistent across populations (Table 6.2). An average of 4.67 markers in each population indicated an excess of homozygotes (ranging from 3 to 7). Marker P68 indicated an excess of homozygotes in all nine populations. There was no indication of large allele dropout for any marker across all populations. The software indicated that for markers P68 and P75, stuttering might have resulted in scoring errors in six populations, due to a significant shortage of heterozygote genotypes with alleles that had one repeat unit difference. For markers P11 and P73 stuttering may have occurred in two populations and for P9 stuttering may have occurred in one population. The peaks were re-checked for stutter errors but no evidence of uncalled alleles confused as stutter peaks was found, and the author is confident that all alleles were called and not confused as stutters. Following correction of p-values for multiple testing, only two allele pairings (both from the FAL sample) showed significant linkage disequilibrium at the p>0.05 level; this is therefore not indicative of linkage of any markers and all markers were retained for further analyses.

Table 6.2: Evidence of null alleles in 11 microsatellite markers across nine populations of *P. maximus* in the English Channel. %: the percentage of populations with evidence of null alleles.

	P11	**P59**	**P60**	**P68**	**P70**	**P73**	**P75**	**P9**	**W12**	**W4**	**W8**	**Total**
FAL	No	No	No	Yes	No	No	Yes	Yes	No	No	No	3
BS	No	No	No	Yes	Yes	Yes	Yes	No	No	No	Yes	5
EC	Yes	No	No	Yes	No	Yes	Yes	Yes	No	No	No	4
EL	No	No	No	Yes	No	Yes	Yes	Yes	No	No	No	4
MF	Yes	No	No	Yes	No	Yes	Yes	Yes	No	No	No	5
NC	No	No	No	Yes	No	Yes	Yes	Yes	Yes	No	Yes	6
SI	Yes	No	No	Yes	No	No	No	No	No	No	Yes	3
WF	Yes	No	No	Yes	Yes	No	Yes	Yes	Yes	No	Yes	7
WL	Yes	Yes	No	Yes	No	No	No	Yes	No	No	No	4
%	**56**	**11**	**0**	**100**	**22**	**56**	**78**	**78**	**22**	**0**	**44**	

Observed heterozygosity (Ho) was similar to expected heterozygosity (He) and ranged from zero at one monomorphic marker in WF and WL samples to 0.833 in BS (Table 6.3). Heterozygosity was lowest at marker P70 in five populations, and highest at marker W12 in six populations. Mean observed heterozygosity (ranging from 0.261 in NC to 0.340 in EL) was lower than the expected heterozygosity (ranging from 0.364 in NC to 0.424 in EC) in all populations. The number of alleles on each locus ranged from 2 to 14. Markers P11 and P70 had the lowest number of alleles (from 2 to 4 across all populations). P68 and W12 had the highest number of alleles (7-11 and 8-12 across populations, respectively). The effective number of alleles ranged from 1.000 in WL to 6.348 in SI. The mean number of alleles ranged from 4.818 in the BS to 6.273 in SI, but the mean number of effective alleles ranged from 2.015 in FAL to 2.414 in WL.

Table 6.3: Microsatellite locus information for all nine populations of *Pecten maximus*. n = sample size; He = expected heterozygosity; Ho = observed heterozygosity; Na = number of alleles; Ne = number of effective alleles, Nr = allelic richness (adjusted for sample size by rarefaction), Np = number of private alleles. He values in bold represent those samples that did not conform to HWE using the Fisher's exact test, after adjusting for multiple tests using the false discovery rate.

Locus	Measure	FAL	BS	EC	EL	MF	NC	SI	WF	WL
P11	n	44	31	38	45	43	44	44	40	36
	He	0.127	0.200	**0.355**	0.287	**0.295**	0.066	**0.201**	**0.265**	**0.296**
	Ho	0.136	0.097	0.105	0.333	0.163	0.068	0.091	0.050	0.250
	Na	2	2	3	4	3	2	2	3	2
	Ne	1.146	1.250	1.549	1.403	1.418	1.071	1.252	1.361	1.420
	Nr	1.563	2.929	2.465	2.166	2.095	1.333	1.757	2.165	2.704
	Np	0.000	0.732	0.113	0.127	0.013	0.000	0.000	0.012	0.162
P59	n	45	37	38	45	41	45	44	41	36
	He	**0.695**	0.656	0.659	0.696	0.705	**0.630**	0.674	0.715	**0.733**
	Ho	0.733	0.676	0.605	0.800	0.683	0.578	0.545	0.610	0.583
	Na	5	4	5	7	6	5	6	8	8
	Ne	3.279	2.903	2.935	3.293	3.386	2.700	3.063	3.513	3.746
	Nr	3.807	3.088	3.469	3.801	3.663	3.194	3.492	4.021	4.321
	Np	0.741	0.019	0.238	0.383	0.146	0.048	0.359	0.417	0.582
P60	n	45	36	37	45	43	41	44	41	36
	He	0.369	0.394	0.307	0.310	0.322	0.360	0.338	0.357	0.387
	Ho	0.311	0.361	0.216	0.356	0.349	0.293	0.364	0.390	0.278
	Na	5	5	2	3	5	4	5	4	5
	Ne	1.586	1.649	1.443	1.450	1.475	1.564	1.510	1.554	1.630
	Nr	2.555	2.397	1.918	2.022	2.475	2.394	2.264	2.294	2.589
	Np	0.467	0.258	0.000	0.094	0.340	0.199	0.144	0.256	0.379
P68	n	45	33	38	45	42	44	44	41	36
	He	**0.724**	0.759	**0.812**	**0.801**	**0.782**	0.818	0.771	0.815	0.831
	Ho	0.467	0.485	0.395	0.533	0.476	0.614	0.545	0.537	0.583
	Na	7	9	9	8	8	10	11	9	9
	Ne	3.623	4.149	5.328	5.031	4.582	5.508	4.360	5.405	5.931
	Nr	4.002	4.897	5.307	5.116	5.057	5.385	5.008	5.310	5.767
	Np	0.666	0.236	0.448	0.090	0.279	0.349	0.381	0.406	0.359
P70	n	45	36	37	44	40	45	42	41	34
	He	0.022	**0.180**	0.104	0.109	0.025	0.085	0.047	**0.094**	**0.085**
	Ho	0.022	0.083	0.108	0.114	0.025	0.044	0.048	0.000	0.029
	Na	2	3	4	4	2	2	3	3	3
	Ne	1.022	1.220	1.116	1.122	1.025	1.093	1.049	1.104	1.093
	Nr	1.122	1.889	1.574	1.583	1.138	1.412	1.262	1.504	1.461
	Np	0.122	0.530	0.218	0.240	0.053	0.142	0.127	0.360	0.210

Locus	Measure	FAL	BDS	EC	EL	MF	NC	SI	WF	WL
P73	n	45	33	38	44	42	44	43	42	34
	He	0.201	**0.388**	**0.363**	**0.267**	**0.279**	**0.265**	0.175	**0.178**	0.266
	Ho	0.133	0.152	0.158	0.205	0.167	0.114	0.140	0.143	0.235
	Na	3	5	6	5	6	5	6	4	5
	Ne	1.252	1.634	1.569	1.365	1.387	1.361	1.212	1.217	1.362
	Nr	1.894	2.942	2.796	2.387	2.457	2.291	1.939	1.928	2.392
	Np	0.144	0.242	0.231	0.264	0.176	0.093	0.284	0.079	0.337
P75	n	45	37	36	45	43	44	42	41	36
	He	**0.496**	0.480	**0.655**	**0.620**	**0.651**	**0.526**	0.617	**0.598**	**0.702**
	Ho	0.200	0.324	0.278	0.200	0.326	0.273	0.500	0.293	0.278
	Na	4	4	5	6	5	5	6	5	5
	Ne	1.982	1.921	2.903	2.633	2.862	2.109	2.611	2.489	3.358
	Nr	2.692	2.677	3.593	3.422	3.317	2.733	3.590	3.080	3.820
	Np	0.413	0.047	0.136	0.118	0.144	0.067	0.737	0.095	0.270
P9	n	45	35	37	44	41	43	42	37	34
	He	**0.363**	0.136	0.245	0.442	**0.324**	**0.111**	0.158	**0.331**	**0.218**
	Ho	0.133	0.143	0.162	0.318	0.220	0.023	0.119	0.162	0.088
	Na	7	5	6	10	6	3	5	6	6
	Ne	1.569	1.158	1.324	1.792	1.478	1.125	1.187	1.495	1.279
	Nr	2.769	1.763	2.228	3.421	2.686	1.581	1.855	2.672	2.201
	Np	0.726	0.205	0.378	1.067	0.348	0.067	0.244	0.315	0.544
W12	n	44	36	35	44	42	44	44	42	33
	He	**0.763**	0.780	0.807	0.754	0.787	**0.740**	0.842	**0.797**	0.784
	Ho	0.705	0.833	0.743	0.659	0.786	0.591	0.795	0.619	0.697
	Na	11	8	9	10	9	9	14	10	12
	Ne	4.227	4.539	5.169	4.067	4.685	3.853	6.348	4.927	4.634
	Nr	5.122	5.374	5.469	4.871	5.193	4.889	6.137	5.330	5.384
	Np	0.617	0.132	0.376	0.352	0.244	0.169	1.063	0.602	0.595
W4	n	45	35	38	44	43	45	44	42	36
	He	0.242	0.056	0.172	**0.151**	0.133	0.166	0.169	0.135	0.000
	Ho	0.267	0.057	0.184	0.114	0.140	0.178	0.182	0.143	0.000
	Na	4	2	4	4	5	3	4	4	1
	Ne	1.319	1.059	1.208	1.177	1.154	1.199	1.203	1.156	1.000
	Nr	2.196	1.292	1.902	1.805	1.737	1.824	1.813	1.698	1.000
	Np	0.275	0.014	0.156	0.266	0.247	0.052	0.142	0.137	0.000
W8	n	42	37	35	44	42	42	42	39	31
	He	**0.136**	**0.225**	0.186	0.109	**0.201**	**0.242**	**0.279**	**0.235**	0.094
	Ho	0.095	0.162	0.143	0.114	0.190	0.095	0.190	0.154	0.097
	Na	4	6	4	4	8	8	7	6	4
	Ne	1.157	1.290	1.228	1.122	1.252	1.319	1.387	1.307	1.103
	Nr	1.738	2.239	1.952	1.583	2.147	2.363	2.466	2.229	1.532
	Np	0.330	0.630	0.305	0.178	0.593	0.963	0.702	0.781	0.233

Mean allelic richness (adjusted for sample size using rarefaction) was similar across all populations, ranging from 2.57 in FAL to 2.94 in WL (Table 6.4). The number of private alleles per locus ranged from 0 to 0.781. The mean number of private alleles across loci varied between populations. NC had the lowest mean number of private alleles (0.19), while FAL and SI had the highest mean number of private alleles (0.38 and 0.35 respectively).

Table 6.4: Mean allelic richness (adjusted for sample size) over loci for each population (Nr) and mean number of private alleles over loci (Np).

	FAL	**BS**	**EC**	**EL**	**MF**	**NC**	**SI**	**WF**	**WL**
Mean Nr over loci	2.57	2.73	2.93	2.86	2.85	2.63	2.83	2.86	2.94
Mean Np over loci	0.38	0.25	0.23	0.28	0.23	0.19	0.35	0.29	0.31

Of the 99 population/locus combinations, 36 did not conform to Hardy-Weinberg equilibrium (HWE) at an adjusted p-value of $p=0.001$. SI had only one loci (W8) out of HWE and MF had six loci out of HWE, while the other populations varied in the number of loci out of HWE within this range (Figure 6.4a). Two loci (W12 and P60) conformed to HWE in all populations, while the remaining loci varied between populations (Figure 6.4b). No loci deviated from HWE in all populations. All but one of the deviations was due to heterozygote deficiency which in some cases may have been caused by the presence of null alleles.

The conformity of allele frequencies to HWE is an assumption for many analyses in population dynamics (e.g. STRUCTURE); however these data clearly violate this assumption therefore these tests were not used. Testing for population bottlenecks was also not carried out, as this test looks for an excess of heterozygotes.

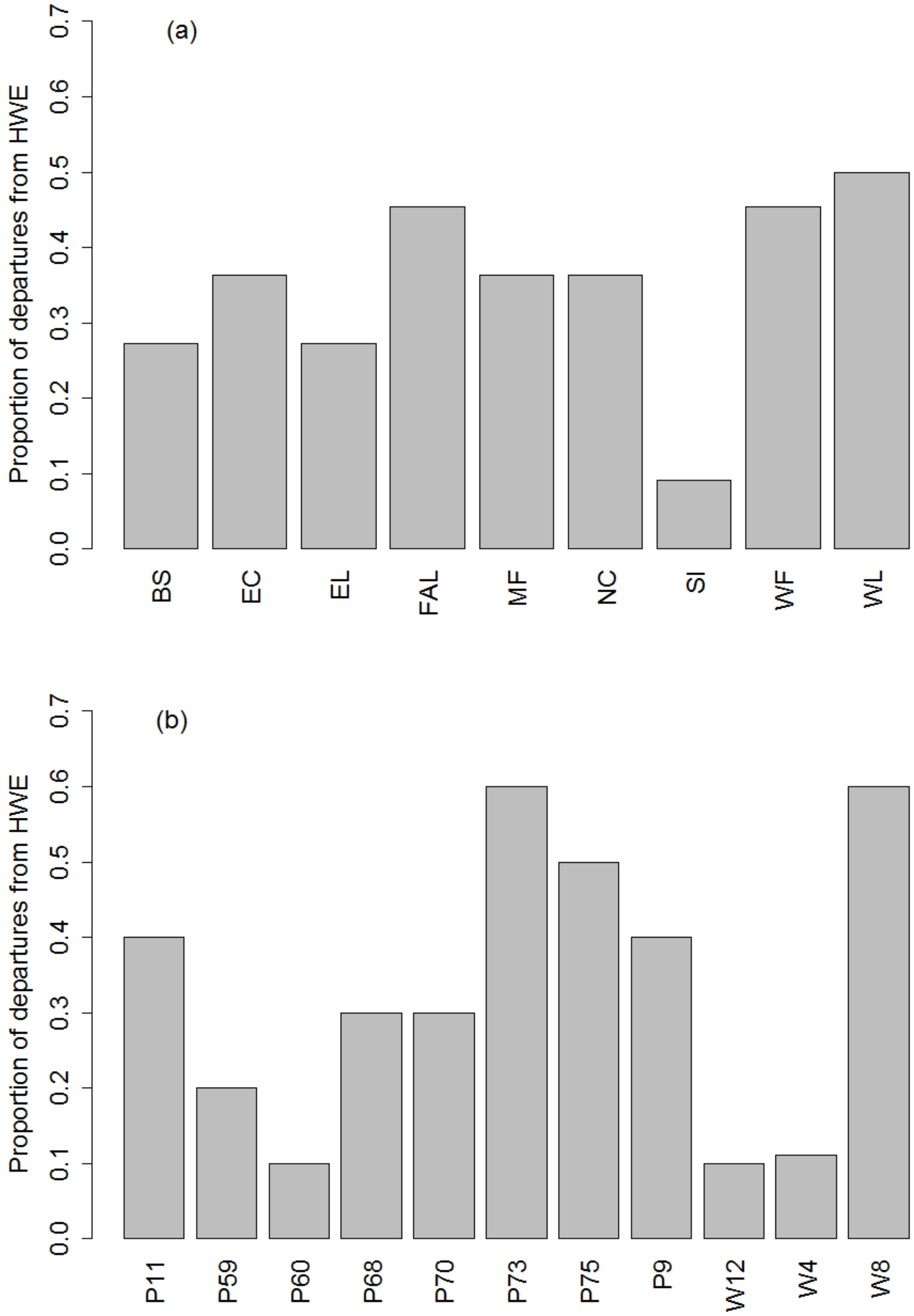


Figure 6.4: Proportion of departures from Hardy-Weinberg equilibrium (HWE) by (a) population and (b) locus. P-values were corrected for multiple comparisons using the false discovery rate method with a significance level of p=0.001 used.

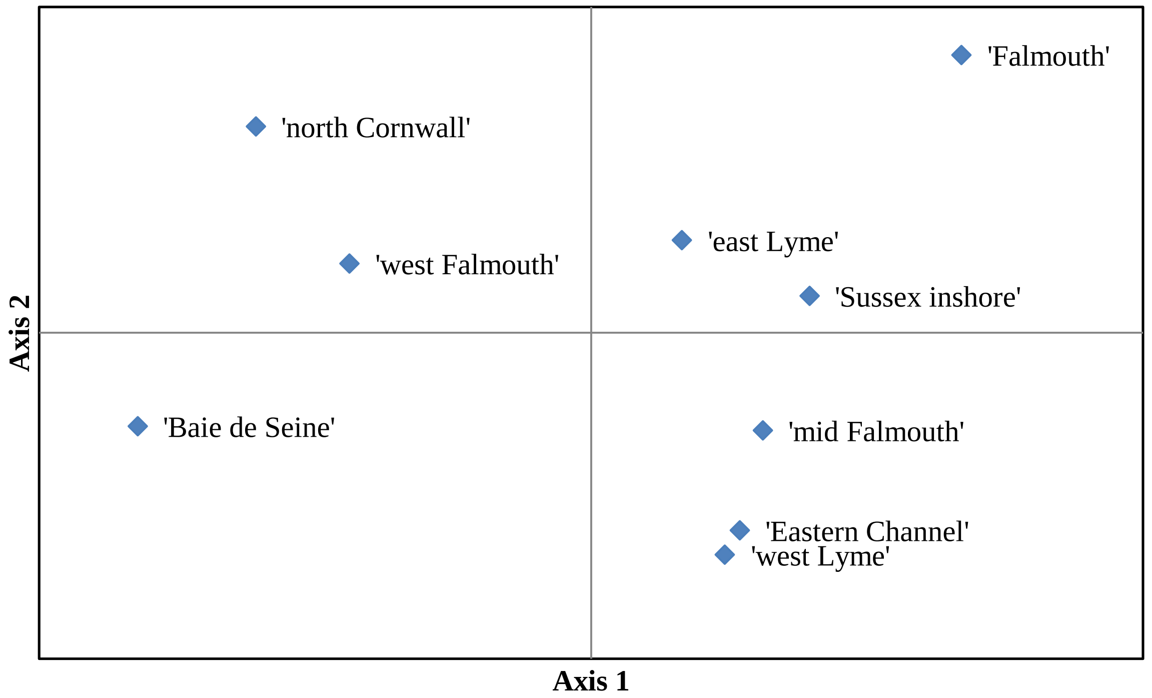
Population structure

Pairwise G"$_{ST}$ values ranged from -0.001 to 0.131 (Table 6.5). F_{ST} values were in most cases, around half the value of the corresponding G" $_{ST}$ values, however the same pairwise comparisons were significant, therefore G"$_{ST}$ values are presented and discussed as G''$_{ST}$ is more suitable for multiallelic markers. F_{ST} relates to the reduction in heterozygosity due to population structure. G"$_{ST}$ values represent the degree of migration, or connectivity between the populations, where a value of 1 represents complete differentiation.

Table 6.5: G"$_{ST}$ values for nine populations of *P. maximus* calculated using the Fisher's exact test shown below the diagonal, with corresponding p-values above the diagonal. Significant pairwise G"$_{ST}$ values at the p=0.05 level in bold indicate significant differentiation between populations.

	FAL	**BS**	**EC**	**EL**	**MF**	**NC**	**SI**	**WF**	**WL**
FAL	*	0.001	0.003	0.007	0.009	0.001	0.008	0.001	0.001
BS	**0.131**	*	0.001	0.001	0.001	0.213	0.001	0.217	0.001
EC	**0.030**	**0.065**	*	0.326	0.621	0.001	0.229	0.001	0.867
EL	**0.027**	**0.057**	0.003	*	0.549	0.001	0.036	0.039	0.512
MF	**0.024**	**0.067**	-0.003	-0.002	*	0.001	0.168	0.003	0.873
NC	**0.116**	0.005	**0.063**	**0.041**	**0.067**	*	0.001	0.380	0.001
SI	**0.022**	**0.083**	0.005	**0.014**	0.007	**0.078**	*	0.001	0.095
WF	**0.083**	0.006	**0.037**	**0.015**	**0.030**	0.001	**0.042**	*	0.002
WL	**0.036**	**0.061**	-0.010	-0.001	-0.009	**0.060**	0.010	**0.031**	*

FAL was significantly differentiated from all other populations and the greatest differentiation occurred between BS and FAL. FAL is in close proximity to WF although these two populations also had relatively high and significant differentiation (G"$_{ST}$ = 0.083). BS was significantly differentiated from SI, MF, EC, WL and EL. PCoA plots (Figure 6.5) indicated that BS and FAL were the most distinct populations, while NC and WF were somewhat distinct from the other sites.



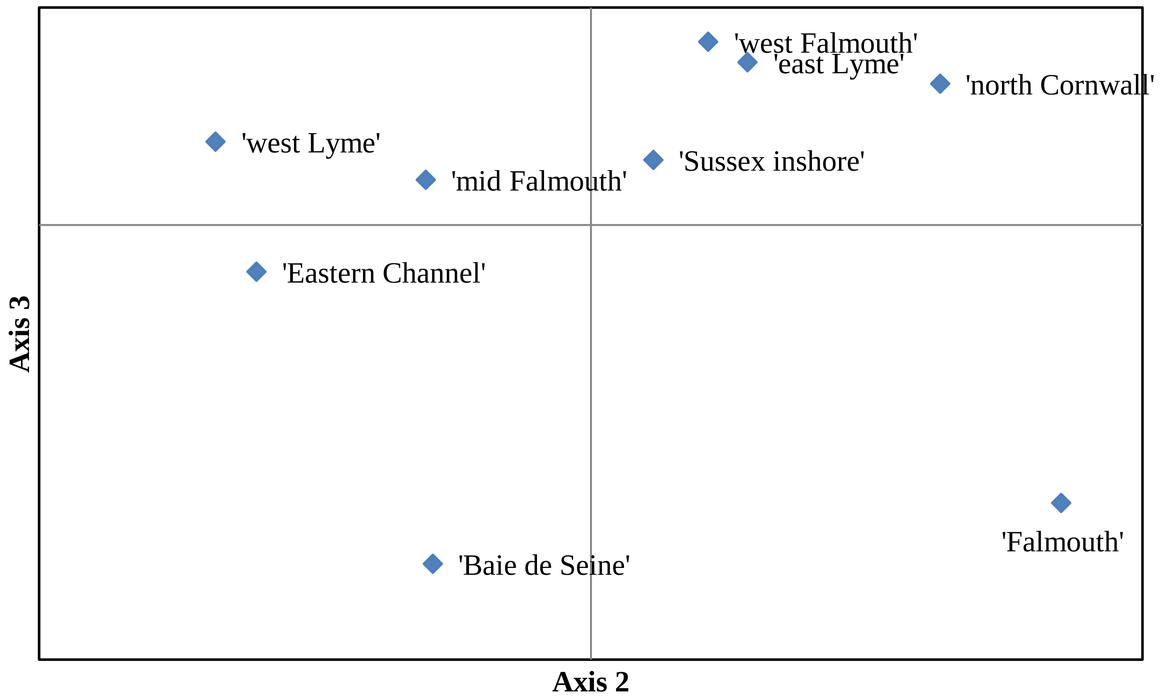


Figure 6.5: Principal coordinates analysis of genetic differentiation among populations of *P. maximus* in the English Channel. PCA axis 1 accounts for 77.5 % of the variation; axis 2 explains 13.4 %; axis 3 (not shown) explains 4.0 %.

Effective population size

For six populations the estimate of N_e was negative, with infinite confidence intervals (CIs) (Table 6.6). This means that any disequilibrium present is attributed to sampling error and there is no evidence of disequilibrium caused by genetic drift due to a limited population size. The method is reliable for populations up to 1000 and it is therefore assumed that N_e of these populations is >>1000.

Falmouth had the lowest N_e of 27 (CI: 14-64, obtained by jack-knifing among pairs of loci), as reported in Hold (2012). North Cornwall (NC) had a low N_e of 110 (CI: 50-3587). West Lyme Bay (WL) also had a low N_e of 137; however the upper CI was infinite using allele frequencies of ≥0.02. Using a higher allele frequency threshold generally downwardly biases N_e.

Table 6.6: Effective population size (N_e) estimates using linkage disequilibrium with 95 % confidence intervals (CI) obtained from parametric and jack-knifing methods. Negative N_e indicates that N_e is >>1000 and no evidence of disequilibrium due to limited number of parents was found. Populations with low N_e are highlighted in bold.

		No. independent				CIs for N_e			
	n	alleles	r^2	Exp(r^2)	Ne	Parametric		Jack-knife Loci	
FAL	44	701	0.0359	0.0244	**26.8**	19.1	39.4	14	64
BS	31	643	0.0314	0.0354	-86.5	-1596.2	∞	460.7	∞
EC	38	832	0.0298	0.0302	-824.8	122.1	∞	81.7	∞
EL	45	1101	0.0233	0.0244	-314	328.9	∞	183.9	∞
MF	43	933	0.0256	0.0261	-745.1	160.8	∞	92.8	∞
NC	44	684	0.0294	0.0265	**109.7**	50.2	3587.3	41.5	∞
SI	44	825	0.0241	0.0251	-333.7	219.5	∞	95.7	∞
WF	40	954	0.0266	0.0281	-226.7	308.8	∞	120.1	∞
WL	36	711	0.0346	0.0322	**137.1**	50.9	∞	42.3	∞

6.5 Discussion

Data Quality

Departure from HWE was observed in 36 of the 99 population/locus combinations; although was not systematic across all loci in any population. It is therefore likely that null alleles at some loci are the underlying cause of deviation. The markers responsible for the null alleles varied between populations therefore it was not appropriate to remove any specific markers from further analysis. Null alleles frequently occur in bivalves (Carlsson 2008). They are problematic in population differentiation analyses as they appear as an excess of homozygotes, in turn removing a population from HWE and violating one of the assumptions for population analyses. Analyses that rely heavily on the assumption that populations are in HWE were therefore avoided.

Population structure

G''_{ST} was used to investigate population structure and identify the largest appropriate units for management (determined by a significant G''_{ST} value). The FAL sample was significantly differentiated from all the other populations and also had the lowest N_e estimate. This sample showed interesting and complex genetic characteristics that are discussed in Hold (2012). This population had high numbers of full siblings and showed significant linkage disequilibrium (the non-random association of alleles at two or more loci). Greater occurrence of non-random mating and inbreeding serve to reduce the rate of decay of linkage disequilibrium. Opposing gyres (anticlockwise in the west and clockwise in the east, Ferentinos and Collins 1979) cause a physical barrier to dispersal across the bay as well as localised retention within the bay, which may explain the high number of siblings. A high number of siblings leads to reduced genetic diversity within a population and may therefore be the reason for the low N_e estimate. Further discussion will focus on the remaining eight sites.

PCoA plots (Figure 6.5) indicate that BS is the most genetically distinct population, with NC and WF also significantly differentiated from other sites, but not from each other or BS. These three sites are the most northerly, southerly and westerly sites. BS was significantly differentiated ($G''_{ST} > 0.057$) to all other populations apart from NC and WF, supporting the findings of Nicolle *et al.*, (2013) who predicted retention of larvae within the bay, although refuting previous suggestions that the population in the Baie de Seine is a source of larvae for the middle of the eastern English Channel. Larvae from the west and north of the bay disperse to the west and north before being trapped by a gyre, although some may travel as far as the Channel Islands, whereas larvae from the east side of the bay are generally retained (Nicolle *et al.,* 2013). A change of wind direction indicated that larvae may disperse from the east side of BS to the eastern English Channel; however the present results do not provide evidence for this. The BS population, and that of the Baie de St Brieuc to the east, have disparate reproductive strategies which suggests that they are demographically separate populations (Lubet *et al.,* 1995). The evidence described above indicates that the BS population is self-recruiting and should be managed as a distinct unit. The non-significant differentiation between BS, NC and WF raises questions. There is a possibility that larvae reaching the Channel Islands could provide further connectivity to WF and NC, via residual currents occurring between the French coast line and Falmouth bay, which continue northwards around the peninsula of Land's End (Figure 6.6). Other explanations for a similarity in allele frequencies could be attributed to random genetic mutations occurring in those three populations, known as homoplasy or convergence. Also, the current shelf-wide circulation system is likely to have been in existence since the last glaciation, about 8000 years ago (Hill *et al.,* 2008). This reflects a relatively short evolutionary timescale for divergence to occur and represents approximately 1600 generations for *P. maximus*, based on observed age structures in contemporary populations. Therefore, HWE may not yet have been reached during this time period. The speed at which populations diverge relates to N_e; a low N_e leads to more rapid divergence. To establish whether this could relate to connectivity between BS and WF/NC, further samples from intermediate locations are required. Such samples, coupled with hydrodynamic particle tracking models incorporating larval behaviour between the north coast of France and southern Cornwall, would provide further insight.

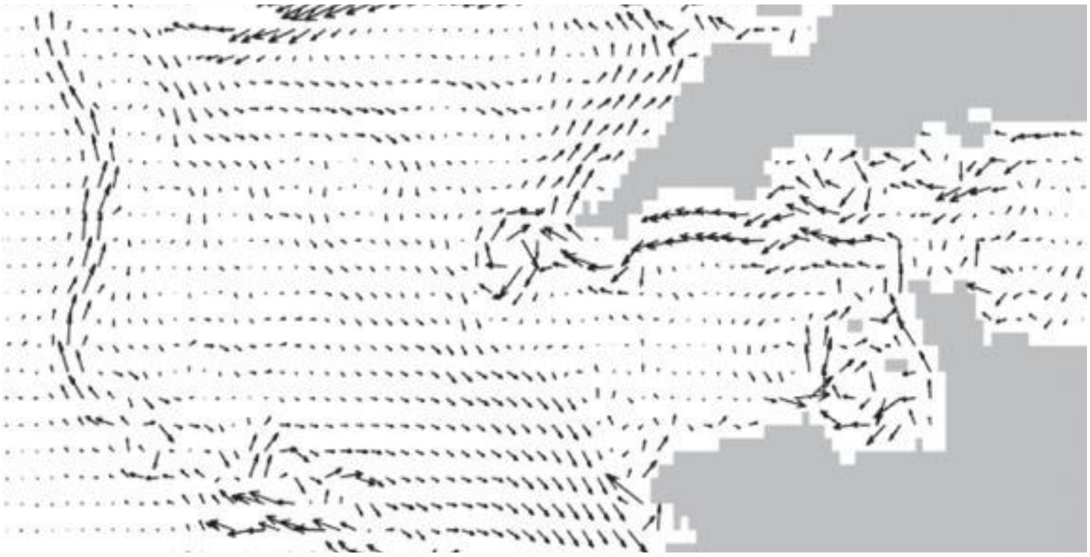


Figure 6.6: Residual flow (21 July–20 August) in the western Channel showing the connectivity of locations by oceanographic currents. The arrows indicate the magnitude and direction of residual flow. From Lee *et al.,* (2013).

NC was significantly differentiated from all sites besides BS and WF, and had the lowest N_e (apart from FAL) of 110 (with an upper CI of 3587). All other sites had either an infinite upper CI (WL) or N_e estimates of >>1000. Therefore NC is likely to have experienced more rapid divergence than the other populations. In populations with a low number of successful breeding individuals the effects of genetic drift (loss of diversity by chance) and migration are amplified and the population is more susceptible to inbreeding depression and over-harvesting (Beebee & Rowe, 2004).

NC also had the lowest diversity (mean allelic richness) apart from FAL, and the lowest number of private alleles. Although larval particle tracking models that link the north and south coasts of Cornwall are not yet available, if the similarity in allele frequencies is due to connectivity rather than homoplasy then it seems more likely that dispersal occurs from NC to WF, than in the opposite direction. This is due to the low N_e found at NC, as just a few individuals can cause homogenisation of allele frequencies so any connectivity from south to north would increase the N_e and genetic diversity of the NC population. Limited contemporary connectivity may occur between NC and WF. But, if they have shared a recent common ancestor and enough time has not yet passed for divergence of allele frequencies to occur, this could cause the observed similarities; although the low N_e in Cornwall would suggest that this is unlikely. Precautionary management suggests that NC should be managed as an isolated population. Fishing effort in this region is low compared to the other sample sites and this may reflect a lower density and smaller census population. There may also be periodic recruitment from the French side of the English Channel. Samples from the Channel Islands and north-western France would provide further information.

There was significant differentiation found between MF and WF (which were both also significantly different from the FAL sample). This is likely due to opposing gyres east and west of the bay, and possibly further complex, as yet undefined, eddies (Ferentinos & Collins, 1979; Hill *et al.*, 2008) which increase local retention of larvae and separate sites on either sides of the bay. MF and FAL are only 8 km apart; therefore a significant G''_{ST} value raises interesting questions about the fine scale oceanographic patterns that would support this level of localised genetic differentiation. High resolution modelling of this area would be needed to investigate this further. This serves to highlight the fact that high connectivity cannot be assumed between adjacent populations with close geographic proximity. MF shows connectivity with the sites further east suggesting that the opposing gyres in Falmouth bay could serve to divide the east and west ends of the English Channel.

WF and MF differed from FAL in that both samples had N_e >>1000, where FAL had the lowest N_e of 27. This could be due to localised retention, or chance sampling of a patch of related individuals. The 'sweepstake effect' is genetic patchiness (variation in allele frequencies) that occurs due to non-random mating, such that a few individuals disproportionately contribute to the next generation. This can be observed over micro-spatial and time scales due to the mosaic nature of habitats and species specific reproductive strategies that occur in the marine environment. Sweepstake recruitment reduces N_e due to the reproductive success of only a few individuals and results in low variation of allele frequencies within a single cohort. Recent studies have indicated a low N_e for certain marine species many orders of magnitude lower than the census size (Hoarau *et al.*, 2005; Hedgecock *et al.*, 2007). If local retention is combined with the sweepstake effect it would be likely that N_e would be low and there would be high levels of relatedness within a population. Hold (2012) found that the sample from Falmouth bay did indeed show low N_e and high relatedness. The presence of differentiation of this sample from MF and WF supports the theory of isolation of this population. As populations with low effective population size and/or low immigration are more susceptible to over-harvesting, consolidation of the degree of connectivity between the western French aggregations and Falmouth bay would serve to inform management of these stocks further.

The G''_{ST} value between EL and WL was negative meaning that the variance and error in the sample produced more noise than the weak signal of differentiation from the data. Due to the location of the two samples within the same bay (and the absence of complex hydrodynamic processes such as those found in Falmouth Bay) it is expected that larval dispersal between them will occur. There was also no significant differentiation found between the following sites: MF, EL, WL, SI and EC. The main current and wind directions are from the west to east up the English Channel (Figure 6.7). These currents serve to link all these sites and with the absence of any strong, localised eddies or gyres, connectivity appears to be maintained. Although Dare *et al.*, (1994) suggest a low level of larval flow occurs from west to east, the migration of only a few individuals per generation is sufficient to homogenise allele frequencies.

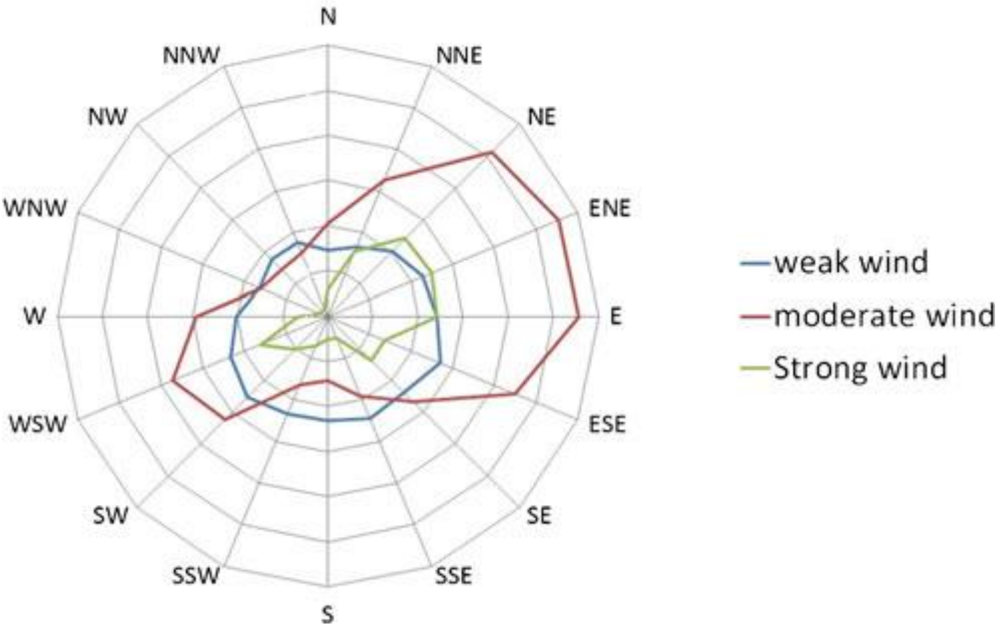


Figure 6.7: Mean wind regime in the English Channel (at the point 0.5 W; 49.5 N) for 2000-2010. Weak wind = <5 m s^{-1}; moderate wind = 5-10 m s^{-1}; strong wind = >10 m s^{-1}. From Nicolle *et al.*, (2013).

Pelagic dispersal distances estimated from genetic data are similar to observed distances, with short pelagic larval durations (PLD). However, as PLD increases, genetic data generally overestimate dispersal distance. This is possibly due to rare individuals dispersing over long distances, thus homogenising genetic differences between populations (Shanks, 2009). Although no transplantation of scallops has been reported on the English side of the Channel; populations in the Channel Islands and the Baie de St Brieuc have received scallops from other parts of the UK. As samples from the latter two areas are not included in the current study, connectivity between these populations and the current samples remains unknown.

Genetic connectivity defines the largest unit at which the stocks in the English Channel should be managed, however demographic differences such as spawning time and duration as well as growth rates act at much smaller spatial scales (Hastings, 1993). Spawning patterns vary over large and small spatial scales. Partial spawnings occur from May to October in the Baie de Seine (Nicolle *et al.,* 2013) and in the western English Channel, spawning occurs as either a single event in May or June, or several events over a protracted period (May to September) depending on the location (CEFAS, 2012). Spawning in the western English Channel is synchronous at spatial scales of <30 km but, beyond this, patterns vary considerably in time and space (CEFAS, 2012). Such factors, coupled with potential low levels of larval transport between scallop grounds are likely to necessitate division of the larger west-east unit. Spawning is thought to be related to an increase in water temperature (Berg & Strand, 2001) therefore due to seasonal variations in temperature and stratification between the east and western English Channel it is likely that different periods of spatial management are required.

Due to the high variability of *P. maximus* spawning behaviour (CEFAS, 2012; Hold, 2012), accurate and realistic modelling of larval transport requires detailed knowledge of spawning patterns over a number of years. Recruitment success is driven by sea temperature and spring phytoplankton availability (Shephard *et al.,* 2010). However, even in optimal tidal and wind conditions enabling dispersal of larvae between two sites, if the time of spawning is not synchronous at both sites, cross-fertilisation will not occur. Effective population sizes were estimated >>1000 for MF, EL, WL, SI and EC but, if connectivity between sites is low, the risk of over-harvesting remains if effort is not managed in relation to the census population size.

Conclusions

The results of the present study generally support previous predictions based on oceanographic and particle tracking models of the study area. The results suggest that there are three populations that were significantly differentiated from the others in the English Channel, but not from each other: the Baie de Seine, north Cornwall and west Falmouth Bay. The identification of the population in the Baie de Seine is significant in terms of management, as previously it has been suggested that this population provides larval input to the large commercial beds present in the mid-eastern English Channel, however the present study indicates that this is unlikely. Along the English side of the English Channel scallop beds are linked by the prevailing currents and winds from west to east, with genetic data suggesting that the opposing gyres in Falmouth Bay may serve to separate populations east and west of this point. There are at least two genetic stocks along the English coast of the English Channel, not including further divisions at a localised scale in Falmouth Bay. Fine scale hydrodynamic models of Falmouth Bay are not currently available but once developed will further inform the management of the stocks. It is clear that management of scallop stocks should occur at spatial scales smaller than genetic structure alone. Genetic information from newer markers that are able to detect local adaptation (e.g. RAD markers) coupled with comprehensive assessment of stocks in each area and more importantly, spawning stock biomass, will help determine appropriate management units and harvesting levels. Additional information on localised spawning events and growth rates will further enhance management to optimise yield, while ensuring sustainability of the stocks.

CHAPTER 7: GENERAL DISCUSSION

King scallops are a high-value resource. Although the biology of the species is well understood (Shumway & Parsons, 2005) until now there have been gaps in our knowledge of aspects regarding their exploitation such as stock connectivity and the environmental impacts of the dredge fishery. The research presented in this thesis increases our understanding of the interactions of the scallop dredge fishery with the habitats and communities where it occurs, and provides evidence of distinct sub-populations within the fishery. The outputs of the thesis can be used to inform and improve management. The study was initiated by the UK scallop industry with the aim of gathering data to work towards ensuring a long-term, profitable and sustainable fishery. Eco-certification is seen as a way of improving public perception of the dredge fishery, ensuring sales security in a global market and demonstrating that the industry is committed to, and willing to engage with scientists and policymakers. Hence, this work provides much of the scientific evidence required for assessing the environmental sustainability of the fishery.

The objectives of this thesis were to enhance knowledge of king scallop populations in the English Channel and provide data regarding the environmental effects associated with the dredge fishery. This was achieved through extensive interaction with the UK scallop industry (fishermen, processors, governmental agencies and industry bodies), utilising their knowledge and expertise to complement robust scientific research methods. Spatial and temporal patterns of scallop fishing activity have been identified (Chapters 2 & 3), larval connectivity between major fishing grounds has been quantified (Chapter 6), bycatch has been assessed (Chapter 5) and compared to bycatch from dredge fisheries around the UK and the habitat impacts of the dredge fishery within the environmental context of the English Channel have been measured against a background of natural disturbance (Chapter 4).

The environmental footprint of the fishery

Previous estimations of fishing effort for towed mobile fishing gears have been spatially limited to certain areas of the English Channel (Vanstaen & Silva, 2010; Vanstaen *et al*., 2010) or temporally limited (Breen *et al*., 2014). Using sightings data from fisheries enforcement vessels provides a fair indication of the location and intensity of fishing activity, however gaps in the data occur in areas that are infrequently or never visited (Breen *et al*., 2014). In the present study, the use of fisher's Local knowledge (LK) has enabled the spatial extent of inshore (<15 m LOA vessels) scallop fleet activity to be mapped across the extent of the English Channel for the first time (Chapter 2). This facilitates identification of areas of high activity and economic importance and is essential information for when impact assessments are undertaken for future Marine Conservation Zones (MCZs). The identification and designation of MCZs is an emotive process for the many stakeholders involved; over 40,000 responses were received by DEFRA during the last consultation (DEFRA, 2013) and a decision on the next tranche of MCZs is due in early 2016.

The spatial footprint of the scallop fishing fleet in the English Channel has altered over the last decade (Campbell *et al*., 2014), due to legislative and economic drivers (Chapter 2). The overall extent of fishing activity has reduced due to effort restrictions, area closures and economic drivers. The mid-eastern English Channel remains an important fishing ground, being subjected to consistently high levels of fishing activity over the last decade. However, alterations in fishing patterns in recent years have left historically productive grounds in the far-western reaches of the English Channel largely untouched due to economic reasons associated with fishing far from a landing port and fishing effort restrictions. Although this is likely to lead to an increase in scallop populations and recovery of the seabed in that area, it implies that pressure has increased on other fishing grounds. As landings of scallops into the UK have more than doubled over the last decade (MMO, 2014), it remains to be seen whether such an increase in fishing effort can be sustained within a reduced spatial footprint.

The maximum dredge limit (14 per side) in Scottish waters has displaced larger vessels into the English Channel where there are no restrictions on the total number of dredges that can be used, therefore greater catches can be achieved. Although the 'Western Waters' legislation (Council Regulation (EC) No 1415/2004) serves to reduce overall pressure on scallop stocks in the English Channel from vessels >15 m LOA, displacement of effort can lead to negative impacts on the stocks, habitat features and other fishing sectors in the recipient areas (Greenstreet *et al*. 2009). The Western Waters legislation causes the displacement of larger vessels away from productive fishing grounds in the English Channel and Irish Sea and into other areas around the UK, such as off the coast of Scarborough. Reports of significant catches in the latter area in 2015 caused an increase in licence requests for the fishery that would have doubled the size of the existing fleet. An impact assessment by the North-eastern IFCA reported that this level of fishing activity was likely to be unsustainable for the scallop stocks, as well as non-target species and the local habitat so an emergency byelaw was implemented to prevent scallop dredging within the 6 NM (nautical mile) zone for a four month period. Hence, the implications of spatial management need to be carefully considered, and management should be strategically designed to negate issues in the fishery, rather than simply shift activity to another area (Horwood *et al*., 1998; Dinmore *et al*., 2003).

In the English Channel, the majority of inshore fishing activity occurs in specific areas, with aggregations of activity occurring in inshore Falmouth Bay and Lyme Bay, and limited activity along the Sussex coastline within the 6 NM zone. There is very little scallop fishing activity between Weymouth in the western English Channel and Beachy Head on the Sussex coastline. In those areas, just a few small vessels target scallops on a limited seasonal basis. LK data provides a reasonable representation of the distribution of inshore scallop fishing activity, although the coarse resolution of the data leads to an overestimation of the total area impacted. Hence, the data should be used with caution in designing spatial management plans and the appropriate scale to be used should be determined by the intended use of the data.

For sedentary species such as scallops, the removal of fishing activity from an area can result in benefits for the stock as well as allowing recovery of the seabed and related biota. (Murwaski *et al*., 2000; Beukers-Stewart *et al*., 2005). Long-term ground closures (6+ years) led to significant increases in scallop biomass on fishing grounds in the north-western Atlantic (Murwaski *et al*., 2000). Such reserves enable scallops to reach a larger size and higher density, result in greater reproductive output and provide protection from the negative physiological and physical effects of being entrained in fishing gear (Collie *et al*., 2000; Kaiser *et al*., 2006). Fishers describe the 'spillover' effect, where scallop biomass at the boundaries of a reserve provides a greater harvestable yield. If closed areas are subsequently reopened, fishers benefit from an increased yield per recruit (Stokesbury *et al*., 2004; Dignan *et al*., 2014). Similarly, ground closures in other areas have reaped considerable benefits to scallop stocks, non-target species and the seabed habitat (Sheehan *et al*., 2014; Howarth *et al*., 2011; Beukers-Stewart *et al*., 2005).

However, careful consideration should be given to the intended management objectives as well as socio-economic considerations. Closure of inshore grounds that have been traditionally, rotationally fished means that inshore fishing activity is likely to be more concentrated in the areas that remain open. Although closed areas potentially provide a refuge for spawning adults (Beukers-Stewart *et al*., 2005; Howarth *et al*., 2015) the benefits of increased reproductive output and larval spill-over will only be seen if there is a sufficient degree of larval connectivity between closed and open areas (Orensanz *et al*., 1991). The environmental context is also relevant when considering ground closures as a potential management tool. Hall-Spencer and Moore (2000) found that scallop dredging had severe and long-lasting impacts on beds of slow-growing coralline algae that had previously not been dredged. However, in more dynamic habitats the impacts of fishing can be low compared to natural levels of disturbance. The Cardigan Bay Special Area of Conservation (SAC) protects habitat features in approximately 1000 km^2 of the Welsh seabed. Scallop dredging is banned in all but a small part of the SAC. However, due to the highly dynamic environment that exists, it is unlikely that scallop dredging causes more disturbance to the seabed than natural perturbations such as that from currents, tides and winter storm events (Sciberras *et al*., 2013). During a recent fishing experiment in the SAC, fishers reported a very dense population of scallops but also a high proportion of dead scallops (Gwladys Lambert, Bangor University, pers. comm). It is not known if the mortality was due to age, disturbance from winter storms, predation or senescence. However, for a potentially productive fishery existing in a habitat that is resilient to fishing disturbance this reflects wasted resource.

Environmental interactions of the scallop dredge fishery

The habitats and biological communities present across the areas exploited by the scallop fleet in the English Channel are dominated by species that are resilient to physical disturbance (Chapter 4). The species assemblages that occur within the footprint of the fishery can be distinguished based on the degree of natural disturbance that occurs on the seabed. It is likely that such communities have been shaped over decadal timescales, due to the existence of the dredge fishery (and the use of other towed bottom-fishing gears) across these fishing grounds during the last century (Kaiser *et al*., 2000; Hall-Spencer & Moore, 2000; Bradshaw *et al*., 2002; Gislason, 1994). As such, the seabed within the fishery is an altered sea-scape that remains productive, rather like agricultural landscapes. Under the MSC guidelines there is no expectation that seabed habitats should be in the state they were decades ago; the environmental impact of a fishery is assessed on the basis of the current status of the habitat (Jodi Bostrom, Marine Stewardship Council, pers. comm.). If a management plan considers and protects vulnerable marine environments (FAO, 2013) and does not cause further damage (e.g. through a further increase in effort or spatial extent) then a fishery would not fail the MSC assessment under this standard and the impacts would be considered to be at sustainable levels.

Scallop fishery bycatch

On average scallops make up over 80 % of the biomass of dredge catches in the English Channel (Chapter 5). The main bycatch species are queen scallops, starfish and crustaceans such as brown and spider crabs. Although bycatch species composition varies with substrate type, environmental conditions and season, the proportion of individual species in the catch is low (for all but five species, <1 % of total catch biomass). The fishery affected a limited number of bycatch species of ecological importance and the population impacts of the dredge fishery on these species are likely to be minimal due to the low proportion retained. Queen scallops comprised the highest mean proportion of bycatch for a single species, contributing on average 6 % to bycatch biomass, although this species is not commercially fished in the English Channel. The latest reform of the European Common Fisheries Policy (CFP) has a commitment to ending the practice of discarding unwanted quota species (European Commission 2013). The regulations came into effect for all pelagic species in 2015 and will apply to demersal species in 2019. Bycatch of quota species such as monkfish, Dover sole, turbot and plaice are all, on average, less than 1 % of the overall scallop dredge catch biomass in the English Channel. This is low compared to many other demersal mixed fisheries (Enever *et al*., 2007), similar to scallop dredge catches in Cardigan Bay and considerably less than on fishing grounds around the Isle of Man. Regular by-catch sampling schemes are necessary to inform spatial and temporal variation in by-catch abundance and possible impacts on community structure (Allen *et al.,* 2002; Borges *et al.,* 2004; Craven *et al.,* 2013). Future sampling must occur over a high enough spatial resolution to incorporate changes in benthic assemblages and distinct fishing grounds, and at a frequency that will account for variations in annual recruitment of by-catch species, or seasonal/spatial variations in abundance (Veale *et al.,* 2001; Craven *et al*., 2013).

Population structure

In the past, oceanographic evidence has suggested that scallop larval exchange is likely to occur between scallop stocks on the French and English sides of the eastern English Channel (Dare *et al.,* 1994; CEFAS, 2012; Nicolle *et al.,* 2013) (Chapter 6). However, the present study revealed that the scallop population in the Baie de Seine is in effect reproductively isolated from the wider eastern English Channel. This fishery is dependent on the recruiting year class each year (Eric Foucher, IFREMER, pers. comm.), therefore it is imperative that exploitation is managed in accordance with informed predictions of the strength of the recruitment. Genetically distinct populations occur on either side of Falmouth Bay due to complex local hydrodynamics that create a physical barrier to larval dispersal. One sample in Falmouth Bay also showed a very low effective population size, meaning it will be susceptible to over-harvesting. If the density of breeding adults becomes too low, fertilisation will not occur (Allee, 1932; Orensanz *et al*., 1991) and populations are at risk of inbreeding which can cause decreased hatching, growth and survival in scallops (Crow 1986). Scallop populations in Lyme Bay and the wider eastern English Channel are genetically undifferentiated, therefore they can be considered as a single management unit. Genetic evidence of population structure defines the largest scale at which scallop stocks should be managed, however in reality much smaller management units may be required when taking into account reproductive variability and effective population size. Future work should incorporate further samples from along the French coast and the Channel Islands to investigate the observed link between scallop populations in the Baie de Seine and western Cornwall. Fine-scale hydrodynamic models for the English Channel would also provide further evidence and these are currently in development at Bangor University.

Management recommendations

Under the EU MFSD, the overarching objective is to achieve good environmental status (GES) of marine waters by 2020 (EC, 2008; Rice, 2001). Key motivations underlying the MFSD are: conservation of habitats and marine resources to ensure long-term sustainability; ensuring that socio-economic activities can be maintained; and protecting sensitive species or features. The MSFD recognises the need for defining criteria and characteristics of GES, as well as environmental targets. There is also a substantial need to develop scientific understanding of marine ecosystems and fishery interactions in order to support an ecosystem-based approach to fisheries (EAF) (Ecosystem Principles Advisory Panel, 1999; Browman *et al*., 2004).

Closed areas provide a number of environmental and economic benefits (Grantham *et al*., 2003; Beukers-Stewart *et al*., 2005) and can act as a buffer to overcome unexpected changes in fisheries such as recruitment failure or a change in fleet behaviour. Closed areas have afforded significant ecological and financial benefits to the Atlantic sea scallop fishery (Murwaski *et al.*, 2000) and are a feasible option for the English Channel fishery. Although the maximum extent of the fishery covers a large proportion of the English Channel, specific areas of concentrated fishing activity have been identified. These represent grounds that fishers return to year after year due to consistently good catch levels. The cyclical, dynamic nature of fleet activity means that a large proportion of the seabed may be visited and fished at frequencies >1 year. For mobile-gear fishing fleets in southwest England and Wales, 90 % of activity occurs in 50 % of the total area impacted (Jennings & Lee, 2012). Therefore, if fishing activity was removed from the outer margins of the fishery that experiences <10 % of overall fishing activity, a relatively large proportion of seabed could be protected with minimal displacement of activity. A secondary benefit may be a reduction in interactions of the dredge fishery with other gears and/or sectors (Hart 1998; Kaiser *et al*. 2000). However, limitations of closed areas include: a lack of protection from environmental perturbations such as intense storms or pollution (Allison *et al.,* 1998); size limits or lack of adequate scientific basis in designation (Allison *et al*., 1998); and lack of enforcement (Gullett, 2003). Closed areas do not address other issues in fisheries such as overcapacity, which is seen as the greatest threat to the sustainability of the scallop fishery in the UK.

Grafton *et al*. (2010) argue that focussing on EAF alone fails to account for fisher behaviour and proposed incentives-based approaches to fisheries management (IAFs) to supplement EAFs and promote ecological and economic sustainability. These involve using either individual harvesting rights, community-based rights or territorial user rights to incentivise more sustainable fishing behaviours (Baskaran & Anderson, 2005). This would be a novel approach to scallop fishery management in the English Channel, although has been successful in other scallop fisheries globally (Beukers-Stewart & Beukers-Stewart, 2009). Fishers with secure harvesting rights are better incentivised to protect the long-term sustainability of the resource and are therefore more likely to display sustainable fishing behaviour, work collectively to improve science and management (Rice, 2001), self-police the fishery and maximise returns from the fishery. It is important to assess whether the current levels of exploitation of the English Channel scallop fishery are economically sustainable for the current fleet size. Overcapacity in a fishery causes over-exploitation and poor economic returns. If entry to a fishery is limited, over-exploitation is less likely to occur and resource ownership provides greater compliance (Grafton *et al.,* 2010). However, measures to reduce overall fishery mortality have the potential for severe socio-economic consequences (Repetto, 2001).

Trade-offs to meet conflicting objectives may be required to decide which of several management options enables a feasible compromise to be made (Sainsbury *et al*., 2000). Management evaluation must include a monitoring programme, along with specification on what data will be gathered, how it will be used, decision 'rules', performance indicators and strategies for implementation (Sainsbury *et al*., 2000). Mechanisms to weigh up the costs and benefits of different management processes have been developed, such as Management Strategy Evaluation (Sainsbury *et al*., 2000), however applications to date have involved relatively simple ecosystems. Incorporating multispecies interactions and accounting for uncertainty pose much more complex issues that are often poorly understood (Sainsbury *et al*., 2000). In addition to this, management evaluation strategies are often developed in retrospect to policies for ecosystem-based fisheries management (EBFM) (Smith *et al*., 2007; Link *et al*., 2011).

Industry engagement

Industry-science collaborations are not without challenges (Lordan *et al*. 2011). The present study was initiated and funded by the UK scallop industry, demonstrating the desire to contribute to the science and governance of the fishery. The industry aspires to lead management, rather than be restricted by regulations that have been imposed in a top-down approach without prior consultation. Improved international strategic collaboration between scientists would also serve to enhance the development of knowledge and the ability to define and implement appropriate marine governance (Mackinson *et al.*, 2011). This can be a complex process in a large area such as the English Channel, with numerous stakeholder countries involved (Glegg *et al*., 2015).

A major obstacle to instigating legislative changes is requirements under EU law. The industry would like an increase in the UK annual Western Waters effort allowance. For this to be considered, the UK Administration would need to present a case to the EU court, backed-up with scientific evidence and a robust stock assessment. Evidence would need to show that any increase in the rate of harvesting would not exceed MSY, or have significant impacts to habitats or other fisheries. Under the reformed CFP, member states have a commitment to fish all stocks to MSY by 2020. An industry self-sampling scheme was launched in 2011 with the aim of collecting sufficient data across the extent of English waters to enable a stock assessment (Bell *et al*. 2014). English waters were divided into seven zones for assessment. Skippers collected samples of scallops during normal fishing activity and the flat shells were retained during processing to be measured and aged. A minimum of six samples per zone, per quarter was required to provide enough data. Unfortunately, participation levels remained low with six vessels contributing > 50 % of samples over the

2.5 year period the scheme was in operation. A high proportion of samples came from inshore areas (specifically Cornwall and Lyme Bay). Hence, these are the only two areas where tentative conclusions about stock levels could be made (Bell *et al.*, 2014. The scheme was concluded in 2013 and new methods of sampling are currently being investigated that will hopefully prove successful.

The Scallop Industry Consultation Group was formed in 2012, primarily in response to the issue of Western Waters effort limitations, following the closure of the area VII scallop fishery for a month in 2011 after the fleet exceeded the annual effort allocation for the year. The quarterly meetings are attended by scallop fishers, processors, representatives of industry bodies such as the Scallop Association and the Scottish Fishermen's Federation and representatives from each of the devolved Administrations. The group discuss what the effort allocation (as kW days) should be for the next three months. The final decision is made by DEFRA and issued to fishing vessels as a licence variation by the MMO; however the Administration actively encourage input and debate from the industry group. The forum also provides an opportunity to discuss potential quota swaps. To date, deals to secure additional effort for the fleet have been negotiated with the Netherlands, France, Belgium and Ireland. Other topics covered during industry meetings include non-compliance by vessels, issues in the queen scallop fishery (which can have a knock-on effect on the king scallop fishery), issues with the electronic recording system used by the MMO to track effort uptake, available data for the fishery, generation of finance to obtain effort from other EU administrations and the formulation of smaller working groups to discuss and tackle issues such as latent effort and gear selectivity improvements. Attendance at meetings is high, with 30-35 stakeholders present at each meeting, indicating the motivation of the wider industry to be involved in management decisions rather than experience enforced, top-down management.

Conclusion

The initial motivation for the research presented in this thesis was to provide data required for the scallop fishery to enter the MSC eco-certification process. MSC Principle 1 requires good understanding of stock status. Although a stock assessment is still lacking, at least three reproductively distinct sub-populations have been identified within the fishery providing information on stock structure and appropriate spatial management units. Principles 1 and 2 require a good understanding of spatial and temporal patterns of fishing activity. The in depth analysis of VMS data presented in Chapter 3, coupled with the LK data gathered from the industry (Chapter 2) provide a comprehensive dataset for the activity of both the inshore and offshore fleets over the last decade. This is the first time inshore scallop fishing activity has been mapped across the full extent of the English Channel. Stipulations under Principle 2 are that fishing activity should not cause irreversible harm to the ecosystem or gross changes in habitat or ecosystem function. The results of the present study provide robust evidence of the prevalence of bycatch in the fishery and suggest that impacts on single species populations are low. The study also failed to demonstrate an impact of the intensity of scallop dredging on species community or biological trait composition across the English Channel with the habitats in their present, albeit potentially altered state. The MSC criteria also state that vulnerable habitats should be avoided and an understanding of recovery timescales is imperative. These aspects can be addressed by management of the fishery at appropriate spatial and temporal scales.

Throughout the duration of this project, a number of issues have afflicted the UK scallop industry. Effort restrictions in ICES area VII, latent effort and area closures have all been high on the agenda during industry discussions (author observations). Hence, at the present time it is unlikely that the industry will prioritise eco-certification of the fishery as focus is shifted towards more pressing concerns. There are other bodies in the UK, such as the SFP (Sustainable Fisheries Partnership), which offer Fisheries Improvement Projects (FIPs) as an alternative to MSC assessment, with less demanding data and financial requirements. Fisheries can benefit from such schemes when MSC assessment is too expensive or restrictive (UNEP, 2009). However, regardless of this the present study has provided many tangible benefits to the fishery. A number of important data gaps have been addressed and the outputs provide a solid foundation on which to design improved management measures which can benefit scallop stocks, habitats and the industry.

Key findings of the study are:

1. Fishing activity varies spatially and temporally. There are certain areas in the English Channel that are fished consistently year on year, whereas other areas are visited sporadically on timescales of <1 year. Temporal and spatial variation in fishing activity is driven largely by economics and legislation.
2. The habitats and species assemblages within the spatial extent of the fishery may exist in an altered state due to prolonged historical fishing activity. Communities are

dominated by species that are resilient to disturbance. The effects of recent fishing activity are not detectable over a background of natural physical disturbance.

3. The incidence of individual bycatch species is low compared to other fishing gears and other scallop fisheries in the UK. Impacts on populations of commercial species are likely to be insignificant. However, continued monitoring of bycatch at appropriate temporal and spatial scales will aid protection of species that are more vulnerable at certain times of the year.
4. At least three reproductively isolated sub-populations of scallops exist in the English Channel. These populations represent the largest spatial scale at which stocks should be managed; however variations in reproductive timing and growth rates suggest that management boundaries should occur at smaller spatial scales.

APPENDICES

Appendix 1.1

Existing fishing management measures and restrictions relating to scallop fishing in the English Channel.

<table>
<tr><th></th><th></th><th>< 6 NM</th><th>6-12 NM</th><th>up to and beyond 12 NM (within territorial waters)</th></tr>
<tr><td rowspan="6">England</td><td>Cornwall IFCA</td><td>• Min. internal ring diameter of 75 mm.
• Min. mesh size of 100 mm if a retaining bag is used.
• Max. 12 dredges in total.
• Max tow bar length 5.18 m.
• Max of 2 tow bars used at once.
• Max vessel length 15.24 m.
• Curfew between 1900-0700</td><td rowspan="6">Max. 8 dredges per side</td><td rowspan="6">Minimum landing size 110 mm (VIId) or 100 mm (VIIe)
Scallop dredges must:
• have a moveable spring-loaded tooth bar and belly bar;
• not exceed 85 cm in width;
• not contain any attachments;
• not exceed 150 kg weight
• not have > 8 rows of belly rings
• max. 9 teeth on tooth bar (VIIe)
• max. 8 teeth on tooth bar (VIId)
• max. tooth width 22 mm if dredge > 80 cm wide
• max. tooth width 12 mm if dredge < 80 cm wide</td></tr>
<tr><td>Devon & Severn IFCA</td><td>• Max. 12 dredges in total.
• Closed season July to September.
• <15.24 m vessels within 3 NM.
• Curfew between 1900-0700. Closed areas.</td></tr>
<tr><td>Southern IFCA</td><td>• Max. 12 dredges in total.
• No more than two tow bars used at once.
• Max tow bar length 5.18 m.
• Curfew between 1900-0700.
• Max vessel length 12 m</td></tr>
<tr><td>Sussex IFCA</td><td>Max. 12 dredges in total.</td></tr>
<tr><td>Kent & Essex IFCA</td><td>Max 12. dredges in total.</td></tr>
<tr><td>Isles of Scilly IFCA</td><td>• Vessel ≤ 10 tonnes.
• Max vessel length 11 m.
• Closed areas.
• Fishing permit required.</td></tr>
<tr><td>Scotland</td><td></td><td>Max. 8 dredges per side.</td><td>Max. 10 dredges per side</td><td>• Maximum 14 dredges per side.
• French dredges banned.
• Minimum landing size 110 mm.</td></tr>
</table>

Wales		• No scalloping <1 NM from shore. • From 1-3 NM only vessels <10 m and with max. 6 dredges in total permitted. • From 3-6 NM max. 8 dredges in total.	max. 14 dredges in total	• <221 kW engine • Fishery closed 1 May - 31 October • All scallop vessels must carry a working VMS • Minimum landing size 110 mm. Scallop dredges must: • have a moveable spring-loaded tooth bar and belly bar • be ≤85 cm in width • not contain any attachments • not exceed 150 kg weight • have ≤ 7 rows of belly rings • max. 8 teeth per tooth bar • max. tooth width 22 mm/length 110 mm
Isle of Man			Some closed areas.	• French dredges are banned. • Fishery closed 1 May to 31 October. • All scallop vessels must carry a working VMS. • Minimum landing size 110 mm • Maximum kW days fishing allocation for vessels >15 m in length (Area VII) Aggregate scallop dredges must: • not exceed 762 cm (25 feet) in width within 3 NM • not exceed 1067 cm (35 feet) in width within 12 NM • have max. 9 teeth per tooth bar • have a tooth spacing of ≥75 mm • have a mesh size of ≥100 mm in the netting cover • have a belly ring internal diameter ≥75 mm • not have a tow bar >185 mm in diameter

Northern Ireland			• Max. 6 dredges per side. • Curfew from 2000-0600. • No weekend fishing. Closed season. • Max. aggregate dredge width 915 cm. • Max tow bar length 5.5 m. Closed areas.	French dredges are banned Maximum kW days fishing allocation for vessels >15 m in length (Area VII) Scallop dredges must: • not exceed an aggregate width of 1219 cm (40 feet) • have max. 9 teeth per tooth bar • have a tooth spacing of ≥75 mm • have a belly ring internal diameter ≥75 mm • have a mesh size of ≥100 mm in the netting cover
Jersey				Dredge ring internal diameter ≥85 mm. Dredge attachments banned. Max 16 dredges. Aggregate mouth size of dredges ≤12.8 m.
Guernsey		• Max. 12 dredges within 3 NM. • From 3-6 NM, max 8 dredges 3-6 N and max. tow bar length 4 m.	• From 6-12 NM, max. • 12 dredges and max tow bar length 5.8 m.	

Appendix 2.1

English Channel Scallop Fishery Survey

Thank you for participating in this questionnaire. The aim is to increase knowledge about the English Channel scallop fishery and the information will be used to support the Scallop Association and its members in the sustainable management of the fishery.

Do you have any questions before we begin........?

Gear information

1. **Gear type used:** Newhaven / other (***please specify***)…………………

What is the:

- Gear width...
- No. of dredges used...
- Dredge tooth spacing..
- Belly ring size..
- Tooth length..

2. Do you plan to increase or decrease **engine size** in next 12 months? **Y/N** (please give details)

…………………………………………………………………………………………………

3. Have you increased or decreased **engine size** in the last 10 years? **Y/N** (please give details)

…………………………………………………………………………………………………

4. Do you plan to increase or decrease no. of dredges used in next **12 months**? (please specify)

………………………………………………………………………………………………

5. Have you increased or decreased no. of dredges used in the last **10 years**? (please specify)

…………………………………………………………………………………………………

Please answer the following questions in relation to your fishing habits in 2011:

6. On average, how many hours a day did you fish?........................hours
7. Approximately how many days did you fish?..............................days
8. What is your average tow time? ....mins
9. What is your average tow speed? ....knots
10. What was your average catch per day (bags)?.......................................
11. What was the average bag weight/size?...
12. What was your average trip length (days)?.....................................days

Location of fishing

Fish Map software used to record areas fished and number of days per month, main by-catch landed no. of years fished, importance of grounds.

13. What are the three most important factors that influence **where** you decide to fish?

For example: Weather (e.g. strong winds), vessel's total catch in that area in previous year, Condition of scallops, Distance from port, Cost of fuel, Number of other fishing vessels present on grounds

i. ……………………………………………………………
ii. ……………………………………………………………
iii. ……………………………………………………………

14. What wind strength prevents you from fishing?..

15. How do you decide **where** you will fish? (*Please tick all that apply*):
    - Skippers knowledge/experience..
    - Sharing knowledge with other boats/fishermen.........................
    - Prospecting for new grounds..
    - Other (*please specify*)...

16. Approximately what percentage of your fishing each year is in the **same** areas as the previous year, and what percentage is in **new / different (occasional)** areas?

Same: ....% **New / different areas**: ....%

17. If you fish in **different** grounds to 'normal', what are the 3 main **reasons** for this (*in order)?*
    i. ……………………………………………………………………………………
    ii. ……………………………………………………………………………………
    iii. ……………………………………………………………………………………

18. Do you spend time **prospecting for new scallop beds?** If yes, approximately how many days per year do you spend doing this?

Yes / No **Number of days per year**…………………………………….

19. If there are grounds that you fish on a rotational basis e.g. once every 2 or 3 years, what are the reason(s) for this?

……………………………………………………………………………………………
……………………………………………………………………………………………

20. In the last **10 years** have there been any **area based** legislative reasons (e.g. area closures) that have affected **where or how** you would normally fish?
    - Location(s)..
    - Why/how affected?..
    ....
    ....
    ....

21. In the last **10 years** have there been any **technical** legislative reasons (e.g. gear/engine size/effort restrictions, curfews) that have affected **where or how** you would normally fish?
    - Location(s)?..
    - Why/how affected?..

....

....

....

22. Thinking about the last 10 years, how far would you **normally** travel from your home port to fish? (*please state a range e.g. 50-200nm*) ....**nm**

23. What is the maximum distance you are **willing** or **able** to travel to fish?.............................**nm**

24. In the last **10 years** have you **needed** to travel **further than normal** from your home port to fish?

No 0-12nm 12-50nm 50-100nm 100-200nm >200nm

When and why? (*e.g. fuel cost / scallop abundance/ restrictions)*

....

....

....

If yes, where did you go?..

25. Where do you do the majority of your fishing? **0-3nm 3-6nm 6-12nm 12+nm**

26. Has the way you fish for scallops changed in any other way over the last **10 years**?

………………………………………………………………………………………

………………………………………………………………………………………

………………………………………………………………………………………

………………………………………………………………………………………

Catch composition & condition

Please indicate on the paper map if you are aware which month(s) spawning occurs in a particular area and provide the following information if possible:

27. Does spawning occur at the same time of year in the area?

a) Yes it varies by less than a week
b) No, it can vary by 2 or 3 weeks
c) No, it can vary by a month or more

28. Do the majority of scallops in this area all spawn at approximately the same time or does it occur over a longer time period?

    a) Yes, most scallops spawn within a day or two of each other
    b) Most scallops spawn within a week of each other
    c) No, the spawning carries on for longer than a week

29. Are there any apparent triggers for spawning? (*e.g. light, temp, sediment, water clarity, tides*)
    …………………………………………………………………………………………
    …………………………………………………………………………………………

30. Does the timing of spawning influence where you decide to fish? (*Please state how/why*)
    …………………………………………………………………………………………
    …………………………………………………………………………………………

Landings & Profitability

30. In the last **10 years** has your **overall catch** increased or decreased?

>50% less ***0-50% less*** ***same*** ***increased 0-50%*** ***increased >50%***

Please give possible reasons for this…………………………………………………………..
…………………………………………………………………………………………
………

31. If possible please say **which years** were:
    particularly **good** (large catch):……………….........

particularly **poor** (small catch):……………………

32. In the last **10 years** has your average **catch weight per tow** of MLS scallops increased or decreased?

>50% less ***0-50% less*** ***same*** ***increased 0-50%*** ***increased >50%***

Please give possible reasons for this……………………………………………

……………………………………………………………………………………

33. What is your **minimum** commercially viable catch rate?
    a. Bags per trawl...........................
    b. Bags per day.............................

***By answering the following questions you will help place an economic value on the areas that you fish*:**

34. For your fishing activity in **2011** please give an indication of:

- your annual gross **landings** (tonnes)……………………........./ prefer not to answer
- the value of your annual **landings** (£)……………………..../ prefer not to answer
- your annual **profit** (£)………………………………………./ prefer not to answer

35. Please estimate the percentage (%) difference between **2011** and **2001**:

- % change in your annual gross **landings** (tonnes)……............% increase / decrease
- % change in the value of your annual **landings** (£)……………% increase / decrease
- % change in your annual **profit** (£)……………………………% increase / decrease

Management

36. Please answer these 3 statements questions using the following scale:

Strongly disagree ***Disagree*** ***Neither agree or disagree*** ***Agree*** ***Strongly agree***

i. The fishery is currently fished at a sustainable level..

Please give a reason for your answer..

ii. The fishery is **at risk** of being overfished...

Please give a reason for your answer...…

37. In your opinion, please indicate the **three most effective** ways of conserving scallop stocks for the future (i.e. fishing sustainably), in order of effectiveness (1-3):

- Dredges per side limits.......
- New dredge design.........
- No. of teeth........
- Belly ring size........
- Vessel size limits........
- Engine size limits........
- Minimum landing size.......
- Permanent closed areas.......
- Seasonal closures.....
- Curfews.......
- TACs........
- Restricted effort........
- Caps on licences.........
- Other (*please specify*)...

38. Do you disagree with any of the current management measures in the English Channel?

…………………………………………………………………………………………………
…………………………………………………………………………………………………
…………………………………………………………………………………………………

39. Are there any other comments you would like to make? (*Continue on separate sheet if necessary*)

…………………………………………………………………………………………………
…………………………………………………………………………………………………
…………………………………………………………………………………………………

Appendix 2.2

Summary of vessel characteristics. Vessels are grouped by length (<15 m; 15-25 m and >25 m LOA). S.D. = one standard deviation of the mean.

	<15 m LOA			15-25 m LOA			>25 m LOA		
	min	**max**	**mean (S.D.)**	**min**	**max**	**mean (S.D.)**	**min**	**max**	**mean (S.D.)**
total dredges	6	16	9.86 (2.5)	16	24	20.9 (3.2)	24	36	31.3 (4.5)
hours per day	8	24	15.2 (5.5)	24	24	24 (0.0)	24	24	24 (0)
days fished scallops in last 12 months	0	337	118.6 (75.8)	100	240	150.7 (54.0)	100	320	222.7 (60.8)
min tow duration (minutes)	15	100	50.3 (19.0)	30	90	56.4 (18.4)	30	60	54.1 (9.7)
max tow duration (minutes)	40	180	82.0 (38.8)	45	120	77.1 (28.0)	60	120	75.0 (19.1)
min tow speed (knots)	1.5	3.0	2.4 (0.38)	2.0	2.6	2.3 (0.2)	1.9	3.0	2.5 (0.3)
max tow speed (knots)	1.5	3.5	2.7 (0.47)	2.0	3.0	2.5 (0.3)	1.9	3.5	2.7 (0.5)
min trip length (days)	1	5	1.2 (0.77)	4	7	6 (1.2)	2	7	5.7 (1.5)
max trip length (days)	1	5	1.2 (0.77)	4	8	6.1 (1.3)	5	8	6.5 (0.9)
max wind (knots)	4	6	5.38 (0.62)	5	10	7.1 (1.7)	6	10	8.2 (1.3)
% same ground fished each year	20	100	87.6 (16.34)	50	100	80.7 (23.0)	0	99	76.7 (27.1)
max distance travelled	9	1000	300.8 (385.15)	150	1000	878.6 (321.3)	300	1000	881.8 (263.9)
vessel length (m)	9.8	15.0	11.7 (2.0)	18.3	25.0	22.8 (2.3)	26.0	40.0	32.6 (3.8)
engine power (kW)	93	300	154.7 (61.7)	221	671	447.1 (173.1)	480	880	690.5 (125.8)
min crew	1	4	2.2 (0.7)	3	5	4.3 (0.8)	4	7	5.5 (0.9)
max crew	1	6	3.0 (1.1)	4	7	5.7 (1.0)	6	9	6.8 (1.0)
skipper age	27	64	48 (11.9)	24	53	43.3 (10.8)	28	57	43.3 (9.3)

Appendix 4.1

Sample site coordinates and physical parameters

Site	Station	Latitude	Longitude	BSS (N m^{-2})	Sand_grv (%)	Depth (m)	Chl-*a* (mg m^{-3})	Stratification	T_{mean} (°C)	T_{range} (°C)	T_{varib} (°C)	FI (h 3yr^{-1})	Inert (kg ha^{-1})
1	1.1	50.6843	-5.211	0.80	26.2	56	1.092	0.195	11.37	7.94	1.92	4.00	56.92
1	1.2	50.7040	-5.2547	0.76	18.7	57	1.042	0.237	11.15	7.87	1.94	5.77	244.95
1	1.3	50.6741	-5.2652	0.76	81.4	59	0.982	0.237	11.15	7.87	1.94	4.05	23.58
1	1.4	50.6661	-5.1868	0.82	100.0	56	0.974	0.195	11.37	7.94	1.92	6.25	467.42
1	1.5	50.7223	-5.2051	0.80	95.5	60	1.033	0.195	11.37	7.94	1.92	7.70	97.07
2	2.1	50.0246	-4.7060	0.24	50.6	72	0.921	0.427	10.66	8.20	2.01	9.67	11.18
2	2.2	50.0152	-4.6652	0.24	35.9	72	0.842	0.427	10.66	8.20	2.01	10.75	90.42
2	2.3	50.0089	-4.6227	0.26	0.0	72	0.847	0.427	10.52	8.27	1.99	53.94	55.17
2	2.4	49.9899	-4.6733	0.29	52.7	75	0.823	0.564	10.16	7.82	1.91	19.44	48.94
2	2.5	49.9941	-4.7337	0.28	46.6	73	0.833	0.564	10.16	7.82	1.91	27.08	100.12
3	3.1	49.8536	-3.2595	0.97	2.0	58	1.300	0.491	10.34	8.05	1.88	13.68	219.15
3	3.2	49.8471	-3.2136	1.00	73.6	51	1.308	0.453	10.43	8.23	1.89	8.03	40.10
3	3.3	49.8689	-3.1392	1.04	22.8	50	1.395	0.453	10.43	8.23	1.89	15.96	167.91
3	3.4	49.8422	-3.1828	1.01	0.0	61	1.292	0.453	10.43	8.23	1.89	23.52	322.42
3	3.5	49.8672	-3.2164	1.00	43.6	54	1.265	0.453	10.43	8.23	1.89	6.43	527.36
4	4.1	50.3224	-3.1257	0.58	89.5	51	1.296	0.071	11.73	9.61	2.00	52.33	3.02
4	4.2	50.2842	-3.1750	0.61	37.2	52	1.250	0.071	11.73	9.61	2.00	70.29	8.41
4	4.3	50.3192	-3.1733	0.57	25.4	51	1.291	0.071	11.73	9.61	2.00	47.67	7.88
4	4.4	50.2698	-3.1066	0.70	73.3	52	1.308	0.071	11.73	9.61	2.00	18.33	17.18
4	4.5	50.2905	-3.0396	0.69	37.8	50	1.371	0.051	11.73	9.61	2.00	51.15	9.53
5	5.2	50.4443	0.1596	1.33	10.0	50	1.503	0.031	11.91	10.50	1.98	61.72	1779.06
5	5.3	50.4264	0.0790	1.31	1.0	53	1.353	0.027	11.98	10.34	1.95	58.05	814.70
5	5.5	50.4022	-0.0097	1.48	0.0	43	1.329	0.027	11.99	10.29	1.95	53.46	513.92
6	6.1	50.4428	-0.2047	0.84	0.0	46	1.326	0.031	11.93	10.47	1.96	92.64	299.43
6	6.2	50.4316	-0.2799	0.75	0.0	39	1.420	0.020	12.02	10.21	1.95	18.83	271.44
6	6.3	50.4164	-0.2391	0.85	0.0	37	1.386	0.020	12.02	10.21	1.95	70.65	132.38
6	6.4	50.4234	-0.3386	0.79	0.0	35	1.498	0.020	12.04	10.13	1.95	65.89	2177.77
6	6.5	50.3879	-0.3082	0.76	0.0	35	1.388	0.020	12.02	10.21	1.95	53.36	561.28
7	7.1	50.5280	0.5673	1.31	0.0	18	1.579	0.022	11.95	10.52	2.00	35.29	25.64
7	7.2	50.5283	0.6460	1.09	30.6	24	1.559	0.022	11.95	10.52	2.00	23.22	114.73
7	7.3	50.4860	0.5496	1.70	0.0	41	1.524	0.025	11.95	10.52	2.00	36.62	1011.37
7	7.4	50.4869	0.6260	1.33	0.0	51	1.577	0.022	11.95	10.52	2.00	29.64	772.01
7	7.5	50.4918	0.6752	1.50	0.0	36	1.659	0.022	12.05	10.56	2.02	36.00	410.52
8	8.1	50.6274	-0.6224	1.11	13.8	48	3.820	0.012	11.84	10.36	2.00	2.00	263.82
8	8.2	50.6167	-0.4765	1.11	0.0	48	3.591	0.023	11.86	10.52	2.00	20.66	485.27
8	8.3	50.5258	-0.6143	1.16	53.8	50	3.009	0.025	11.96	10.28	1.96	6.47	61.19
8	8.4	50.5656	-0.4965	1.13	31.4	50	2.309	0.023	11.86	10.52	2.00	2.00	44.18
8	8.5	50.5624	-0.5757	1.19	0.0	51	2.700	0.012	11.84	10.36	2.00	7.08	179.38

Appendix 4.2

References for biological traits matrix

Literature

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Appendix 4.3

SIMPER output using square-root transformed beam trawl biomass data giving the average similarity in species composition for stations grouped by environmental characteristics identified by a PCA, 'Deep' (sites 1, 2 & 3) and 'Shallow' (sites 4, 6, 7, 8). Av.Abund = Average biomass per hectare. Taxa which contributed to 80 % (cumulative) of the similarity within groups are shown. Sim/SD is the similarity/standard deviation ratio. Sim/SD values of <1.3 indicate greater variation within stations than between and therefore those taxa are not considered to be a reliable representative of the similarity within the group.

Deep **Average similarity: 28.96 %**	**Av.Abund**	**Av.Sim**	**Sim/SD**	**Contrib%**	**Cum.%**
Pagurus spp.	1.75	6.58	1.84	22.73	22.73
Aequipecten opercularis	1.38	2.23	0.53	7.70	30.43
Asterias rubens	0.91	1.86	0.70	6.44	36.87
Alcyonium digitatum	1.09	1.76	0.74	6.08	42.95
Psammechinus miliaris	0.83	1.60	0.73	5.54	48.49
Inachus spp.	0.49	1.56	0.90	5.40	53.89
Macropodia spp.	0.40	1.52	2.10	5.23	59.12
Ciona intestinalis	1.12	1.19	0.37	4.11	63.23
Nemertesia spp.	0.41	1.05	0.74	3.63	66.87
Callionymus spp.	0.51	1.03	0.54	3.55	70.41
Cellaria spp	0.94	1.02	0.30	3.51	73.92
Ophiura spp.	0.33	0.91	0.86	3.16	77.08
Alcyonidium diaphanum	0.36	0.55	0.66	1.90	78.98
Caryophyllia smithii	0.38	0.54	0.31	1.88	80.86
Shallow **Average similarity: 41.74 %**					
Aequipecten opercularis	2.90	10.11	1.00	24.22	24.22
Pagurus spp.	1.83	7.52	3.12	18.02	42.24
Asterias rubens	1.84	6.49	1.46	15.55	57.80
Psammechinus miliaris	1.51	6.28	2.82	15.06	72.85
Liocarcinus spp.	0.64	1.33	0.93	3.19	76.05
Ophiothrix fragilis	0.37	1.21	0.94	2.89	78.94
Alcyonium digitatum	0.68	1.16	0.61	2.77	81.71

SIMPER output using square-root transformed beam trawl biomass data for stations grouped by environmental characteristics identified by a PCA, 'Deep' (sites 1, 2 & 3) and 'Shallow' (sites 4, 6, 7, 8). Taxa which contributed to 80 % (cumulative) of the dissimilarity between groups are shown. Av.Abund= Average biomass per hectare. Diss/SD is the dissimilarity/standard deviation ratio. Diss/SD values of <1.3 indicate greater variation between stations than between groups and therefore those taxa are not considered to be a reliable representative of the dissimilarity between groups.

Average dissimilarity = 70.89 %	**Deep Av.Abund**	**Shallow Av.Abund**	**Av.Diss**	**Diss/ SD**	**Contrib %**	**Cum. %**
Aequipecten opercularis	1.38	2.90	7.60	1.24	10.72	10.72
Asterias rubens	0.91	1.84	4.22	1.59	5.95	16.67
Psammechinus miliaris	0.83	1.51	3.43	1.69	4.84	21.51
Ciona intestinalis	1.12	0.33	3.40	0.69	4.80	26.32
Alcyonium digitatum	1.09	0.68	3.15	0.90	4.44	30.75
Pagurus spp.	1.75	1.83	2.97	1.48	4.18	34.94
Cellaria spp.	0.94	0.00	2.81	0.61	3.97	38.91
Callionymus spp.	0.51	0.61	2.01	1.17	2.84	41.75
Trisopterus spp.	0.34	0.46	1.79	0.56	2.53	44.28
Buccinum undatum	0.00	0.66	1.78	0.72	2.51	46.79
Liocarcinus spp.	0.16	0.64	1.57	0.75	2.22	49.00
Inachus spp.	0.49	0.27	1.46	1.18	2.06	51.07
Chelidonichthys cuculus	0.37	0.18	1.26	0.77	1.77	52.84
Caryophyllia smithii	0.38	0.08	1.24	0.70	1.75	54.59
Anapagurus laevis	0.14	0.44	1.21	0.66	1.70	56.29
Nemertesia spp.	0.41	0.11	1.15	1.06	1.62	57.90
Glycymeris glycymeris	0.28	0.15	1.06	0.61	1.49	59.40
Ophiothrix fragilis	0.08	0.37	1.03	0.92	1.45	60.84
Microchirus variegatus	0.16	0.26	1.02	0.73	1.44	62.28
Macropodia spp.	0.40	0.09	0.98	1.42	1.38	63.67
Alcyonidium diaphanum	0.36	0.02	0.94	0.76	1.33	65.00
Scyliorhinus canicula	0.45	0.00	0.93	0.37	1.31	66.31
Arnoglossus spp.	0.16	0.26	0.92	0.76	1.30	67.61
Ophiura spp.	0.33	0.35	0.90	1.31	1.27	68.87
Solea spp.	0.10	0.29	0.88	0.43	1.24	70.11
Hyperoplus lanceolatus	0.21	0.00	0.87	0.33	1.23	71.34
Eledone cirrhosa	0.37	0.00	0.85	0.49	1.20	72.54
Echinus esculentus	0.34	0.00	0.82	0.62	1.16	73.71
Astropecten irregularis	0.28	0.06	0.82	0.42	1.15	74.86
Limanda limanda	0.00	0.34	0.81	0.26	1.14	76.00
Hydrallmania spp.	0.11	0.33	0.79	0.91	1.11	77.11
Colus gracilis	0.26	0.01	0.76	0.75	1.07	78.18
Crangon spp.	0.01	0.28	0.67	0.72	0.95	79.12
Pomatoschistus spp.	0.19	0.08	0.62	0.63	0.87	80.00

Appendix 5.1

Species recorded during CEFAS dataset and author sampling trips

Species	CEFAS data	Author data	Species	CEFAS data	Author data
Aequipecten opercularis	Y	Y	*Lophius piscatorius*	Y	Y
Agonus cataphractus		Y	*Luidia ciliaris*		Y
Alcyonidium diaphanum		Y	*Lutraria lutraria*		Y
Alcyonium digitatum		Y	*Maja squinado*	Y	Y
Ammodytidae sp.	Y		*Marthasterias glacialis*		Y
Anseropoda placenta		Y	*Merlangius merlangus*	Y	
Aphrodita aculeate		Y	*Microchirus variegatus*	Y	Y
Arnoglossus imperialis		Y	*Microstomus kitt*	Y	Y
Arnoglossus laterna		Y	*Mullus surmuletus*	Y	
Aspitrigla cuculus	Y		*Mustelus asterias*	Y	
Astarte sulcata		Y	*Mytilus edulis*		Y
Asterias rubens		Y	*Octopus vulgaris*		Y
Astropecten irregularis		Y	*Ophiura spp.*		Y
Atelecyclus rotundatus		Y	*Ostrea edulis*	Y	Y
Blennius gattorugine	Y		*Pagurus spp.*		Y
Blennius ocellaris	Y		*Palliolum tigerinum*		Y
Botryllus schlosseri		Y	*Papillicardium papillosum*		Y
Buccinum undatum	Y	Y	*Pecten maximus*	Y	Y
Buglossidium luteum	Y	Y	*Pegusa lascaris*	Y	
Callionymus lyra	Y		*Phrynorhombus norvegicus*		Y
Callionymus spp.		Y	*Pisidia longicornis*		Y
Cancer pagarus	Y	Y	*Pleuronectes platessa*	Y	Y
Chelidonichthys cuculus		Y	*Pollachius pollachius*	Y	
Chelidonichthys lucerna	Y	Y	*Porania pulvillus*		Y
Ciona intestinalis		Y	*Porcellana platycheles*		Y
Crepidula fornicate		Y	*Psammechinus miliaris*		Y
Crossaster papposus		Y	*Raja brachyura*	Y	
Diplecogaster bimaculata	Y		*Raja clavata*	Y	Y
Dromia personata		Y	*Raja montagui*	Y	
Ebalia spp.		Y	*Raja naevus*		Y
Echinus esculentus		Y	*Raja undulata*	Y	Y
Eledone cirrhosa		Y	*Sardina pilchardus*	Y	
Ensis spp.		Y	*Scophthalmus maximus*	Y	Y
Eunicella verrucosa		Y	*Scophthalmus rhombus*	Y	Y
Gadus morhua	Y		*Scyliorhinus canicula*	Y	Y
Galathea spp.		Y	*Sepia officinalis*	Y	Y
Henricia sanguinolenta		Y	*Sepiola atlantica*		Y
Hippoglossoides platessoides		Y	*Solea solea*	Y	Y
Homarus gammarus	Y		*Spatangus purpureus*		Y
Hyperoplus lanceolatus	Y		*Syngnathus acus*		Y
Inachus spp.		Y	*Syngnathus spp.*		Y
Laevicardium crassum		Y	*Tapes rhomboides*		Y
Lepidorhombus whiffiagonis	Y	Y	*Torpedo marmorata*	Y	
Leucoraja naevus	Y		*Trigloporus lastoviza*	Y	Y
Limanda limanda	Y		*Trisopterus esmarkii*		Y
Liocarcinus spp.		Y	*Trisopterus luscus*	Y	
Lipophrys pholis		Y	*Trisopterus minutus*	Y	Y
Loligo vulgaris		Y	*Zeus faber*	Y	Y
Lophius budegassa	Y				

Appendix 5.2

Location, month, year (2012 or 2013) and duration of sampling trips. Author survey trip IDs are prefixed with ‘S’, CEFAS trip IDs are prefixed with ‘C’.

Trip ID	Latitude	Longitude	Area	Month	Year	Location	Trip duration (days)
C6	50.7000	0.4667	VIID	March	12	East Sussex, 6-12 NM	2
S10	50.8849	1.0131	VIID	March	13	East Sussex, <6 NM	1
C1	50.3500	0.9667	VIID	September	11	East Sussex, mid-Channel	7
S9	49.6316	-0.3324	VIID	September	12	Baie de Seine, >12 NM	6
C16	50.4333	-0.1667	VIID	December	12	West Sussex, mid-Channel	8
C3	50.1667	-3.8167	VIIE	February	12	Start point, <6 NM	1
C4	49.4667	-3.2333	VIIE	February	12	west of Guernsey, mid Channel	8
C7	50.3000	-4.4167	VIIE	March	12	east Falmouth Bay, <6 NM	1
C17	50.2167	-3.5167	VIIE	March	13	Start Bay, <6 NM	1
C18	50.2167	-3.5167	VIIE	March	13	Start Bay, <6 NM	1
C10	50.2500	-4.3500	VIIE	May	12	east Falmouth Bay, <6 NM	1
C19	50.2667	-4.5500	VIIE	May	13	east Falmouth Bay, <6 NM	1
C11	50.2167	-4.7500	VIIE	June	12	mid Falmouth Bay, <6 NM	1
C20	50.2167	-3.5500	VIIE	June	13	Start Bay, <6 NM	1
S1	50.1666	-4.7445	VIIE	June	12	mid Falmouth Bay, <6 NM	1
S2	49.9986	-5.0345	VIIE	June	12	west Falmouth Bay, <6 NM	1
S8	49.9915	-5.1119	VIIE	June	13	west Falmouth Bay, <6 NM	1
C23	49.9500	-5.0167	VIIE	July	13	west Falmouth Bay, <6 NM	1
C24	50.1167	-3.5500	VIIE	July	13	Start point, 6-12 NM	1
S3	50.1485	-4.7750	VIIE	July	12	mid Falmouth Bay, <6 NM	1
S4	50.6482	-2.7089	VIIE	July	12	Lyme Bay (east), <6 NM	2
S5	50.2980	-4.4791	VIIE	July	12	east Falmouth Bay, <6 NM	1
C12	50.2667	-4.3667	VIIE	August	12	east Falmouth Bay, <6 NM	3
C25	50.2500	-4.4333	VIIE	August	13	east Falmouth Bay, <6 NM	1
S6	50.2538	-4.3087	VIIE	August	12	east Falmouth Bay, <6 NM	1
C27	50.2333	-4.4000	VIIE	September	13	east Falmouth Bay, <6 NM	1
C28	50.2167	-4.6667	VIIE	September	13	mid Falmouth Bay, <6 NM	1
S7	50.5921	-3.2905	VIIE	October	12	Lyme Bay (west), <6 NM	2
C2	50.6667	0.3167	VIID	February	12	East Sussex, <6 NM	1
C5	50.8333	0.9667	VIID	February	12	East Sussex, <6 NM	1
C13	50.5500	-2.4667	VIIE	August	12	Portland, <6 NM	1
C14	50.5500	-2.4667	VIIE	October	12	Portland, <6 NM	1
C22	50.2500	-3.5500	VIIE	June	13	Start Bay, <6 NM	1
C26	50.5333	-2.4667	VIIE	August	13	Portland, <6 NM	1

Appendix 5.3

Values for length-weight parameters or mean weight of an individual, used to estimate the biomass of species retained in scallop dredge samples. Mean weight relationships calculated using the formula: weight = aL^b where L is body length. A list of literature used follows this table.

species	common name	a	b	mean weight
Aequipecten opercularis	queen scallop			0.0242
Arnoglossus imperialis	imperial scaldfish	0.007	3.0541	0.0130
Astarte sulcata		-3.3380	2.925	0.0573
Astropecten irregularis	five-armed starfish			0.0399
Atelecyclus rotundatus	circular crab			0.0126
Cancer pagurus	brown crab	0.0001	3.1170	
Echinus esculentus	common sea urchin			0.0620
Lophius piscatorius	monkfish	0.0290	2.8401	
Luidia ciliaris	seven-armed starfish			0.0773
Marthasterias glacialis	spiny starfish			0.2124
Pagurus spp.	hermit crab			0.0111
Pecten maximus	king scallop (Area VIIE)	0.0002	2.9676	
Pecten maximus	king scallop (Area VIID)	0.0004	2.7724	
Crepidula fornicata	slipper limpet			0.0010
Inachus spp.	spider crabs			0.0033
Lepidorhombus whiffiagonis	megrim	0.0077	3.0000	
Microstomus kitt	lemon sole	0.0080	3.1310	
Anseropoda placenta	goosefoot starfish			0.0065
Solea solea	dover sole	0.0097	3.0000	
Eledone cirrhosa	curled octopus	0.3360	2.281	
Buccinum undatum	common whelk	0.0001	3.0114	
Cancer pagurus	brown crab	0.0001	3.1170	
Eunicella verrucosa	pink sea fan	n/a	n/a	n/a
Galathea spp.	squat lobster			0.0006
Crossaster papposus	common sun star			0.0372
Porania pulvillus	cushion star			0.0414
Asterias rubens	common starfish			0.0343
Henricia sanguinolenta	bloody Henry starfish			0.0317
Leucoraja naevus	cuckoo ray	0.0020	3.2870	
Botryllus schlosseri	star ascidian			0.0032
Ciona intestinalis	unitary sea squirt			0.0038
Aphrodita aculeata	sea mouse			0.0082
Callionymus spp.	dragonet	0.0136	2.5930	
Eledone cirrhosa	curled octopus	0.3360	2.2810	

Maja squinado	spiny spider crab	0.0002	3.0896	
Laevicardium crassum	norway cockle			0.0500
Ophiura spp.	brittlestar			0.0019
Callionymus spp.	dragonet	0.0136	2.5930	
Ophiura spp.	brittlestar			0.1900
Aequipecten opercularis	queen scallop			0.0242
Psammechinus miliaris	green sea urchin			0.0036
Raja clavata	thornback ray	0.0020	3.2870	
Scyliorhinus canicula	small spotted catshark	0.0025	3.0958	
Pleuronectes platessa	plaice	0.0093	3.0000	
Scophthalmus rhombus	brill	0.0101	3.0900	
Alcyonium digitatum	dead man's fingers			0.0085
Syngnathus spp.	greater pipefish	0.0001	3.729	
Sepia officinalis	cuttlefish	0.0004	2.7490	
Chelidonichthys cuculus	red gurnard	0.0086	3.0373	
Lipophrys pholis	shanny	0.0093	3	
Liocarcinus spp.	dwarf swimming crab			0.0041
Tapes rhomboides	banded carpet shell			0.0236
Pisidia longicornis	pea crab			0.0008
Porcellana platycheles	broad-clawed porcelain crab	n/a	n/a	n/a
Zeus faber	John Dory	0.0399	2.7536	
Lipophrys pholis	shanny	0.0093	3.0000	
Alcyonidium diaphanum				0.0045
Hippoglossoides platessoides	long rough dab	0.0044	3.2039	
Microchirus variegatus	thick-backed sole	0.0142	2.9536	
Papillicardium papillosum		n/a	n/a	n/a
Loligo vulgaris	common Squid	0.0500	2.4181	
Phrynorhombus norvegicus	Norwegian topknot	0.0262	2.7505	
Ebalia spp.	nut crab			0.0010
Microchirus variegatus	thick-backed sole			0.0350
Octopus vulgaris	common octopus			0.1890
Palliolum tigerinum	tiger scallop			0.0030
Buglossidium luteum	solenette	0.0101	3.008	
Asterias rubens	common starfish			3.4300
Lutraria lutraria	common otter shell	n/a	n/a	n/a
Maja squinado	spiny spider crab	0.0006	2.8632	
Ostrea edulis	common flat oyster	0.1270	3.1480	
Pecten maximus	king scallop	0.0004	2.7724	
Solea solea	Dover sole	0.0095	3.0000	
Ensis spp.	razor clam	0.0000	0.03	
Mytilus edulis	blue mussel	-4.8834	2.6616	

Pleuronectes platessa	plaice	0.0106	3.0000	
Scophthalmus maximus	turbot	0.0105	3.1730	
Laevicardium crassum	Norway cockle	0.2553	3.1060	
Raja undulata	undulate ray	0.0040	3.1346	
Scophthalmus rhombus	brill	0.0101	3.09	
Buccinum undatum	common whelk			0.0657
Chelidonichthys lucerna	tub gurnard	0.0134	2.9165	
Agonus cataphractus	pogge	0.0196	2.6139	
Sepiola atlantica	little cuttlefish	2.8426	2.6035	
Spatangus purpureus	purple heart urchin	0.3981	2.8870	
Dromia personata	sponge crab	n/a	n/a	n/a
Scyliorhinus canicula	small spotted catshark	0.0043	2.9436	
Trisopterus minutus	poor cod	0.0157	2.8713	
Sepiola atlantica	little cuttlefish	2.8426	2.6035	0.1762
Trisopterus esmarkii	Norway pout	0.0066	3.0000	
Arnoglossus laterna	scaldfish	0.0139	2.7989	
Syngnathus spp.	pipefish	0.0001	3.7290	
Trigloporus lastoviza	streaked gurnard	0.0170	2.8685	

References for length/weight parameters

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Appendix 5.4

Estimated individual weight of bycatch species for which length was not recorded. Mean values taken from chapter 4 data, unless specified otherwise.

Species	mean weight per individual (g)	notes
Anseropoda placenta	0.0065	
Aequipecten opercularis	0.0242	
Buccinum undatum	0.0395	
Eledone cirrhosa	0.2114	
Ostrea edulis	0.1400	
Atelecyclus rotundatus	0.0126	
Alcyonidium diaphanum	0.0045	
Alcyonium digitatum	0.0085	
Aphrodita aculeata	0.0082	
Arnoglossus imperialis	0.0130	
Asterias rubens	0.0343	
Astropecten irregularis	0.0399	
Botryllus schlosseri	0.0032	
Ciona intestinalis	0.0038	
Crepidula fornicata	0.0220	
Crossaster papposus	0.0372	
Dromia personata		no data available
Ebalia spp.	0.0010	
Echinus esculentus	0.0620	
Galathea spp.	0.0006	
Henricia sanguinoleta	0.0317	
Inachus spp.	0.0033	
Liocarcinus spp.	0.0041	
Luidia spp.	0.0773	
Lutraria lutraria		no data available
Marthasterias glacialis	0.2124	
Octopus vulgaris	0.1890	
Ophiura spp.	0.0019	
Pagurus spp.	0.0111	
Palliolum tigerinum	0.0030	
Papillicardium papillosum		no data available
Pisidia longicornis	0.0008	
Porania pulvillus	0.0414	source: Jennings *et al.*, 2002
Porcellana platycheles		no data available
Psammechinus miliaris	0.0036	
Tapes rhomboides	0.0236	

Appendix 5.5

Summary of mean haul weights and proportions from each of the ten sampling trips. n = number of hauls sampled.

		Proportion of total catch (living biomass and substrate)						Proportion of living biomass	
		substrate (kg m^{-2})	catch (kg m-2)	% substrate	% bycatch	% *P. maximus*	n	% bycatch	% *P. maximus*
S1	**mean**	**6459.45**	**1262.93**	**77.1**	**2.5**	**20.5**	7	**11.9**	**88.1**
	min	687.79	896.65	38.8	0.5	7.0		5.1	79.3
	max	12679.11	1584.65	91.2	4.7	56.5		20.7	94.9
	S.D.	3807.46	243.57	18.0	1.7	16.9		7.4	7.4
S2	**mean**	**17518.73**	**1975.84**	**89.5**	**0.9**	**9.6**	9	**7.6**	**92.4**
	min	11977.89	1384.44	86.0	0.1	6.1		0.7	79.0
	max	24655.84	3059.20	93.8	2.4	12.3		21.0	99.3
	S.D.	4192.33	492.18	2.9	0.9	2.3		7.0	7.0
S3	**mean**	**9777.89**	**1540.26**	**85.8**	**1.4**	**12.8**	8	**9.8**	**90.2**
	min	6868.16	938.70	78.2	0.3	7.1		4.2	82.9
	max	16206.30	2107.03	92.6	2.4	19.8		17.1	95.8
	S.D.	2847.14	423.94	4.7	0.7	4.3		4.7	4.7
S4	**mean**	**20639.97**	**2895.90**	**86.4**	**5.5**	**8.1**	28	**37.3**	**62.7**
	min	8494.80	1613.04	74.2	1.1	4.2		14.8	39.4
	max	35074.69	4466.13	93.4	15.5	14.3		60.6	85.2
	S.D.	7017.92	688.18	5.7	3.6	2.7		12.1	12.1
S5	**mean**	**4232.32**	**1393.79**	**75.0**	**7.5**	**17.5**	6	**28.5**	**71.5**
	min	3184.80	861.38	69.1	2.2	14.2		13.1	52.5
	max	6376.89	1859.85	83.2	14.7	25.0		47.5	86.9
	S.D.	1127.02	388.63	5.8	4.9	3.9		14.3	14.3

S6	**mean**	**1792.14**	**480.64**	**78.0**	**4.5**	**17.4**	5	**20.5**	**79.5**
	min	1195.81	385.16	67.7	2.8	12.4		15.4	69.2
	max	2343.90	571.83	83.8	7.6	24.8		30.8	84.6
	S.D.	455.97	76.95	6.7	2.0	5.3		6.6	6.6
S7	**mean**	**26435.39**	**2055.01**	**92.4**	**1.2**	**6.4**	9	**15.7**	**84.3**
	min	13759.58	1244.91	88.1	0.5	4.5		8.6	76.0
	max	36702.21	3028.21	94.7	2.0	10.2		24.0	91.4
	S.D.	8252.34	507.15	1.9	0.5	1.7		5.1	5.1
S8	**mean**	**16995.11**	**3668.91**	**82.3**	**4.1**	**13.6**	5	**23.5**	**76.5**
	min	15143.62	2997.59	79.7	2.9	11.7		15.8	70.8
	max	18665.40	4246.93	84.3	5.2	15.8		29.2	84.2
	S.D.	1511.34	501.80	1.8	0.9	2.0		5.3	5.3
S9	**mean**	**9124.83**	**1696.99**	**83.6**	**3.5**	**12.9**	17	**21.6**	**78.4**
	min	4512.68	1193.14	70.7	2.0	7.5		10.9	67.3
	max	13570.74	2601.14	90.5	7.6	26.1		32.7	89.1
	S.D.	2492.59	459.96	5.2	1.5	4.6		6.7	6.7
S10	**mean**	**9858.63**	**1623.07**	**85.7**	**1.5**	**12.8**	5	**10.5**	**89.5**
	min	6925.84	1279.62	83.0	0.9	9.3		5.1	85.7
	max	11195.87	1917.33	89.7	2.1	16.1		14.3	94.9
	S.D.	1790.35	306.11	2.7	0.6	2.6		3.6	3.6

Appendix 5.6

Proportion of the total catch contributed by each species identified in catches from commercial scallop vessels in the English Channel. For five species, no size-weight parameters were available, therefore biomass was not calculated. Cum.=cumulative

species	common name	no. of sites present at	mean biomass (kg km^{-2})	mean % of catch weight	cum. %
Pecten maximus	king scallop	10	1476.33	81.0	81.0
Aequipecten opercularis	queen scallop	8	130.20	6.1	87.1
Marthasterias glacialis	spiny starfish	7	82.96	3.5	90.6
Maja squinado	spiny spider crab	8	26.96	1.4	92.0
Sepia officinalis	cuttlefish	5	26.29	1.3	93.3
Cancer pagurus	brown crab	10	15.99	1.1	94.4
Lophius piscatorius	monkfish	7	15.77	1.0	95.4
Asterias rubens	common starfish	6	20.65	1.0	96.4
Luidia ciliaris	seven-armed starfish	7	13.74	0.8	97.3
Buccinum undatum	common whelk	6	6.70	0.3	97.6
Ostrea edulis	common flat oyster	1	5.41	0.3	97.9
Raja clavata	thornback ray	4	4.96	0.2	98.1
Solea solea	Dover sole	8	3.20	0.2	98.3
Scyliorhinus canicula	small spotted catshark	7	3.51	0.2	98.5
Scophthalmus maximus	turbot	2	2.69	0.2	98.7
Pleuronectes platessa	plaice	6	2.41	0.2	98.8
Echinus esculentus	common sea urchin	6	1.78	0.1	99.0
Pagurus spp.	hermit crab	9	2.44	0.1	99.1
Tapes rhomboides	banded carpet shell	4	2.15	0.1	99.2
Microstomus kitt	lemon sole	8	1.68	0.1	99.3
Inachus spp.	spider crabs	6	0.85	0.1	99.4
Ophiura spp.	brittlestar	4	1.67	0.1	99.4
Laevicardium crassum	Norway cockle	2	0.88	0.1	99.5
Crepidula fornicata	slipper limpet	8	1.11	0.1	99.5
Astarte sulcata	bivalve	5	1.15	0.1	99.6
Henricia sanguinolenta	Bloody Henry starfish	4	1.31	0.0	99.6
Atelecyclus rotundatus	circular crab	4	0.54	0.0	99.7
Porania pulvillus	cushion star	1	0.60	0.0	99.7
Liocarcinus spp.	dwarf swimming crab	5	0.53	0.0	99.7
Lepidorhombus whiffiagonis	megrim	2	0.52	0.0	99.8
Sepiola atlantica	little cuttlefish	2	0.57	0.0	99.8
Crossaster papposus	common sun star	2	0.49	0.0	99.8
Psammechinus miliaris	green sea urchin	4	0.56	0.0	99.8
Mytilus edulis	blue mussel	2	0.33	0.0	99.9
Callionymus spp.	dragonet	5	0.34	0.0	99.9
Spatangus purpureus	purple heart urchin	1	0.33	0.0	99.9
Scophthalmus rhombus	brill	2	0.33	0.0	99.9

Astropecten irregularis	five-armed starfish	1	0.19	0.0	99.9
Alcyonium digitatum	dead man's fingers	5	0.14	0.0	99.9
Anseropoda placenta	goosefoot starfish	6	0.13	0.0	99.9
Aphrodita aculeata	sea mouse	4	0.07	0.0	99.9
Hippoglossoides platessoides	long rough dab	3	0.12	0.0	100.0
Botryllus schlosseri	star ascidian	1	0.12	0.0	100.0
Eledone cirrhosa	curled octopus	4	0.05	0.0	100.0
Buglossidium luteum	solenette	1	0.02	0.0	100.0
Microchirus variegatus	thick-backed sole	3	0.04	0.0	100.0
Leucoraja naevus	cuckoo ray	1	0.06	0.0	100.0
Octopus vulgaris	common octopus	1	0.04	0.0	100.0
Ensis spp.	razor clam	2	0.07	0.0	100.0
Zeus faber	John Dory	2	0.06	0.0	100.0
Arnoglossus imperialis	imperial scaldfish	1	0.02	0.0	100.0
Ciona intestinalis	unitary sea squirt	3	0.03	0.0	100.0
Lipophrys pholis	shanny	1	0.05	0.0	100.0
Chelidonichthys cuculus	red gurnard	2	0.05	0.0	100.0
Palliolum tigerinum	tiger scallop	1	0.01	0.0	100.0
Galathea spp.	squat lobster	5	0.01	0.0	100.0
Trigloporus lastoviza	streaked gurnard	1	0.04	0.0	100.0
Phrynorhombus norvegicus	Norwegian topknot	1	0.01	0.0	100.0
Alcyonidium diaphanum	sea chervil	1	0.02	0.0	100.0
Agonus cataphractus	pogge	1	0.01	0.0	100.0
Loligo vulgaris	common Squid	1	0.01	0.0	100.0
Chelidonichthys lucerna	tub gurnard	1	0.01	0.0	100.0
Raja undulata	undulate ray	1	0.01	0.0	100.0
Trisopterus minutus	poor cod	1	0.01	0.0	100.0
Arnoglossus laterna	scaldfish	1	0.01	0.0	100.0
Trisopterus esmarkii	Norway pout	1	0.01	0.0	100.0
Ebalia spp.	nut crab	1	0.00	0.0	100.0
Syngnathus spp.	pipefish	2	0.01	0.0	100.0
Pisidia longicornis	pea crab	1	0.00	0.0	100.0
Dromia personata	sponge crab	1	n/a	0.0	100.0
Eunicella verrucosa	pink sea fan	4	n/a	0.0	100.0
Lutraria lutraria	common otter shell	1	n/a	0.0	100.0
Papillicardium papillosum	bivalve	1	n/a	0.0	100.0
Porcellana platycheles	broad-clawed porcelain crab	1	n/a	0.0	100.0

Appendix 5.7

Results from a SIMPER analysis using square-root transformed biomass data giving the average similarity in species composition for stations grouped by environmental characteristics identified by a PCA. Av.Abund = Average biomass (kg km^{-2}). Taxa which contributed to 80 % (cumulative) of the similarity within groups are shown. Sim/SD is the similarity/standard deviation ratio. Sim/SD values of <1.3 indicate greater variation within stations than between and therefore those taxa are not considered to be a reliable representative of the similarity within the group.

Group A					
Average similarity: 66.58 %					
Species	**Av.Abund**	**Av.Sim**	**Sim/SD**	**Contrib%**	**Cum.%**
Pecten maximus	1585.63	44.23	**5.55**	66.43	66.43
Aequipecten opercularis	224.70	7.19	0.82	10.80	77.23
Asterias rubens	54.02	5.64	**1.46**	8.47	85.70
Maja squinado	8.82	2.69	**1.78**	4.04	89.73
Sepia officinalis	5.29	1.42	0.72	2.14	91.87
Crepidula fornicata	2.59	1.36	**1.97**	2.04	93.91
Buccinum undatum	5.24	1.16	0.64	1.74	95.65
Group B					
Average similarity: 63.64 %					
Pecten maximus	2122.44	55.69	**4.59**	87.51	87.51
Marthasterias glacialis	104.24	3.28	0.50	5.15	92.66
Lophius sp.	24.30	1.90	0.46	2.99	95.65
Group C					
Average similarity: 63.94 %					
Pecten maximus	973.44	45.58	**4.48**	71.29	71.29
Aequipecten opercularis	45.16	5.13	1.18	8.03	79.32
Marthasterias glacialis	22.47	3.63	0.71	5.68	85.00
Luidia ciliaris	12.32	3.48	1.11	5.44	90.44
Lophius sp.	5.81	1.78	0.62	2.79	93.23
Cancer pagurus	2.76	1.08	0.72	1.69	94.92
Maja squinado	1.21	0.61	0.44	0.96	95.88

Results from a SIMPER analysis using square-root transformed biomass data for stations grouped by environmental characteristics identified by a PCA. Taxa which contributed to 80 % (cumulative) of the dissimilarity between groups are shown. Av.Abund= Average biomass (kg km^{-2}). Diss/SD is the dissimilarity/standard deviation ratio. Diss/SD values of <1.3 indicate greater variation between stations than between groups and therefore those taxa are not considered to be a reliable representative of the dissimilarity between groups.

Groups A & B Average dissimilarity: 47.14 % **Species**	**Group A** **Av.Abund**	**Group B** **Av.Abund**	**Av.Diss**	**Diss/SD**	**Contrib %**	**Cum. %**
Aequipecten opercularis	224.70	0.20	8.43	1.10	17.89	17.89
Marthasterias glacialis	0.00	10.21	5.38	0.89	11.42	29.31
Pecten maximus	1585.63	46.07	4.84	**1.36**	10.27	39.58
Asterias rubens	54.02	0.48	4.23	**1.69**	8.97	48.55
Lophius sp.	0.00	4.93	2.98	0.79	6.32	54.87
Maja squinado	8.82	1.43	1.60	**1.49**	3.40	58.27
Luidia ciliaris	0.09	2.75	1.59	0.74	3.38	61.65
Sepia officinalis	5.29	0.84	1.49	1.06	3.16	64.80
Cancer pagurus	1.23	1.99	1.42	0.66	3.02	67.82
Buccinum undatum	5.24	0.00	1.38	1.00	2.92	70.74
Crepidula fornicata	2.59	0.11	0.96	**1.82**	2.04	72.78
Ophiura spp.	2.76	0.15	0.94	0.79	1.99	74.77
Raja sp.	0.50	0.89	0.78	0.61	1.66	76.43
Ostrea edulis	1.54	0.00	0.78	0.39	1.66	78.09
Pagurus sp.	0.62	1.01	0.76	0.97	1.62	79.71
Tapes rhomboides	0.67	0.65	0.73	0.64	1.56	81.26
Groups A & C Average dissimilarity: 47.34 %	**Group A**	**Group C**				
Aequipecten opercularis	224.70	6.72	8.50	1.22	17.96	17.96
Pecten maximus	1585.63	31.20	7.27	1.26	15.36	33.32
Asterias rubens	54.02	0.06	4.98	**1.77**	10.53	43.84
Marthasterias glacialis	0.00	4.74	3.24	1.10	6.84	50.69
Luidia ciliaris	0.09	3.51	2.40	**1.44**	5.08	55.77
Lophius sp.	0.00	2.41	1.73	0.94	3.65	59.42
Sepia officinalis	5.29	0.09	1.67	1.05	3.52	62.94
Maja squinado	8.82	1.10	1.58	**1.50**	3.34	66.28
Buccinum undatum	5.24	0.51	1.53	1.08	3.24	69.51
Cancer pagurus	1.23	1.66	1.28	0.95	2.71	72.22
Ophiura spp.	2.76	0.08	1.03	0.79	2.17	74.39
Crepidula fornicata	2.59	0.15	1.02	**1.90**	2.15	76.54
Solea solea	1.04	0.88	1.00	0.86	2.10	78.65
Ostrea edulis	1.54	0.00	0.87	0.39	1.85	80.49
Groups B & C Average dissimilarity: 45.12 %	**Group B**	**Group C**				
Pecten maximus	2122.44	31.20	10.99	**1.61**	24.36	24.36

Marthasterias glacialis	104.24	4.74	6.80	1.27	15.06	39.42
Aequipecten opercularis	0.04	6.72	4.55	1.14	10.08	49.50
Lophius sp.	24.30	2.41	3.71	1.02	8.22	57.72
Luidia ciliaris	7.56	3.51	2.79	**1.43**	6.17	63.90
Cancer pagurus	3.96	1.66	1.84	0.71	4.07	67.97
Maja squinado	2.04	1.10	1.20	1.05	2.66	70.64
Pagurus sp.	1.02	0.96	0.96	1.03	2.12	72.76
Echinus esculentus	0.90	0.61	0.95	0.71	2.11	74.87
Microstomus sp.	0.58	0.61	0.77	0.81	1.70	76.57
Porania pulvillus	0.79	0.00	0.71	0.51	1.57	78.14
Scyliorhinus canicula	0.62	0.52	0.69	0.71	1.54	79.68
Solea solea	0.00	0.88	0.64	0.60	1.42	81.09

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