Spoilt for choice?

Linking individual fishing behaviour with fleet dynamics

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Declaration of Originality

This thesis is the result of my own work. The work of all others is appropriately acknowledged and referenced in the text.

Alex Tidd, 2013

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Abstract

Much progress has been made in developing a precautionary approach to fisheries management, however in most cases, this has been largely confined to biological elements and a more balanced application needs to address social and economic risks as well. A current challenge for global fisheries governing bodies is to manage fishing capacity so that it is commensurate with the availability of the resource. Fisheries science is by its nature an interdisciplinary field, and combining information has proven to be increasingly important in achieving sustainable fisheries management. One factor of increasing importance is the ability to anticipate fisher behaviour in response to management regulation, in order to reduce the unanticipated side effects of management actions aimed both at the fishery sector and at other sectors. The primary aim of this work is to improve understanding of fisher behaviour to support fisheries management.

Statistical modelling tools were applied to determine the relative importance of, and improve understanding of, selected drivers for both short term and long term behavioural responses to fishery management measures, to quantify the relationships between capacity, effort and fishing mortality and to investigate spatial competition with other marine sectors. The results demonstrate that expected revenues from target species, experience or habit, management measures, fuel prices, aggregate activity and maritime traffic are significant factors in determining fisher decision-making on when and where to fish. Some of the unobserved random components of the model causing heterogeneity in the selection of fishing grounds by fishers could be attributable to individual variations in decision-making, along with other factors such as skipper skill, age, nationality and vessel attributes.

Detailed individual-level vessel data that take into account the heterogeneity and dynamics of a beam-trawl fishing fleet were analysed to draw linkages between capacity, effort and fishing mortality. These relationships could be developed for use as indicators for spatial and temporal management. A key finding from this study was the detection of a switch in species targeting and fishing efficiency over time, with an estimated 6.2% annual decrease in plaice (*Pleuronectes platessa*) and an estimated 0.6% increase in sole (*Solea solea*) over the 11-year study period.

The research demonstrated how knowledge of drivers of fisher behaviour can lead to better understanding of responses of fishing fleets to management and how more detailed information on fleet structure and dynamics (including effort and capacity) improves knowledge of the relative contributions of different components of a fleet to fishing mortality.

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Chapter 1. Introduction

Scallop dredger in Holyhead Harbour (Source: Jim Ellis)

1.1 Problem statement and rationale

1.1.1 The state of the world's fish stocks

According to the Food and Agriculture Organisation (FAO, 2006), since the 1950s, there has been a consistent downward trend in the proportion of marine stocks with potential for expanded production and a concurrent increase in the proportion of stocks classified as depleted or overexploited. The State of World Fisheries and Aquaculture (SOFIA) (FAO, 2010; Figure 1.1) recently estimated that 20% of the world's marine resources were under- or moderately and sustainably exploited, 52% fully exploited (i.e. harvested at their maximum biological productivity) and 27% overexploited and under strict management plans, depleted or recovering from depletion (FAO, 2010). Fish and shellfish are the primary source of protein for some 950 million people worldwide and represent an important part of the diet of many more (UNEP, 2001). Given the nutritional importance of fish for so many people, large-scale collapse of fisheries or a significant increase in the price of fish products (the likely result of smaller catches and increased demand) could seriously influence the nutritional status and food security of many populations (World Bank, 2005), especially in developing countries.

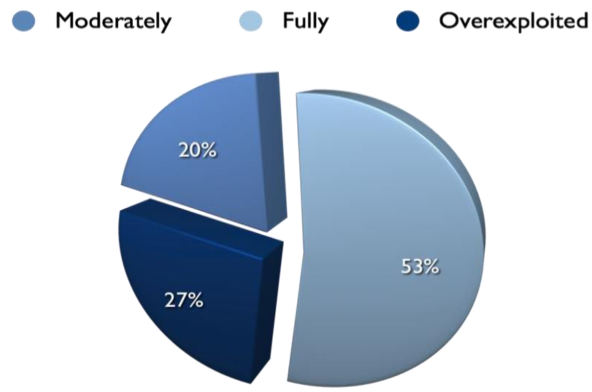

Figure 1.1 The exploited levels of the world's fish stocks (SOFIA) (Source: FAO, 2010).

1.1.2 Technological change and the demand for fish

The past two centuries have seen dramatic changes taking place, affecting commercial fisheries and fish stocks. The Industrial Revolution of the 18th and 19th centuries induced changes that spread throughout Europe, North America and eventually the world. Technological, scientific and medical innovations have resulted in a human population increase and in many cases the availability of more disposable income. Technological advancement has also allowed the development of larger, more-efficient fishing vessels (Figure 1.2), better storage and preservation of commodities (including fishery products), and better transport networks for their distribution, which together facilitated the greater demand for fish (Caddy and Cochrane, 2001). With the ever-increasing demand for fish and more efficient ways to find, catch and process it, along with the expanding number of industrial-scale fishing boats being built, there has been a build-up of excess fishing capacity, with often too many vessels or excessive harvesting power in a number of fisheries harvesting depleted stocks (FAO, 2003a).

1.1.3 Too many boats catching too few fish

When fisheries are poorly regulated, there is a race to fish because of competition for a shared resource. Such a situation, often referred to as the Tragedy of the Commons, is a situation not only relevant to fisheries but to other natural resources (Hardin, 1968). For fisheries to remain sustainable and profitable, the fishing effort applied must be in proportion with the fishing opportunities, i.e. excess fishing capacity needs to be reduced to an optimum level for those fishing opportunities. Excess capacity can be expressed as a short-term occurrence whereby fishers produce less than under normal operating conditions because of changes in market conditions and stock abundance. Conversely, overcapacity is explicitly defined as a long-term phenomenon whereby fishers continually operate under normal conditions but their production falls short of the target optimal yield, i.e. Maximum Sustainable Yield (MSY) or Maximum Economic Yield (MEY) (FAO, 2003a).

Figure 1.2 Modern powerful fishing vessels (left) in contrast to less powerful inshore vessels (right) (Source: Jim Ellis).

A current challenge for global fisheries governing bodies is to manage fishing capacity so that it is commensurate with the availability of the resource. Excess capacity and overcapacity affects many fisheries throughout the world, and fishery managers have attempted to limit capacity by introducing a form of regulated access to the finite and sometimes diminishing resources (Beddington *et al.,* 2007). Although fish stocks are renewable, they are not infinite and their exploitation needs to be managed if they are to be sustained. Capacity has not declined to the same extent as stocks (Cunningham and Gréboval, 2001), and as resources have become depleted, many fishers have redistributed their fishing effort across other fisheries, implemented new technology such as advanced fish-finding devices (Thurstan *et al.,* 2010), or participated in Illegal, Unreported and Unregulated (IUU) fishing activities as a response to regulation (Agnew *et al.,* 2009). Often, vessels and/or gears are modified to circumvent regulations and/or increase effective fishing power (Cunningham *et al.,* 1985; OECD, 1997), in an attempt to continue harvesting depleted resources at profitable levels (Gréboval, 1988). Such modifications effectively increase the efficiency of fishing vessels. Measures are needed to guard against excessive exploitation, and tough regulations are now common in most fisheries throughout the developed nations of the world.

1.1.4 Managing complex untamed systems

The major problems facing the world's fisheries are well-documented in the literature (e.g. Pauly *et al.,* 2002; Peterman, 2004; Beddington *et al.,* 2007; Costello *et al.,* 2008; Mora *et al.,* 2009; Worm *et al.,* 2009). In addition to those relating to capacity management, other problems of fisheries management relate, inter alia, to there being important sources of uncertainty in stock assessments and fishers' responses to imposed regulations designed to manage fishing effort. Major sources of uncertainty include different sources of error in stock assessment models, biased input data, and implementation error (i.e. where the outcomes of practical implementation of management measures differ from those intended; Peterman, 2004). Of course, many processes are complex, interrelated and subject to natural variation (Figure 1.3), and is difficult to account for all uncertainties, but each process needs to be understood better (e.g. how stocks and ecosystems fluctuate in both the short and long term in response to management and natural variation), as do the sources of uncertainty.

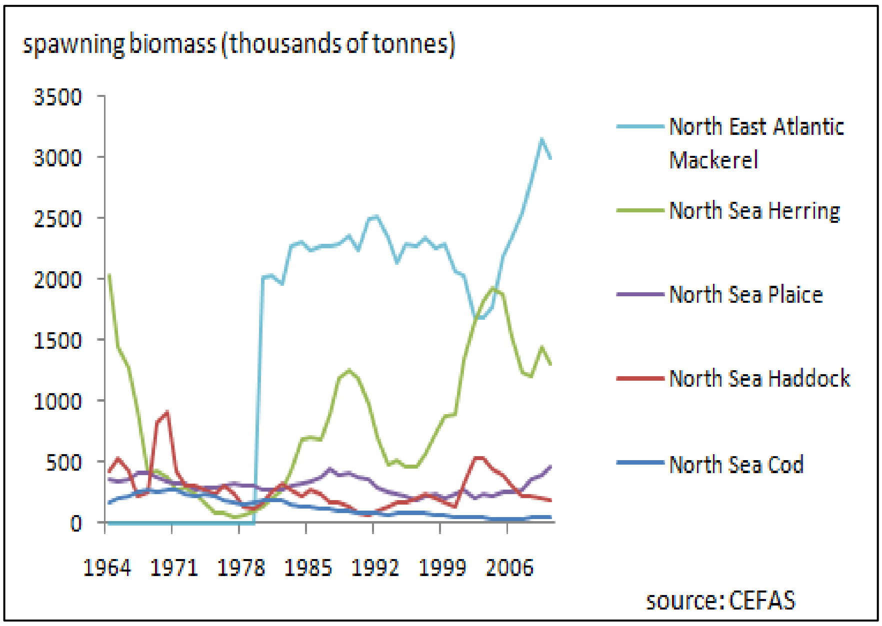

Figure 1.3 The variability in spawning stock biomass (SSB) of five North Atlantic fish stocks (Source: CEFAS).

Whilst natural variability is an inherent feature of fisheries that cannot be removed by management, minimising the effects of such variability should be possible through adaptable management strategies that deal with the inherent risks and uncertainty (Cunningham and Maguire, 2002). Further, one of the greatest challenges to fishery science and management currently is to identify causes of changes in abundance and to differentiate between those due to fishing and those due to natural factors (Cunningham and Maguire, 2002).

1.1.5 Anticipating fisher behaviour

Fisheries science is by its nature an interdisciplinary field, and combining information has proven to be increasingly important in achieving sustainable fisheries management (Mumford *et al.,* 2009). One factor of increasing importance is the ability to anticipate fisher behaviour in response to management regulation, in order to reduce implementation error (Dugan and Davis, 1993; Allison *et al.,* 1998; Fulton *et al.,* 2011). Understanding and influencing fisher behaviour in a management context is complex and involves many processes and interactions that may involve manifold actors and actions within a system, and not fishers alone. Different fishers do not act or behave identically, but management of fisheries in the past has treated them as fixed components with no consideration of individual behaviour concerning their fishing operations and individual aspirations (i.e. spatial, temporal, social, ecological and economic heterogeneities; McKelvey, 1983; Smith and McKelvey, 1986; Salas and Gaertner, 2004). Several authors have stressed the importance of integrating fisher behaviour into management and stock assessments, but progress has been slow (Wilen, 1979; Hilborn and Walters, 1992; Charles, 1993; Fulton *et al.,* 2011) and despite research in this area burgeoning (e.g. Hutton *et al.,* 2004; Poos and Rijnsdorp, 2007; Vermand *et al.,* 2008; Ran *et al.,* 2011), few have been able to quantify the uncertainty associated with a fleet's response to management decisions (Little *et al.,* 2004; Grafton *et al.,* 2006).

A fleet's response cannot be predicted with absolute certainty because the drivers that influence strategic and tactical behaviour change over time and are not necessarily predictable. Tactics can be described as short-term decisions, such as where and when to go fishing, what gear(s) to deploy, and where to land the fish (all of which can be affected by fuel costs, weather, crew availability and market price), but strategies associated with long-term decision-making include factors such as fuel price rises, costs for replacing gears, modifications to vessel (including general refurbishment as well as changes to allow deployment of other gears), stock status, catch prices (Figure 1.4) and incentives such as decommissioning schemes, investment or disinvestments for modernisation. The omission of such information in management systems, which rely solely on biological assessments, can lead to overconfidence in the likely effectiveness of proposed management actions.

Figure 1.4 A selection of catch at a Spanish fish market (Source: Jim Ellis).

1.1.6 Ecosystem-based fisheries management

Over the past two decades, fishery managers have attempted to address past failures of fishery management by acknowledging that sustainability and conservation need to be focused on through implementing an ecosystem approach to fisheries management (EAFM), also referred to as ecosystem-based fisheries management (EBFM), a term formally accepted at the Earth Summit in Rio de Janeiro in 1992. The FAO state that "*An ecosystem approach to fisheries strives to balance diverse societal objectives, by taking into account the knowledge and uncertainties about biotic, abiotic and human components of ecosystems and their interactions and applying an integrated approach to fisheries within ecologically meaningful boundaries*" (FAO, 2003b). An EAFM therefore implies that consideration of conflicting objectives in the fisheries management decision-making process needs to be taken into account. Some of the more high-profile issues that are incorporated in the ecosystem approach include increased protection for threatened species and habitats, and to address this there have been increased studies on the by-catch of threatened species and their survival, gear modifications to minimise by-catch and/or maximise discard survival and spatial management (i.e. manage anthropogenic activities in space and time, precluding or minimising conflicts between competing sectors without negatively impacting the ecosystem).

1.2 Project summary and aims

A primary aim of worldwide capacity management is to regulate fleet capacity in line with fishing opportunities. Capacity reduction and effort limitation tools are major instruments used in managing the EU's fisheries. However, the relationships between capacity, effort and fishing mortality are not well understood, so it is not straightforward to predict the likely effect of changes in capacity and effort on fishing mortality. Fisher behaviour and their reactions to management measures can result in implementation error, rendering management measures less effective.

The primary aim of this thesis is to improve understanding of fisher behaviour to support fisheries management. Using the fleet dynamics of the English beam trawl and scallop fishing fleet as case studies, I develop and apply statistical modelling tools i) to determine the relative importance of and improve understanding of selected drivers on both short term and long term behavioural responses to fishery management measures, ii) to quantify the relationships between capacity, effort and fishing mortality and iii) to investigate spatial competition with other marine sectors.

1.3 Case Studies

The first case study focuses on the North Sea and the behaviour of the English and Welsh beam trawl fleet (Figure 1.5) during a period (1997−2007) when that fleet underwent notable change with its capacity changing and its ownership transferred to operators in the Netherlands, but continuing to operate under the UK flag and utilising UK quota allocations. Initially, some vessels were leased in 2001, but ownership was formally transferred from 2002 to 2005. A key component of the case study is analysing the factors and processes that affected fleet capacity, fishing mortality and effort, and examining specific drivers that affected choices by fishers in terms of their response to capacity management and effort controls.

Figure 1.5 A beam trawl (Source: Jim Ellis).

The second case study focuses on the English and Welsh scallop dredging fleet (Figure 1.6) operating in the eastern English Channel and looks at how capacity was affected by competition between maritime sectors between 2005 and 2010. The study area contains one of the busiest shipping lanes in the world between the Atlantic Ocean and the North Sea, there is a Traffic Separation Scheme (TSS) in operation, and there are also several active aggregate-extraction sites. Since 2008, the European Union has set objectives for its member states to achieve common principles, the so-called “Roadmap for spatial planning” (EC, 2008a) under the Integrated Maritime Policy (IMP; EC, 2007). A common framework known as Maritime Spatial Planning (MSP) has been developed. The objectives of MSP are to manage anthropogenic activities in space and time, avoiding or minimising conflicts between competing sectors without negatively impacting the ecosystem within the Marine Strategy Framework Directive (MFSD; EC, 2008b) and human activities.

Figure 1.6 A scallop dredge (Source: Jim Ellis).

1.4 Structure of the thesis

The thesis consists of seven chapters:

Chapter 1 presents the problem statement and rationale (this chapter).

Chapter 2 provides an overview of the topics central to the research, (i) fisher strategic behaviour, (ii) fisher tactical behaviour, and (iii) the different factors that influence changes in fishing capacity. This chapter provides an insight into the academic literature in the research area.

Chapter 3 explores the factors that cause changes in capacity (strategy), focusing on fisher incentives for altering fishing capacity. It considers the different means of managing a fishery, including decommissioning schemes, an instrument used by the European Commission to achieve the goals of the CFP. This chapter formed the basis of the following paper: Tidd, A.N., Hutton, T., Kell, L.T., Padda, G. (2011). Exit and entry of fishing vessels: an evaluation of factors affecting investment decisions in the North Sea English beam trawl fleet. ICES Journal of Marine Science, 68: 961–971.[1]

Chapter 4 determines how capacity and effort are utilised (tactics), considering bio-economic drivers to determine where and how fishing effort is applied. The spatial allocation of effort links to Chapter 5, which explores the link between effort and fishing mortality. This chapter formed the basis of the following paper: Tidd, A. N., Hutton, T., Kell, L. T., and Blanchard, J. L, (2012). Dynamic prediction of effort re-allocation in mixed fisheries. Fisheries Research, 125–126: 243–253.[1]

Chapter 5 investigates parameters that provide the best links between capacity, effort and fishing mortality. The second objective is to identify subfleets within the demersal beam trawl fleet mixed fishery, in order to apply the spatial and temporal

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[1] The ideas, development and writing up of this paper in the thesis were conducted by Alex Tidd. The inclusion of co-authors reflects the fact that the work came from active collaboration between researchers and acknowledges input into team-based research.

management measures required when controlling capacity, effort and fishing mortality. This chapter formed the basis of the following paper: Tidd, A.N., Effective fishing effort trip indicators and their use for efficient spatial management in mixed demersal fisheries. Fisheries Management and Ecology, doi: 10.1111/fme.12021.

Chapter 6 analyses the key determinants of fisher behaviour of the ≥ 15 m scallop dredge fleet, including fisher response to the burgeoning competing sectors within their fishing grounds (aggregates, maritime traffic and inshore limits, as a proxy for the <10m fishing fleet sector) and how capacity is affected. Parameter estimation is at a fine spatial resolution using estimated recorded VMS positions.

Chapter 7 provides an overview of the work developed in the thesis, highlights the importance and potential limitations of each study, and suggests future work to take the ideas forward.

Chapter 2. Background and literature review

Fishing vessels in Cork harbour (Source: Jim Ellis)

This section gives an overview of the literature on fisher behaviour and fleet dynamics and why better understanding of fisher responses to regulatory measures is needed for effective fisheries management. To place the literature on fleet behaviour in context of the case study areas, I will provide background on fisheries management in European waters and its associated problems. I will also introduce and define some of the technical terms used throughout this thesis.

2.1 Why is understanding fisher behaviour and fleet dynamics important?

Since the latter part of the 20th century, behavioural ecologists have been observing the responses by humans to changing environments (see Smith *et al.,* 1992; Cronk *et al.,* 2000; Smith, 2000). Their goal was to determine how social and ecological factors influence human responses within and across groups, and to predict patterns of behaviour, using mathematical models to quantify the relationships and processes observed. Although biological advice is a key element in the successful management of fisheries, it is increasingly evident that fisheries management is not solely a biological or ecological issue. The aim of fisheries management is to balance the needs of fishers while maintaining and sustaining healthy stocks while also reducing impacts to the ecosystem. For fisheries management to be successful several authors have stated that understanding fisheries dynamics and the drivers that influence the behaviour of fishers is necessary (Wilen, 1979; Hilborn and Walters, 1992; Charles, 1993; Fulton *et al.,* 2011).

Many factors influence a fisher's decision where and when to fish, including fish distribution, fuel price, regulations, the weather, their habits and experience, previous catch rates, market prices, and the proximity to landing ports. These factors can lead to differences in observed individual fisher behaviour and the way a group of fishers (a fleet) allocate their effort in time and space (See Table 2.1 for a break down of studies that include factors that influence fisher behaviour).

Management of the resource can only be achieved by managing the resource user (fisher or fishers) and hence the fishing mortality of the resource. The numbers of vessels, size of vessels or engine power are capacity measures and can be linked to fishing effort; as effort is the amount of time fishing capacity is deployed in the fishery. Fisheries management often assumes that a reduction in capacity or effort is assumed to reduce fishing mortality but that may not necessarily always be the case and both capacity and effort can be challenging to quantify and collect accurate data on. Also, the data acquired can be heavily influenced by fisher behaviour.

Overcapacity means that there are too many boats chasing too few fish, which means that as stocks decline it is difficult to implement recovery and long-term management plans. For example, while EU fish stocks have declined, fishing capacity has not (http://www.ieep.eu/assets/437/overcapacity.pdf). Instead the fishers are motivated to redistribute their fishing effort across fisheries, implement new technologies such as advanced fish-finding devices (Branch *et al.,* 2006), or participate in illegal, unreported and unregulated (IUU) fishing (Agnew *et al.*, 2009) or increase power to try and better harvest a dwindling resource (Khalilian *et al.,* 2010). In a number of fisheries, fleet capacity is said to be 2–3 times the sustainable level (http://ec.europa.eu/fisheries/documentation/publications/cfp_brochure/fleet_en.pdf). However with excessive levels of capacity economic problems arise, with knock-on effects to the conservation of other stocks.

The definition of fleet dynamics is widely understood as changes in fishing capacity, the FAO (2003) define 'capacity' of a vessel, or a fleet as its ability, or power to generate fishing effort per period of time. In fisheries science, fishing effort (E) is an essential parameter in the assessment of fish stocks and their effective management. It is linked to fishing mortality (F) via the catchability (q) at age of a stock, a term that generally means the extent to which the stock is susceptible to fishing and that would be captured by one unit of effort. Catchability is therefore equally important to managers as effort in assessing fish stocks and ultimately in supporting effective management. The relationship is assumed to be linear and takes the form $F=qE$. Fishing effort, however, is difficult to quantify because the sizes and types of vessels and gears differ. It is usually approximated by a metric of capacity, such as gross tonnage or engine power, with a measure of activity (e.g. days at sea or hours fished), and is therefore an aggregated measure of fisher behaviour in locating the greatest densities of marketable fish (Rijnsdorp *et al.*, 2006).

The efficiency of fishing vessels and hence catchability tends to increase over time because of factors such as fishing technology improvements. Improvements in vessel design, marine electronics, and fish finding equipment and innovative developments in fishing gear potentially make catching fish more efficient, i.e. an increase in technical efficiency, which is commonly referred to as technical creep (Gulland, 1956). Technical creep can be quantified in relation to fishing mortality with constant nominal effort (E_n) and intensified effective effort (E_e). These relationships are important to fishery managers because they are crucial in reducing fishing mortality through effort control, and ignoring them could prove meaningless in limiting fishing mortality (Pauly *et al.,* 2002). Shepherd (2003) suggested that for a given amount of effort exerted, and because of variations in vessels and their activity, can induce different effects on stocks in terms of fishing mortality that can be generated.

2.1.2 The Common Fisheries Policy

The European Community has one of the world's largest fishing fleets, and is the world’s third largest catching power. It is also the largest importer of fishery products in the world, and in 2007 imported nearly 15 billion euros worth (ACP, 2009). The Common Fisheries Policy (CFP), provides the framework for fisheries management within the EU for its Member States, and is currently under reform (EC, 2009a). The regulatory regime governing the community waters is complex and vast compared to the size of the fishing industry and yet 88% of these stocks are over-fished (House of Lords Paper, 2008; EC, 2009a), owing to a combination of poor compliance, ineffective enforcement in many regions, overcapacity in fishing fleets due in part to subsidies (Lindebo, 2005).

The management of European mixed fisheries is primarily based on total allowable catches (TACs) along with effort restrictions (days at sea), technical measures (gear and/or mesh size regulations, size restrictions) and seasonal closures. Effort management differs from TACs in that controls on effort manage the input rather than the outputs specified by a TAC, although they both aim to limit fishing mortality. The difficulties in managing fish stocks through TACs are widely recognised (Shepherd, 2003; Beddington *et al.,* 2007). The main issue is that a TAC set to protect one species within a mixed fishery can have an undesired effect on another through increased discarding, or indirectly through foodweb interactions. Hence, a conservation policy cannot achieve its goal through this single management action alone. North Sea cod (*Gadus morhua*), for example, and many other commercially exploited species, have dramatically declined due to high fishing mortality in mixed fisheries (Hentrich and Salomon, 2006).

Some fisheries/fishing gears are highly selective, but some of the main gears used by EC fleets (e.g. trawls) are not particularly species-specific, often leading to unintended catch and discarding of undesired species (Horwood *et al.,* 2006). This can result from management. For example an inappropriate quota coupled with catch composition rules and minimum-landing size gave rise to an unavoidably high level of discarding e.g. North Sea beam trawl fishery for sole where high discarding of plaice (*Pleuronectes platessa*) occurs (Feekings *et al.,* 2012). Species that are discarded can survive but a high proportion die on return to the sea or the illegal retention of the catch leads to socially undesirable results, such as trading illegally on the black market (Copes 1986).

Another serious issue in fisheries management is when scientific advice is systematically ignored by management decision makers, or not acted upon with enough urgency usually for political expediency (Daw and Gray, 2005). The resulting policy from the governing bodies rarely reflects the scientific advice, as is can be 'bargained' through a political process (Ritchie and Zito, 1998; Payne, 2000; Lesquesne, 2001). The political process often involves member states looking after their (short-term) national interests, (i.e. the fishing fleet) rather than conservation of the resource. The result is that quotas are often set at a level higher than the predicted landings corresponding to scientific advice on what should be appropriate exploitation rates. Fishers argue about the scientific advice, saying that the stock assessments are out of date and too reliant on modelling approaches, that scientific trawl surveys are not representative of the stocks, and that commercial fisher data (e.g. logbook information) is not often used in the stock assessments and provision of advice (Richie, 2003). Fishers and various stakeholders within member states involved in policy implementation also argue about transparency and equitability of the process. The fishing industry often considers that it's too centralised, a top down regulatory measure dictated by Brussels with a lack of representation of national/regional interests. In turn these decisions result in an imbalance between fishing capacity and available resource and as such can have a knock on effect in terms of strategic behaviour of fishers.

2.1.3 Compliance and enforcement

The past has shown that when fisheries are poorly regulated, there may be a race to exploit them, a situation commonly known as the "Tragedy of the Commons" (Hardin, 1968). In an environment where change takes place, fishers develop tactics and strategies to survive when faced with rising fuel costs, fluctuating stock levels, regulations, and market conditions. In a management context it is important to understand fisher behaviour in the face of a changing environment in order to manage the system better. For instance, if managers impose regulations, they will ultimately change the behaviour of the fisher. Such a change may have great implications for fishers economically, socially, ecologically, and geographically. If the resulting behaviour is not understood correctly, ineffective management outcomes and conflict can be expected. However without detailed knowledge of variation in behaviour among fishers, managing fisheries becomes very difficult. Kuperan and Raja Abdullah (1994) described the social implications of small scale fishers competing with illegal commercial fishers who are let down by governing institutions, and as such make management difficult. Poor international regulations and enforcement are said to be the single most important drivers for IUU fishing (Agnew and Barnes, 2004). Nevertheless, the variability in compliance between fishers and fisheries is poorly understood (Branch, 2006).

There are several enforcement theories in the economics literature; see, e.g., Becker and Stigler (1974), or more recently Polinsky and Shavell (2001) which researches corruption amongst law enforcers and the level of honesty for a given bribe. Becker and Stigler (1974) found that law enforcers under supervision and are paid a higher wage would be less corrupt. In terms of fisheries studies Sutinen and Anderson (1985) based their theories within the field of the economics of law, whereby non-compliers maximise the benefits of their law-breaking compared to the expected probability of being caught. However, an early empirical study by Furlong (1991) compatible with the seminal economics of crime paper by Becker (1968) showed signs that expected fines for violations and monitoring have a positive effect on deterring offences. Fisheries managers are beginning to look at how social and normative factors influence the fisher behaviour and compliance. Recent compliance studies in the literature investigate personal characteristics such as age and years within the fishery, fairness of regulation and the risk and benefits associated with illegal activity. For example Hatcher *et al.* (2000) applied a binary model to data and discovered factors affecting UK fishermen's violation and compliance towards quotas including risks with respect to detection, and the expected financial penalty. The social outcomes of the study showed that the fishers felt they would adhere to regulations if they were involved in their implementation. Nielsen and Mathiesen (2003) studied factors influencing rule compliance in Danish fisheries, they concluded that there where 6 important factors that influenced compliance and non-compliance; (i) economic gains to be obtained, (ii) deterrence and sanctions, (iii) compatibility between regulations and fishing practices and patterns, (iv) efficacy of imposed regulations, (v) norms (behaviour of other fishers) and morals, (vi) perception of being part of the decision-making process (indication). Some of their findings complemented results from a UK survey conducted by Hatcher and Gordon (2005) who found that quota regulations were deemed unfair and treated with a lack of respect. Responses showed that more than one-quarter of UK fishers exceeded their quotas by >20%, and that just one-fifth never exceeded their quotas. The financial constraint of the quota was responsible for them exceeding the quota in order to stay in business. There was also strong evidence that, generally, the EU was the least respected regulatory authority and that local institutions were the most respected. For example two-thirds of UK fishers who were members of producer organizations (POs) were more likely to comply with the quota laid down by the PO because they had more involvement in the regulatory process. However few empirical studies exist in the area of compliance and enforcement for fisheries, which is surprising since it is an important aspect of managing fisheries.

2.1.4 Implementation error

Of increasing importance to fisheries science and management is the ability to anticipate fisher behaviour in response to management regulation, in order to reduce implementation error (Fulton *et al.,* 2011), i.e. where the effects of management differ from that intended. An example of implementation error is where fishing effort is redistributed following a spatial closure to protect a stock (or cohort), in a way that was not anticipated by management. As a general statement: all management plans (e.g. using TACs to reduce fishing mortality) seem like a good idea in advance and they mainly fail due to the response of the fisher (e.g. they may mis-report or high-grade). Implementation error is increasingly being integrated into the discussions that include stakeholder involvement in the design of management plans, i.e. spatial and temporal closures, effort control and discard management. Implementation error usually arises from a combination of non-compliance with regulations by harvesters, changing catchability, and other dynamic processes in the fleet (Peterman, 2004), in addition to changing tactics and strategies in terms of location choice.

2.1.5 Fisher tactics: location choice

Many factors influence a fisher's decision where and when to fish, including fish distribution, fuel price, regulations, their habits and experience, previous catch rates, market prices, and the proximity to landing ports. These factors can lead to differences in observed individual fisher behaviour and the way a group of fishers (a fleet) allocate their effort in time and space. Several studies have looked at behavioural aspects of the way fishers spatially allocate their effort (Rijnsdorp *et al.,* 2000; Hilborn *et al.,* 2005; Smith *et al.,* 2009). An important element influencing fisher behaviour is stock density, because fishers tend to have prior knowledge (Begossi, 2001) of resource distribution and habitat (Hilborn and Ledbetter, 1979; Gillis *et al.,* 1993; Pet-Soede *et al.,* 2001). Catch rates are related to stock density and will have a large impact on fisher behaviour (Eales and Wilen, 1986; Marchal *et al.,* 2006). This means that fishers will gravitate towards areas where catch rates are greatest, and gravity models have been specified and applied to model fishing vessel spatial distribution (e.g. Walters and Bonfil, 1999). Economic factors and management measures in the form of technical measures (size restrictions or gear restrictions; Bene and Tewfik, 2001), marine protected areas (MPAs), and spatial closures may also force fishers to search for new fishing grounds (Hutton *et al.,* 2004).

Over the past few years, much attention has been paid to predicting fisher location choice by applying models of discrete choice as the common assumption with these models is that of 'random utility maximisation' (RUM) (Anderson *et al.,* 2010; see Section 2.2.1). Simply, the main assumption is that a fisher, who is faced with a finite set of locations to choose from, chooses the location which gives the maximum amount of utility. Predicting fisher behaviour using discrete choice models has increased in popularity with the increasing availability of appropriate data (vessel-by-vessel trip data), because such models offer an opportunity to study individual behaviour at finer resolutions of time and space than other techniques (Coglan *et al.,* 2004; Hutton *et al.,* 2004). These models can be applied to various theoretical policy scenarios (See Table 2.1 for studies), which can also be simulated.

2.1.6 Fisher tactical response to management

It is widely recognised that decision makers and managers now require an ecosystem-based approach to address current interlinked problems for social well-being (FAO, 2003b). Understanding and predicting choice of fishing area can help fisheries managers when implementing management plans to protect marine resources, reducing the impact of spatial closures on the economic viability of a fishery and reducing also the unintended side-effects. There have been numerous discussions by fisheries managers on the benefits of MPAs or spatial closures for managing fisheries (e.g. larger fish, an increase in spawning stock biomass, protection of endangered habitat (Holland and Brazee, 1996)), because they are, apparently, relatively cost-effective and a useful tool and "safe bet" (insurance) against management failure (Lauck *et al.,* 1998; Sumaila, 1998; Murray *et al.,* 1999) to protect resources. Although they are widely advocated and increasingly used around the world's tropical and temperate fisheries, very few studies have explored the implications of MPAs for subsequent fisher behaviour, including spatial displacement, and therefore the extent of the biological impact (Sanchirico and Wilen, 2002; Smith and Wilen, 2003; Wilcox and Pomeroy, 2003). Empirical studies have shown that if the MPA is implemented for long enough, marine life may recover in terms of size, fecundity and abundance within its boundaries (Hixon, 2002; Halpern, 2003). Halpern (2003) and Alcala *et al.* (2005) have discussed the benefits of MPAs in relation to poor or little scientific information, depleted stocks and situations when enforcement is insufficient.

A number of models have been developed to assess the effects of reallocation of fishing effort on target stocks (Horwood *et al.*, 1998; Steffansson and Rosenberg, 2006), but knowledge of the biological effects of displacement of fishing effort outside the boundaries of MPAs is not well developed (Jennings, 2009). Simulation models have been used widely to validate hypotheses on the design of MPAs, and to evaluate their biological benefits (Botsford *et al.*, 2007). In order for MPAs to work successfully for a fish species, there needs to be a complete cessation of fishing activities (or those fishing activities that affect the species in question) within a sufficiently large area (Steffansson and Rosenberg, 2005), protecting the encompassed species and allowing stock recovery.

Table 2.1 Summary of fisher tactical and strategic studies in the literature showing the different factors influencing fisher decisions.

Factors used in this study	**References**
Expected revenue/catch	Bockstael and Opaluch, 1983; Mangel and Clark, 1983; Eales and Wilen, 1986; Dupont, 1993; Gillis, 1993, 2003; Prince and Hilborn, 1998; Holland and Sutinen, 1999; Walters and Bonfil, 1999; Mistiaen and Strand, 2000; Smith, 2000; Eggert and Tveras, 2001; Fonteneau and Richard, 2003; Swain and Wade, 2003; Bertrand, 2004; Hutton *et al.,* 2004; Pradhan and Leung, 2004; Salas and Gaertner, 2004; Murawski *et al.,* 2005; Anderson and Christensen, 2006; Marchal *et al.,* 2006; Vermand *et al.,* 2008; Ran *et al.,* 2011
Operating costs (e.g. distance and fuel prices)	Eales and Wilen, 1986; Dupont, 1993; Holland and Sutinen, 1999; Curtis and McConnell, 2004; Hutton *et al.,* 2004; Little *et al.,* 2004; Pradhan and Leung, 2004; Smith, 2005; Hicks and Schnier, 2006; Valcic, 2009; Venables *et al.,* 2009; Ran *et al.,* 2011
Vessel aggregation	Dasgupta and Heal, 1979; Walters and Bonfil, 1999; Pradhan and Leung, 2004
Compliance	Becker, 1968; Sutinen and Anderson, 1985; Sutinen and Gauvin, 1989; Furlong, 1991; Charles, 1993; Kuperan and Raja Abdullah, 1994; Hatcher *et al.,* 2000; Arnason, 2003; Hatcher and Gordon, 2005
Management (e.g. MPAs,quotas, capacity and effort)	Frost *et al.,* 1995; Holland and Brazee, 1996; Horwood *et al.,* 1998; Lauck *et al.,* 1998; Sumaila, 1998; Murray *et al.,* 1999; Sanchirico, 1999; Walters and Bonfil, 1999; Rijnsdorp *et al.,* 2001; Hixon, 2002; Sanchirico and Wilen, 2002; Botsford *et al.,* 2003; Halpern, 2003; Halpern and Warner, 2003; Smith and Wilen, 2003; Wilcox and Pomeroy, 2003; Hutton *et al.,* 2004; Salas and Gaertner, 2004; Alcala *et al.,* 2005; Hatcher and Gordon, 2005; Murawski *et al.,* 2005; Steffansson and Rosenberg, 2005, 2006; Botsford *et al.,* 2007; Jennings, 2009; Needle and Catarino, 2011
Vessel characteristics (e.g. age and length)	Mistiaen and Strand, 2000; Rijnsdorp *et al.,* 2000; Pradhan and Leung, 2004; Mardle *et al.,* 2005, 2006; Murawski *et al.,* 2005;
Habit or experience (past effort)	Hilborn and Ledbetter, 1979; Allen and McGlade, 1986; Gillis *et al.,* 1993; Vignaux, 1996; Dorn, 1997; Campbell and Hand, 1999; Begossi, 2001; Pet-Soede *et al.,* 2001; Hutton *et al.,* 2004; Salas and Gaertner, 2004; Anderson and Christensen, 2006
Personal characteristics (e.g. skipper age, years fishing)	Sutinen and Gauvin, 1989; Furlong, 1991

For example, with cod at a historical all-time low and as part of the North Sea cod recovery programme, the European Commission closed a large selection of spawning grounds in the North Sea for 10 weeks to protect spawning cod during 2001 (EC, 2001). Fishing vessels confronted with this closed area redistributed their effort or simply relocated. Rijnsdorp *et al.* (2001) found that the Dutch beam trawl fleet concentrated fishing along the borders of the closed area, with these observations consistent with what would be expected from other studies (Botsford *et al.,* 2003;

Halpern and Warner, 2003; Halpern *et al.,* 2003). Hutton *et al.* (2004) applied a discrete choice model to the fishery and considered the implications of the 2001 closure. Unfortunately, owing to the short period of its implementation, the closure unsurprisingly showed no positive results for cod. It was also suggested that it had a negative impact on the rate of discarding for demersal species and had created additional damage to benthic communities by forcing continued trawling over the same fishing grounds, and displacing effort to some areas that were previously little fished. Fishing outside the closure, also known as 'fishing the line' can have its benefits, in terms of increases in yield and mean revenues per tow (Murawski *et al.,* 2005). The overall outcome in terms of the management objectives associated with implementing the MPA with respect to the protected stock or the fishery are less clear (Walters and Bonfil, 1999).

2.1.7 Fisher strategic behaviour: Entry/Exit

Pioneering research in fisheries economics (Gordon, 1954; Scott, 1955) presented equilibrium models based on the rate of fishers entering and exiting common property fisheries. Those authors argued that fishing effort would increase, with new vessels entering, as long as the fishery remained profitable. In contrast, as profits declined, vessels were assumed to exit the fishery if they could achieve greater returns on their capital investment elsewhere. These classic models assume that fishing effort and boats can move freely in and out of fisheries as a result of open access to a stock and to other stocks in other fisheries, or of uses for other purposes than fishing. In many areas, however, fisheries are managed by a limited licensing system, thereby constraining the ability of individuals to move in and out of fisheries. Entry is restricted by the availability of licences or quota and capital, and exit is made more difficult because there are limited alternative uses for the boat, which cannot be used in any other production process, so is not malleable (Clark *et al.,* 1979). Additionally, opportunities for exploiting other fishing grounds and/or stocks will depend on regional biogeography. There is an extensive literature on the theoretical economics of entry–exit schemes within industrial organizations (e.g. Scherer and Ross, 1990), but very little empirical work.

Most industry research that has considered the dynamic nature of a firm has concentrated on new entrants, and views exit predominantly as a symbol of failure (Jovanovic, 1982; Hopenhayn, 1992; Jovanovic and MacDonald, 1994). Various authors have suggested that firms exit in one of two ways (Holmes and Schmitz, 1990; Agarwal and Gort, 1996; Dunne *et al.,* 2005; Plehn-Dujowich, 2009). Firstly, a firm could terminate its operations and sell its assets at salvage value, or secondly, it could exit its current business and reallocate the assets and know-how towards another line of business.

Fishing firms, be they owner–operator or larger firms, behave in a similar way, but with greater uncertainty, attributable to changing stock levels, management regulations, market prices, and fuel costs. Hence, the decisions of vessel operators to stay in, enter into, or exit from a fishery are influenced by a combination of economic and biological factors, as well as personal reasons (e.g. family tradition). In studies on the North Sea flatfish fishery, Mardle *et al.* (2005, 2006) showed that vessel age, realized and expected revenues, and the status of the main target species had a bearing on the decision for a vessel to participate in a fishery. Dasgupta and Heal (1979) stated that ever-increasing numbers of competing fishers led to externalities, i.e. the more one catches, the less is available to others, so that each operator believes that none of its competitors will adhere to a future conservation policy and in turn sees no benefit to pursue it personally.

As fleet size expands, average landings and revenues per vessel decline along with catch per unit effort (cpue), the costs of fishing effort increase, and resource rents dissipate (Ward and Sutinen, 1994). Pradhan and Leung (2004) showed a decision to enter or exit a fishery depended on the expected benefits, which included future revenue, the stock status of the main target species, and crowding effects (the number of vessels in the fleet). They also described how operators make decisions on the basis of age of the vessel; an older vessel is less likely to enter a fishery than a newer vessel; decommissioning and exit probabilities increase with age and are possibly the result of vessels being older and inefficient, and hence more costly to run which highlights the importance of fuel costs and the potential effects of fuel subsidies. Fuel subsidies have a direct effect on fishing effort (Sumaila *et al.,* 2006). They are controversial, because they encourage wasteful, uneconomic fishing practices in already overcapitalized fisheries, and they maintain fishing effort even when stock levels decline.

2.1.8 Fisher strategic response to management

The perceived overcapacity and the declining stocks in the North Sea require policy action. However the considerations above mean that a decommissioning scheme may not be the most cost-effective method of reducing capacity. Frost *et al.* (1995) studied the effect that decommissioning schemes had on fishing mortality; however no relationship was conclusively found, presumably due to the continued excess capacity in the fishery, such that the reduction in fishing mortality was negligible. Decommissioning tended to lead to older, less-efficient boats being removed from the fleet, and consequently created controversy as the more-efficient boats continued to fish (Nautilus Consultants, 1997). It is generally accepted that older vessels catch fewer fish than more modern ones (Seafish, 1989). For those efficient vessels, the strategy was to stay in the fishery, owing to the financial incentive to decommission not being high enough and the possibility that future rewards could potentially be higher (Nautilus Consultants, 1997). This meant that decommissioning created a fleet dominated by modern, efficient vessels, essentially failing to reduce capacity and hence reduce fishing mortality, especially because the quota for the decommissioned vessels' fish made its way back into the pool of quota entitlements that were traded and/or leased (Shotton, 2001). Decommissioning is said to be only a short-term incentive for solving the underlying overcapacity issue, although some advocate it as the only solution (Lindebo, 2005). However, it cannot solve the inherent incentive problem of over-investment and technical creep (Weninger and McConnell, 2000; Mahévas *et al.,* 2004; Marchal *et al.,* 2006, 2007; Millischer *et al.,* 1999). Subsidies and technical creep have caused difficulties when applying effort limitations due to the ongoing modernisation of vessels funded by government grants and subsidies. Technical creep makes it difficult to fix a capacity constraint, particularly in the context of fluctuating stocks (Jensen, 1999). TACs and effort limitation measures which impact on revenues also affect future stock levels, because a TAC is based on the proportion of mature fish that can be harvested from a stock. Therefore, if the proportion of mature older fish, (known as the spawning stock biomass (SSB)) falls by a set amount, this will lead to proportional reductions in TAC, quota, revenue and thereby impact the proportion of vessels that enter or exit the fleet.

2.2 Modelling approaches for investigating fisher behaviour

2.2.1 An empirical method for studying fisher behaviour

Predicting fisher behaviour using Random Utility Models (RUM) has increased in popularity with the increasing availability of appropriate data, and offers an opportunity to study individual behaviour at finer resolutions of time and space than other frameworks (Coglan *et al.,* 2004; Hutton *et al.,* 2004, Bockstael and Opaluch, 1983; Eales and Wilen, 1986; Holland and Sutinen, 1999; Wilen *et al.,* 2002; Branch *et al.,* 2006; Vermand *et al.,* 2008; Anderson *et al.,* 2010, Pradhan and Leung, 2004). The key characteristics of RUMs are that they model discrete decisions. RUMs can be described as follows: A decision-maker is faced with making a choice among a number of alternative options, obtaining differing levels of utility from each alternative option, and tending to choose one that maximizes utility. As such, and like other economically-based choice models, utility influences individual choice with a deterministic and stochastic error component. Prior to implementation in fisheries behaviour models, RUMs were used in the travel industry to analyse the behaviour of consumers of transportation services and facilities (McFadden, 1974; Ben-Akiva and Lerman, 1985). These models can also be applied to theoretical policy scenarios, and to simulate longer term strategies (e.g. the choices made year by year) in relation to the availability of decommissioning grants, stock status, catch quotas, investment, and other key factors which can play a critical role in the decision of a fisher. For example, Pradhan and Leung (2004) used revenue by gross tonnage within a multinomial logit framework to model exit and entry strategies of Hawaiian longliners.

2.2.2 Theoretical methods for studying fisher behaviour

Prior to the application of RUMs, most models assumed the ocean to be a homogenous space in which fish were uniformly distributed and fishing locations were identical. Sanchirico and Wilen (1999) incorporated space into their renewable resource model and developed a theory that provided insight and tested hypotheses about how decision-makers behave in heterogeneous space, in terms of resource availability and the cost of fishing. Their model described the behavioural dynamics under conditions of open access, including both temporal and spatial aspects and incorporating the behaviour of the fleet and the fish stocks. Their analyses suggested that fishing effort across a system of interconnected spatial patches is driven by bio-economic conditions in each patch, and the biological dispersal between patches. In patches where costs were high or the catchability and prices low, effort moves away, affecting in turn the distribution of stocks. Economic theories suggest that the distribution of fishing effort will be determined by the expected economic returns to individual fishers from fishing in alternative fisheries or locations (Gordon, 1954). In ecology, animals are thought to behave in a similar way to fishers, whereby they assess the patch quality, as they know how profitable each available patch is. This theory is known as the Ideal Free Distribution, (IFD; Fretwell and Lucas, 1970; Fretwell, 1972) and is a hypothesis predicting the ways in which animals distribute themselves among several patches of resources. The term 'ideal' indicates that the animals have perfect knowledge in their assessment of the patch and 'free' implies freedom of moving unhindered from patch to patch. In terms of the application in fisheries, fishers have ideal knowledge of their 'target' fish distributions and are free to move between fishing grounds unrestricted (Gillis, 1993; Gillis, 2003), this is unrealistic of course but has worked quite well in some instances.

2.3 Conclusion and the way forward

In Europe under the Common Fisheries Policy (CFP), previous attempts have been made to control fishing capacity via multi annual guidance programmes (MAGP's) and fishing effort targets whereby the reduction in capacity and effort will lead to a reduction in mortality on the stock or stocks. The links between fishing mortality, effort and capacity are important for fisheries managers to understand and to quantify, in order for sustainability. In 2013, expected changes to the CFP are requiring member states to develop cooperation among the various stakeholders and to rationalise and devolve management of fisheries (EC, 2009a). Whilst it is necessary to build on existing capabilities to ensure implications of our advice are explained in socio-economic terms as well as environment, many tools are necessary to improve cooperation amongst various stakeholders still need to be developed. For example to implement Results Based Management (RBM), an alternative management framework to the current over-centralised and top-down legislative process of the CFP needs to be developed. In the case of RBM, where strategic decisions would continue to be taken centrally in Brussels, but decisions relating to delivery and implementation could be delegated to regional bodies, subject to central auditing of outcomes. Within RBM there is a need for stakeholders to be able to evaluate alternatives prior to implementation to show that management objectives can be met within a cost effective and equitable framework. This research will demonstrate a better understanding of the links between fishing mortality, effort and capacity and as such provide the basis for the development for alternative direct conservation tools in order for sustainability.

Chapter 3. Exit and entry of fishing vessels: an evaluation of factors affecting investment decisions in the North Sea English beam trawl fleet[2]

Beam trawler fishing (Source: Jim Ellis).

[2] Tidd, A.N., Hutton, T., Kell, L.T., Padda, G. (2011). Exit and entry of fishing vessels: an evaluation of factors affecting investment decisions in the North Sea English beam trawl fleet. ICES Journal of Marine Science, 68: 961–971

Abstract

A profitable fishery attracts additional effort (vessels enter), eventually leading to overcapacity and less profit. Similarly, fishing vessels exit depending on their economic viability (or reduced expectations of future benefits) or encouraged by schemes such as decommissioning grants and/or when there is consolidation of fishing effort within a tradable rights-based quota system (e.g. individual transferable quotas, ITQs). The strategic decision-making behaviour of fishers in entering or exiting the English North Sea beam trawl fishery is analysed using a discrete choice model by integrating data on vessel characteristics with available cost data, decommissioning grant information, and other factors that potentially influence anticipated benefits or future risks. It is then possible to predict whether operators choose to enter, stay, exit, or decommission. Important factors affecting investment include vessel age and size, future revenues, operating costs (e.g. fuel), stock status of the main target species, and the impact of management measures (e.g. total allowable catches, TACs) and total fleet size (a proxy for congestion). Based on the results, the predicted marginal effects of each factor are presented and the impact of each is discussed in the context of policies developed to align fleet capacity with fishing opportunities.

3.1 Introduction

Pioneering research in fisheries economics (Gordon, 1954; Scott, 1955) presented equilibrium models based on entering and exiting common property fisheries. Those authors argued that fishing effort would increase with the entry of new vessels as long as the fishery remained profitable. In contrast, as profits declined, vessels were assumed to exit the fishery if they could achieve greater returns on their capital investment elsewhere. These classic models assume that fishing effort and boats can move freely in and out of fisheries as a result of open access to a stock and to other stocks in other fisheries, or be used for other purposes than fishing. In the UK, however, fisheries are managed by a limited licensing system, thereby constraining the ability of individuals to move in and out of fisheries. Entry is restricted by the availability of licences or quota, and exit is made more difficult because there is limited alternative use for the boat, which cannot be used in any other production process, so is not malleable (Clark *et al.,* 1979). There is extensive literature on the theoretical economics of entry–exit schemes within industrial organizations (e.g. Scherer and Ross, 1990), but very little empirical work. Most industry research that has considered the dynamic nature of a firm has concentrated on new entrants, and views exit predominantly as a symbol of failure (Jovanovic, 1982; Hopenhayn, 1992; Jovanovic and MacDonald, 1994). Various authors have suggested that firms exit in two ways (Holmes and Schmitz, 1990; Agarwal and Gort, 1996; Dunne *et al.,* 2005; Plehn-Dujowich, 2009). First, a firm could terminate its operations and sell its assets at salvage value, and second, it could exit its current business and reallocate the assets and know-how towards another line of business. Fishing firms, be they owner–operator or larger firms, behave in a similar way, but with greater uncertainty attributable to changing stock levels, management regulations, market prices, and fuel costs. Hence, the decisions of vessel operators to stay in, enter into, or exit from a fishery are influenced by a combination of economic and biological factors, as well as personal reasons.

In other studies on the North Sea flatfish fishery, Mardle *et al.* (2005, 2006) showed that vessel age, realized and expected revenues, and the status of the main target species had a bearing on the decision for a vessel to participate in a fishery. We extend these analyses, providing additional data on the rates of decommissioning and the costs of fishing, by including fuel-price data and data on the catches of sole (*Solea solea*) and anglerfish (*Lophius spp.)* separately, in addition to the main target stock, plaice (*Pleuronectes platessa*). Apart from voluntary decommissioning schemes, the option to trade quota (with an individual transferable quota, ITQ, system in the Netherlands and a quasi-ITQ system in the UK, described below) provides the opportunity for fleet rationalization in this case study. Therefore, within the context of non-market and market means to reduce capacity, we evaluate here the choices available to fishers and their responses, either to (i) stay in a fishery, (ii) exit, (iii) decommission, or (iv) join and enter the fishery. In both exit and decommission options, the vessel is assumed to leave the fishery, but in the case of decommissioning, a premium is paid to the owner. Here, we discriminate between the two options in order to test whether different factors affect each uniquely.

Under the Common Fishery Policy of the EU, each Member State has a fixed proportion of a species quota, referred to as relative stability (based on that country's historical access rights) and, apart from minor deviations from the rule, each boat in the UK has quota which is a proportion of the country's share (European Commission, 1996, 1997, 1999). Also in the UK, a quasi-ITQ system exists where quota entitlements and their trade are administered (or at the very least recorded) by Producer Organizations (POs), but the government has never endorsed a system of fully tradable harvest rights. Prior to 1999, the English fleet was managed by licence and quota restrictions, where quotas could be transferred to other fishing vessels within the POs. Quota could be leased but not permanently traded, although occasionally the government allowed once-off permanent trades (within and across POs) in order to rid the system of all leasing arrangements that had become permanent. Post-1999, quotas were allocated directly to vessels, as a Fixed Quota Allocation (FQA). While being a fixed nominal amount of quota rather than a proportion of the total country total allowable catch (TAC), FQAs could be traded by individuals on a permanent basis or leased annually. Compared with the management arrangements in the UK, flatfish fisheries in the Netherlands before the 1990s were managed on an Individual Quota (IQ) system, whereby IQs could not be sold permanently or leased because it was suggested that quotas would be concentrated in an undesirable way (Smit, 2001). In the early 1990s, a new policy was adopted, with groups of vessels operating within a PO framework given full quota-management responsibilities. The fishers within those groups pooled their ITQs and days at sea, allowing the PO board to control the transfer of ITQs and days at sea on a permanent basis (van Hoof, 2010).

In the UK, apart from management via quotas, as just described, a system of vessel capacity units (VCUs), based on size and engine power, was implemented in order to administer fishing capacity. Attempts were made in the 1980s, 1990s, and in 2002 to reduce fishing capacity and effort through Multi-Annual Guidance Programmes (MAGPs), with many countries including the UK implementing decommissioning schemes (European Court of Auditors, 1994, 1997). The MAGPs were funded by various financial instruments. This funding significantly reduced vessel numbers in the UK, decommissioning 225 vessels between 1984 and 1986, under MAGP I (Pascoe *et al.,* 2002). Then, between 1987 and 1991 under MAGP II, another 686 vessels were decommissioned in order to cut engine power tonnage and effort. MAGP III, introduced in 1992, ran for five years and resulted in the removal of another 578 vessels. Then, from 1997 to 2002 under MAGP IV, another ~170 boats were decommissioned based on fleet segment and the extent of overexploitation of targeted stocks. Capacity control since the end of MAGP IV has been replaced by effort ceilings, controlled by rules for entry and exit. Simply, a vessel can only enter a fishery when the equivalent capacity has exited. Decommissioning tended to result in older, less-efficient boats being removed, creating a modern, efficient fleet, essentially failing to reduce capacity and hence reduce fishing mortality, especially with the quota for decommissioned vessels making its way back into the pool of quota entitlements that were traded and/or leased.

Here, we assume that investment (or dis-investment) decisions are related primarily to actual or expected profits and the availability of decommissioning schemes. However, because the computation of individual profits requires detailed cost data, which is difficult to obtain and in many cases confidential, revenues are utilized as a proxy for economic viability. Pradhan and Leung (2004) used revenue by gross tonnage within a multinomial logit framework to model exit and entry strategies of Hawaiian longliners. Given the value of information in their results, and using a random utility framework provided by McFadden (1974), we accommodate a multinomial logit model (unordered) and evaluate the probability of vessels to enter, stay, exit, or decommission from the English North Sea beam trawl fleet. This information is used to evaluate potential alternative management strategies, and significant factors influencing investment are discussed in the context of policies developed to align fleet capacity with fishing opportunities.

3.1.1 The English North Sea beam trawl fleet

In the North Sea, English vessels that target flatfish have traditionally caught plaice in a directed beam trawl fishery using 120 mm mesh north of 56°N (Figure 3.1), and in a mixed fishery targeting sole, using 80 mm mesh in the southern North Sea. In 2006, international landings of plaice in the North Sea amounted to 57 943 t, well below the peak of 170 000 t in 1989. Some 40% of the total international landings of plaice were reported by Dutch vessels. The UK accounted for 23% of the plaice landings, Denmark for 20% of the landings, and Belgium, Germany, France, and other countries for the remaining 17% of the total landings. Of international sole landings in 2006 (12 600 t), 71% were made by the Netherlands, 8% by Belgium, and the balance of 21% by France, Germany, the UK, and Denmark. Sole landings by beam trawlers in the early 1990s were dominated by two good year classes and yielded ~32 000 t in total. The combined 2006 total first sale value of these two flatfish species was estimated at ~€350 million, of which €140 million was from plaice sales. The English beam trawl fleet expanded in the early 1990s, through investment in newer trawlers and an expansion of the beam trawl fleet in English east coast ports. From the mid-1990s, fleet size dropped, either as a consequence of older vessels leaving the fishery or declines in the value per unit effort (vpue) of plaice at the same time, or both. Exploitation rates of plaice and sole in the North Sea (Figure 3.2) clearly follow the trend in the number of vessels in the fleet (Figure 3.3). Between 2000 and 2005, increases in the fuel price (31.1%) reduced the viability of many fishing operations. Beam trawling is fuel-intensive because heavy gear is dragged relatively fast over the seabed. Until 2003, the English North Sea beam trawl fleet operated mainly out of English east coast ports in ICES Division IVb and c, typically spending an average of 250 days at sea annually on 6-d trips (Hutton *et al.,* 2004). Towards the end of 2002, an English east coast beam trawl company ceased fishing as fishing became unprofitable, claiming that it was not economically viable to catch fish for which they had a quota entitlement, that prices were poor, and that fuel costs were burgeoning as vessels had to operate far from port, near the Norwegian sector, in order to catch their quota (Hansard, 2002). At this time the company operated eight vessels (down from 12 a few years earlier). Subsequently, the vessels were first leased then sold to Dutch operators, but they retained their English flag and quota entitlement. The relocation of many of the larger beam trawlers to Dutch ports provided an opportunity for rationalization as quota allocations for two or more vessels were transferred to newer fishing vessels by Dutch firms. When fishing out of English ports, English beam trawlers generally chose to target both plaice and sole, but in recent years, Dutch skippers increasingly targeted sole because of its greater commercial value and the proximity of the sole fishing grounds in the southern North Sea, generally ICES Division IVc (Figure 3.1), to ports in the southern Netherlands.

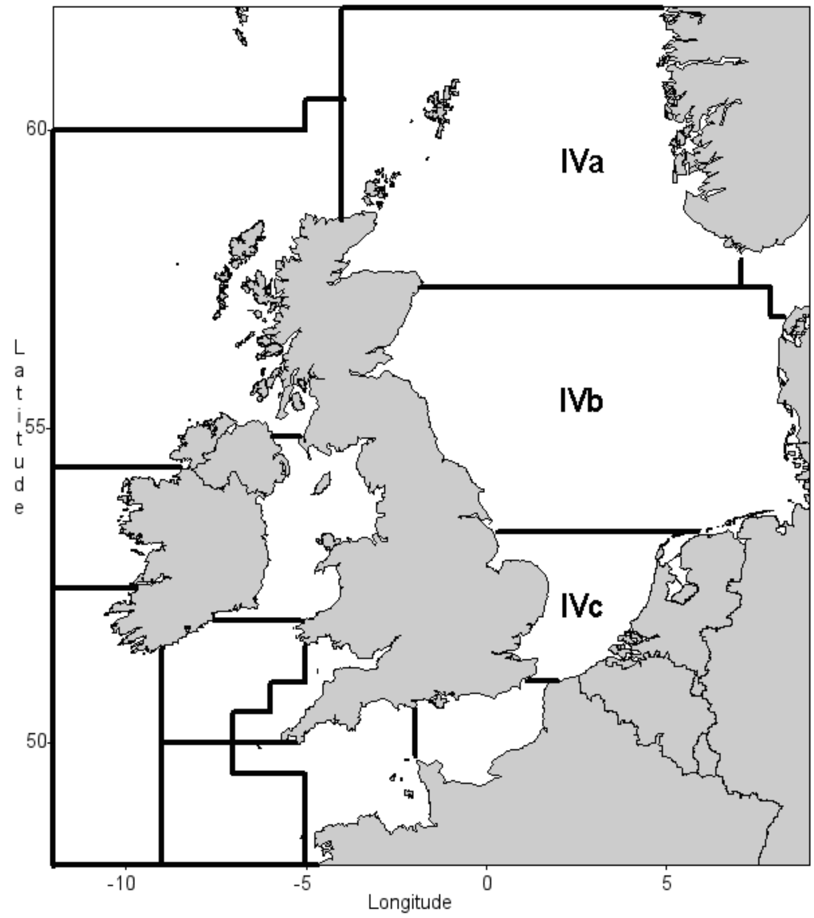


Figure 3.1 The study area (ICES Divisions IVb and IVc).

This change in tactical behaviour is evident in Figure 3.2, where from 2002 on, there is an increase in the exploitation rate for sole. We postulate that Dutch skippers acquired additional quota to fish for sole, but we do not have data on individual vessel quota entitlements to provide evidence of this transfer. A summary of physical vessel characteristics (over the period 1989–2007) is presented in Figure 3.3. The number of vessels within the fleet in a given year varies considerably. In 1993, for example, there were 152 vessels, but by 2007 just 29 remained, a considerable reduction in fleet size over just 15 years.

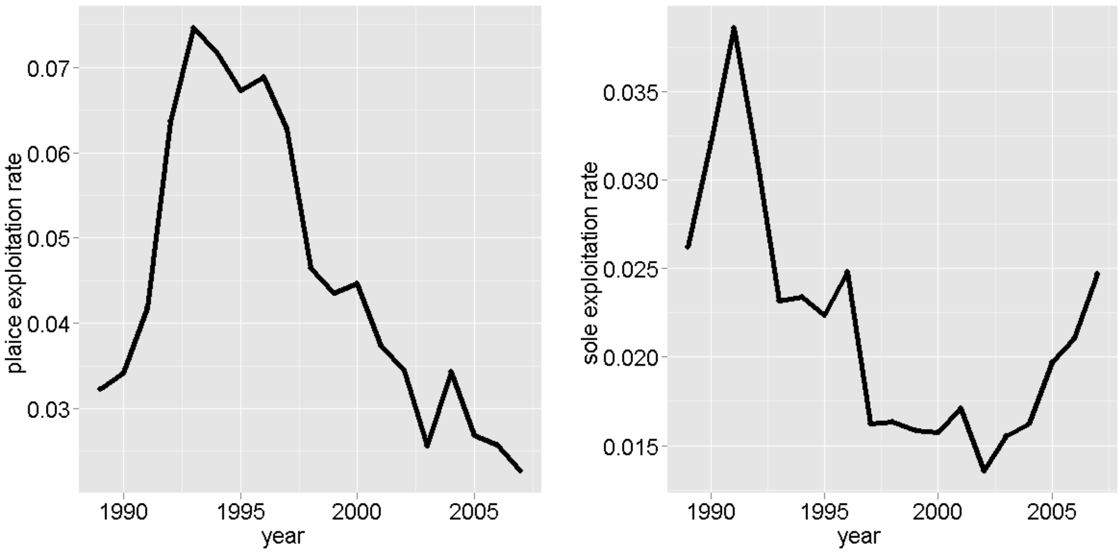


Figure 3.2 Exploitation rates (landings of the North Sea English beam trawl fleet, divided by SSB) for plaice (left) and sole (right).

From the summary statistics, it is apparent that there has been a slight increase in average vessel age, implying that as the number of vessels decreased, few newer vessels entered the fishery. Also noticeable from the fleet statistics is that the average vessel power, tonnage, VCUs, and length all increased slightly, suggesting that less powerful, smaller vessels left the fleet. Over this period too, beam trawlers were purchased originally from the Dutch, operated out of English ports for a while, before being purchased back again by the Dutch from the English. Also, some fishing vessels were occasionally tied up in ports in the Netherlands for more than a year at a time, awaiting engine refits. Observed decisions for English North Sea beam trawlers to exit, enter, decommission, or stay for the period 1989–2007 are presented in Figure 3.4. By 2007, the fleet consisted of 29 vessels that were part of the stay group, with four entering, compared with a peak of 152 in 1993, of which 32 entered, 19 exited, 88 stayed, and 13 left through a decommissioning scheme.

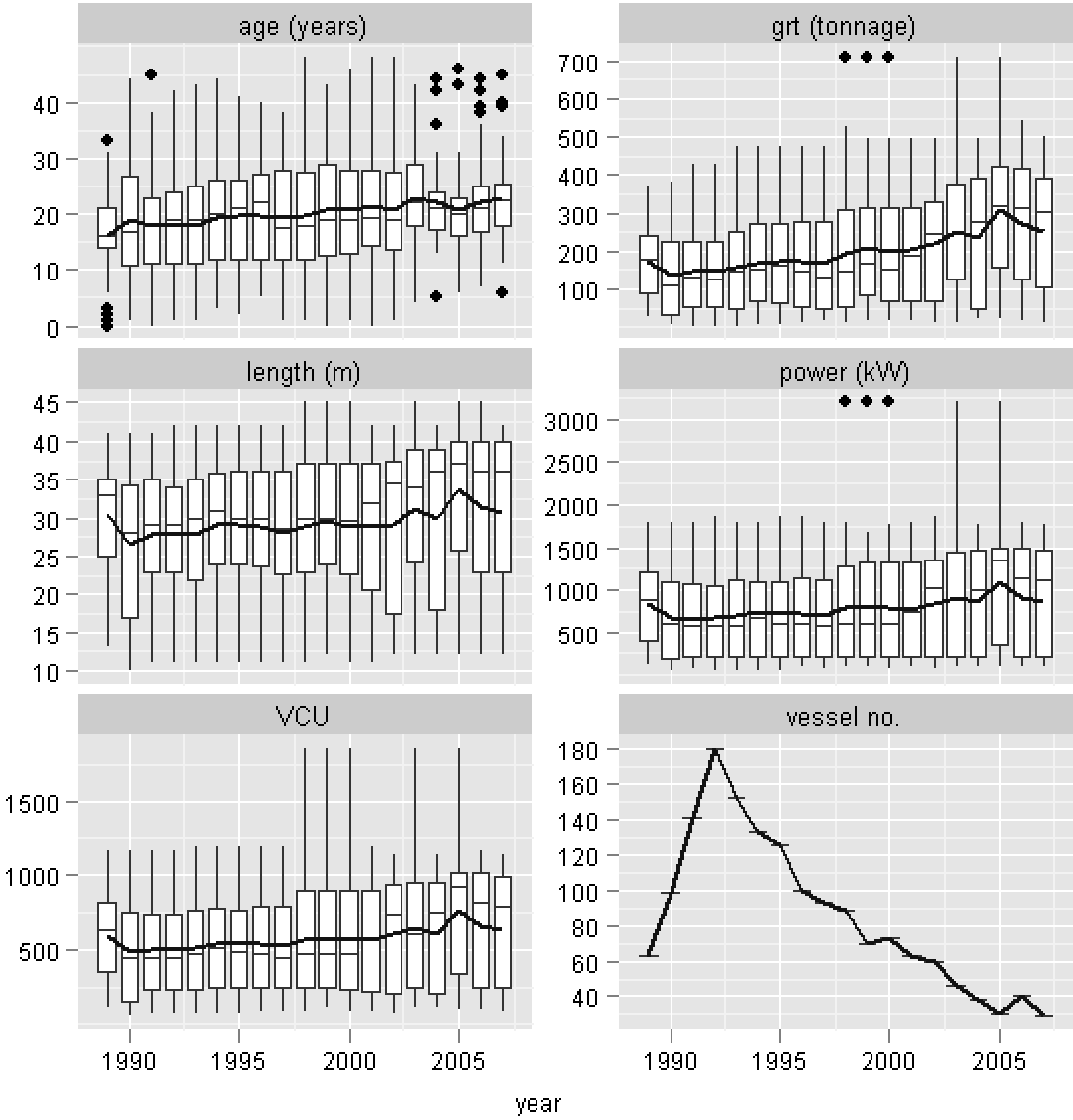


Figure 3.3 Box and whisker plots of vessel characteristics over the study period, the line representing the mean, the horizontal bar the 50th percentile, the top of the box the 75th percentile, and the base of the box the 25th percentile. Whiskers represent the range of data, and the solid diamonds are outliers.

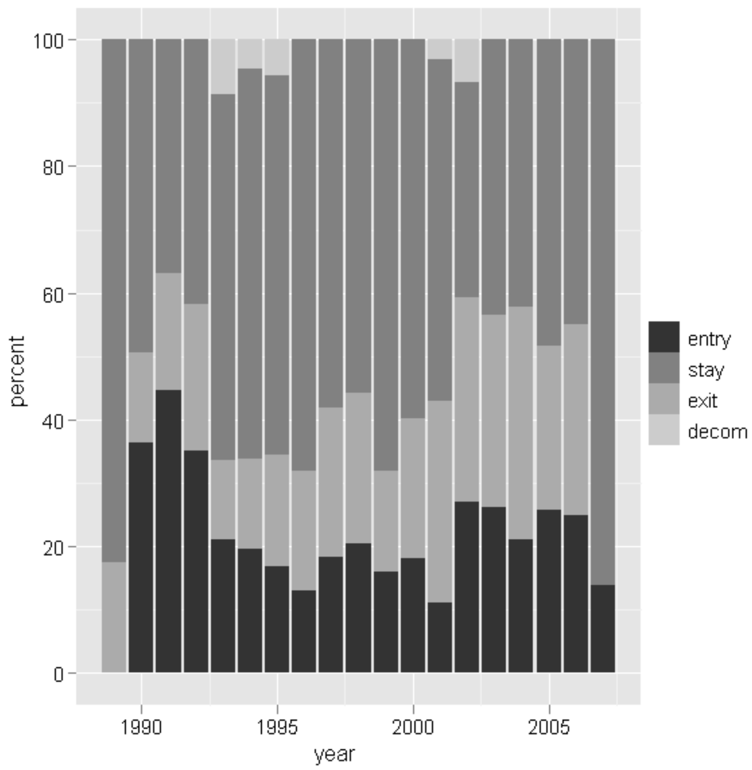


Figure 3.4 Exit, enter, stay, and decommission decisions observed in the study fishery over the period 1989–2007.

3.2 Material and methods

3.2.1 Data

The UK Department for the Environment, Food and Rural Affairs (Defra) database for fishing activity and the fleet register were used to select commercial landing and vessel data of English (and Welsh) beam trawl fleets operating in ICES Divisions IVb and IVc from 1989 to 2007, for input into the model. The fleet register contains information on vessel characteristics such as gross registered tonnage, grt, vessel length, and date of registration. We defined the beam trawl fleet based upon the Data Collection Regulation (DCR) of the European Commission (EC, 2006a), and from 2009 under a new regulation, the Data Collection Framework (DCF; EC, 2008c). The DCR and the DCF define the beam trawl fleet according to its use of beam trawl gear for >50% of each trip. The study used information on vessels ≥10 m.

The specific beam-trawl fleet activity or métier is defined as the fisher's tactic at a trip level, which is based on the group of targeted species. Métiers are characterized as an outcome of a trip based on the landing composition, assuming that what is landed in port is a reflection of what was originally targeted. Here, just demersal fish métiers are analysed, by analysing the statistics from beam trawl gear used to target brown shrimp. The landing composition is calculated as a fraction of the total monetary catch, removing differences in catch rates attributable to vessel capacity. Moreover, fractions of the catches are based on economic value, rather than weight, so reflecting the perception that fishers are profit-maximizers, in that uncommon but valuable species being targeted are given more weight in the analysis.

Decommissioning cost data were acquired from Defra for the years where decommissioning grants were offered (1991–2002). Based on data on grant offers and vessel tonnage, Figure 3.5 is the output from a linear model that predicts the premium that would have been offered based on vessel tonnage. The UK addressed MAGP requirements to reduce capacity and effort to meet with specific segment targets. It did not, however, identify overfished stocks or specific fleet segments where capacity needed to be reduced (Cappell *et al.,* 2010).

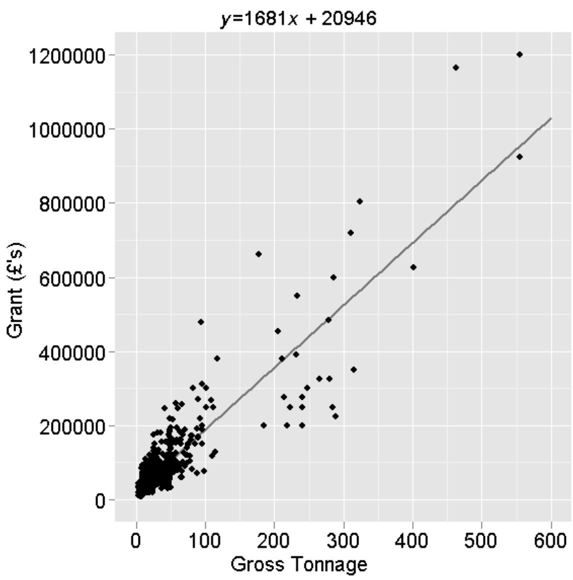


Figure 3.5 North Sea decommissioning grants offered plotted against vessel gross tonnage. Dots are the observations and the line the linear regression.

As an example, the 1993–1998 decommissioning scheme was aimed at vessels more than 10 m long and 10+ years old, which had been at sea for a minimum of 100 d during each of the calendar years 1992 and 1993 (NIAO, 2006). Written applications from vessel owners stating the bid they would require for them to part with their vessels were requested, and these bids were ranked nationally based on the lowest cost per VCU. Over the years as the scheme progressed, the average bids increased as a result of collusion among vessel owners. For instance, the average successful bid in the UK scheme increased progressively from £349 per VCU in 1992 and 1993 to £758 per VCU in 1997 and 1998. The level of UK bids in 1997/8 was significantly in excess of the average EU bid of £650 per VCU (NIAO, 2006). For 1993–1996, the schemes attracted 331, 431, 203, and 255 eligible applications annually, respectively, and the numbers decommissioned were 13 (in 1993), 6 (1994), 7 (1995), 2 (2001), and 4 (2002), a total of 32 beam trawlers. Most of the grant take-up was based on the decisions of the applicants, who were required to satisfy a number of qualification conditions, e.g. based on a minimum number of days at sea and vessel age. The target for this fleet segment was a 15% reduction in fishing capacity. That target was not met, however, possibly indicating that vessel owners had either made a decommissioning bid below market rate or that they valued both licences and track record as higher than the value of decommissioning. Using recent decommissioning data, we assume a vessel would have a grant uptake of 100% if successful. In terms of fuel costs, marine diesel prices excluding value-added tax (VAT) and duty were obtained from the Department of Energy and Climate Change (DECC), and are presented in Figure 3.6. Most noticeable is the steady increase from 2002 compared with the relatively stable prices during the 1990s.

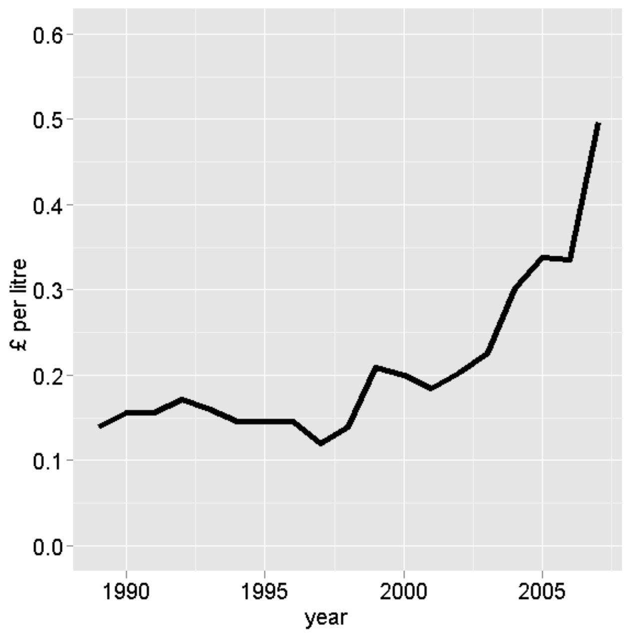


Figure 3.6 Average marine fuel prices (£ per litre, excluding VAT and duty). Source, DECC (UK Department of Energy and Climate Change).

3.2.2 Model description

Over the past few years, considerable attention has been applied to predicting fisher choices, particularly those concerning fishing location, by applying random utility methodology and models (Bockstael and Opaluch, 1983; Eales and Wilen, 1986; Holland and Sutinen, 1999; Wilen *et al.,* 2002; Hutton *et al.,* 2004). The key characteristics of random utility models (RUMs) are that they model discrete decisions and can be described as follows. A decision-maker is faced with making a choice among a number of alternative options, obtaining differing levels of utility from each alternative option, and tending to choose one that maximizes utility. As such, and like other economics-based choice models, utility influences individual choice with a deterministic and stochastic error component.

For the most general form of the conditional logit choice model (McFadden, 1974, 1981), a set of unordered choices is assumed, and this can be written as

$$U_{ij} = \beta_{ij} z_{ij} + \varepsilon_{ij}\ , \tag{3.1}$$

where U is the utility, i the individual, j the choice (such as a fishing trip), z_{ij} are attributes of choice $[x_{ij}\ w_i]$, where x_{ij} are attributes of choice j of individual i, and w_i are attributes of individual i, ε_{ij} is the stochastic error component, which is random, and β_{ij} is a coefficient. It is assumed that values of ε_{ij} are independent across the different choices. This assumption implies a condition known as independence of irrelevant alternatives (IIA), which itself implies that providing other choices or changing the characteristics of a third choice does not affect the relative odds between the two choices considered. The probability of a given choice being made can be estimated from the utility derived. The multinomial logit model (Equation 3.2), used in this study differs from the conditional logit choice Equation (3.1) in that only the characteristics of the individual (w_i), are included:

$$U_{ij} = \beta_j w_i + \varepsilon_{ij}\ . \tag{3.2}$$

The probability that an individual i makes choice j is then

$$\Pr\mathrm{ob}(\gamma_i = j) = \frac{\exp(w_i\beta_j)}{\sum_{j=1}^{J}\exp(w_i\beta_j)}\ , \tag{3.3}$$

where γ_i is an indicator variable (with the same length as vector $\mathbf{J}$) referring to the choice (j) made by individual i. SAS 9.0 software was used in the model estimation (Logistic procedure, SAS institute Inc., 1999).

The key independent response variables (see Table 3.1) in the model include vessel age in years, because it is assumed that older vessels may exit because of higher costs of maintenance and operation, and that newer vessels will enter. Fisher skills, knowledge, and experience are expected to relate to the annual revenues of the target species of the fleet, specifically plaice, anglerfish, and sole. Fishers are assumed to be profit maximizers, and it is expected that fishers with high revenues are more likely to stay in the fishery, whereas those with lower incomes are more likely to exit to seek other opportunities in alternative fisheries or industries. It is important to note that higher revenues for a vessel might not mean greater profit because costs vary considerably between operators. In keeping with the thesis of Mardle *et al.* (2005), we assume that the performance based on the total revenue of the species caught by a vessel in its first year of entry to the fishery meets the expectations of the decision unit, because they expect on entry to perform as well the rest of the fleet. Pradhan and Leung (2004) assume that a vessel's performance in its first year is equivalent to its previous year's performance elsewhere. For the English beam trawl fleet, we cannot assume this, however, because of the different target species and quota limitations elsewhere. For those already in the fishery, we assume that the decision to exit, to stay, or to decommission is based on the previous year's performance. The decision to enter a fishery may also be based on poor performance in another fishery, with the fisher perhaps seeking a better investment opportunity.

The variable decommissioning grant offered is included in the model to evaluate the effects of a fisher's decision to accept a grant to have their vessel removed permanently from the fishery. It is anticipated that a fisher will accept the grant if it is considerably more profitable to do so than to remain in the fishery. The model assumes that vessels have open access to the fishery, i.e. that they purchase a vessel and the licence with the entitlement to fish, but in reality the total number of UK beam trawlers is restricted. Congestion and overcrowding effects are investigated by the inclusion of the number of vessels operating within a given year in the fishery as a variable (Bockstael and Opaluch, 1983; Ward and Sutinen, 1994).

The price per litre of subsidized marine diesel (excluding VAT and duty) was considered a key variable for inclusion, because higher fuel costs could reduce profit directly and lead to a decision to exit the fleet, especially if the value of the catch does not increase to compensate for the higher fuel cost. Alternatively, if fuel costs decrease, then the expectation would be that more vessels would enter the fishery. The fuel cost variable was lagged (i.e. $t + 1$, where t is year), because it was assumed that fishers would not enter or exit the fishery immediately in response to a change in fuel price, but rather as a strategic decision based on the average annual costs in the previous season.

Plaice was considered to be the main target species of the English fleet, so the spawning-stock biomass (SSB) of plaice was included as a variable in the model. As stock assessments use the previous year's catch to predict the next year's quota, this variable was lagged ($t + 1$). It is assumed that a fisher would be likely to leave the fishery if past SSB was low. Conversely, if stock levels increase, then the assumption is that more vessels will enter the fishery.

Overall vessel length (m) was included as a variable in order to determine whether being within any particular vessel-length group influenced a fishers decision to enter, stay, or exit. Vessel size is correlated with capital invested and may affect a fisher's decision. Smaller vessels have fewer decisions on where to fish, because they are restricted primarily to inshore fishing grounds, but should have lower fuel costs. By comparison, medium-sized or large vessels can operate farther offshore for longer, so have more variable fishing opportunities. Other variables initially included in the models were removed because the observation was made that they were not significant. These included cod (*Gadus morhua*) revenue, the SSB of sole, total revenue, VCU, grt, engine power (kW), turbot (*Psetta maxima*) revenue, and the monetary sum of other landings (excluding plaice, sole, turbot, anglerfish, and cod).

Table 3.1 The explanatory variables used in the model.

Variable	Description
1	Vessel age (years)
2	Annual individual plaice revenue (£)
3	Individual decommissioning grant offered (£)
4	Annual fleet size in numbers
5	Lagged spawning-stock biomass (SSB) of plaice (t)
6	Annual individual anglerfish revenue (£)
7	Lagged average annual fuel price (£)
8	Annual individual sole revenue (£)
9	Individual vessel length (m)

3.3 Results

The results for the multinomial logit model (unordered) are given in Tables 3.2 and 3.3. The coefficients are interpretable in terms of the direction of the influence of a variable on the utility, and the probability of entering, exiting, staying, or decommissioning vs. staying (Table 3.3). Only variables with significance levels of $p < 0.05$ were included in the models with respect to the Type 3 analyses of effects (Table 3.2), which shows the overall significance of each variable retained in the final model, using Wald Chi-squared statistics. The model is highly significant, and has a likelihood ratio χ^2, 27 d.f., of 393.8138, $p < 0.0001$, and a value of r^2 of 0.22, where $n = 1595$.

Table 3.2. Type 3 analysis of effects, showing the overall significance of each variable retained in the final model, using Wald χ^2 statistics and given that the other variables are in the model.

Variable	d.f.	Wald χ^2	Pr > χ^2
1	3	21.2413	<0.0001
2	3	44.991	<0.0001
3	3	20.1455	0.0002
4	3	18.7338	0.0003
5	3	10.2478	0.0166
6	3	9.6155	0.0221
7	3	10.2033	0.0169
8	3	8.4166	0.0381
9	3	15.5148	0.0014

Table 3.3 Parameter estimates from the multinomial logit model.

Variable	Entry	Exit	Decommission
Intercept	–1.33**	+0.06	–7.31***
Vessel age in years	–1.70E–02**	–5.78E–03	+6.65E–02***
Annual individual plaice revenue (£)	–4.49E–06***	–4.79E–06***	+5.72E–06**
Individual decommissioning grant offered (£)	–9.19E–07*	–2.74E–07	+6.87E–06***
Annual fleet size in numbers	+4.04E–03**	–1.82E–03	+2.97E–02***
Lagged SSB of plaice (t)	+3.14E–06**	+2.85E–07	–6.46E–06
Annual individual anglerfish revenue (£)	–1.00E–05**	–1.00E–05**	–3.00E–05
Lagged average annual fuel price (£)	+4.31E–02**	+1.10E–02	+1.88E–01**
Annual individual sole revenue (£)	–1.21E–06	+5.64E–06	–5.00E–05**
Individual vessel length (m)	–1.88E–02	–2.57E–02**	–1.24E–01***

* Statistical significance at 10% level.
** Statistical significance at 5% level.
*** Statistical significance at 1% level.

The results for the variable vessel age indicate, as expected, that older vessels are more likely to leave the fishery. Essentially, older vessels are replaced by newer ones, resulting in an increase in the efficiency of the fleet. Hutton *et al.* (2008) considered the implications of older vessels leaving the fleet and the resulting changes in technical efficiency of the fleet remaining. Our results suggest that the bigger the fleet, the more vessels that enter, and the smaller the fleet, the greater the odds of vessels exiting. Historically, the trends reflect an expansion of the fleet as fishing opportunities increase (large catches of plaice and high revenues), resulting in more vessels entering the fishery. Fishers tend to follow others into a profitable fishery, so enter an ever-growing fleet. In recent years, the few newer and larger vessels remained active, acquiring newly available quota from vessels that exited the fleet. The coefficient for decommissioning suggests that the odds on a younger vessel in the fishery taking up a decommissioning scheme are low. Alternatively, older vessels were more likely to take up a decommissioning offer. Similarly significant was fleet size, because the bigger the fleet, the greater the odds of decommissioning. This result is intuitive, because the fleet size when it was at its largest coincided with the decommissioning schemes under MAGPIII. The stock status of plaice also had an important influence; the odds on entering the fishery increased when plaice stock levels were high. As anticipated, the results for plaice revenue indicated that vessels with lower revenues would have greater odds on exiting the fleet. The implication of vessels with low revenues (or low vpue) departing the fisheries is also likely to have an impact on the overall efficiency of the fleet. A highly significant negative coefficient on the variable plaice revenue for entry is not intuitive, but the explanation for this may be that cost factors such as high fuel costs dominated during this period. Decommissioning programmes during the period of study did show signs of enticing beam trawlers to decommission.

The positive significant coefficient for fuel prices for vessels entering suggested that the vessels would enter at lower costs of subsidized fuel. However, the choice to exit was not significant, possibly because of the relatively stable fuel prices throughout the 1990s. The significant coefficient for decommissioning suggests that when fuel prices were at their highest and a decommissioning scheme was available, fishers were likely to accept a grant to exit the fishery.

All the estimated parameters for vessel length were significant. The options all possess negative coefficients and suggest that smaller vessels were more likely to exit, enter, and decommission from the fleet. With lower capital cost outlays, smaller vessels tend to be more mobile in their movement in and out of fisheries. The effect of vessel size is also related to the variable decommissioning grant, which understandably has insignificant coefficients for exit, and a highly significant small coefficient for the decision to decommission. Fishers on the large most-modern vessels would have to be offered a good financial incentive to leave the fishery, fitting in with the observation that smaller vessels accepted the scheme offered. Of interest were the significant parameter estimates for anglerfish, showing similar trends for exit and entry as the revenue estimates for plaice. However, sole revenue provided a different set of trends, notably the insignificant variables for entry and exit, which suggests that it was not of great importance to fisher decision-making in terms of whether to enter or exit the fishery.

3.4 Discussion

A decision to enter or exit a fishery depends on the expected benefits, which include future revenue, the stock status of the main target species, and crowding effects (the number of vessels in the fleet). Operators also make decisions on the basis of age of the vessel. Here, the results indicate this to be the case for English North Sea beam trawlers. Management measures (TACs and effort limitations) which impact on revenues also affect future stock levels, because a TAC is based on the proportion of mature fish that can be harvested from a stock. Therefore, if SSB falls by a set amount, this will be transferred to revenue and impact the proportion of vessels that enter or exit the fleet.

In order to illustrate the marginal effects of each significant explanatory variable, the mean model coefficients from Table 3.3 were kept constant, and the predicted probability of exit, entry, and decommission were computed over a range for each explanatory variable. The outcomes of the simulations are shown in Figure 3.7.

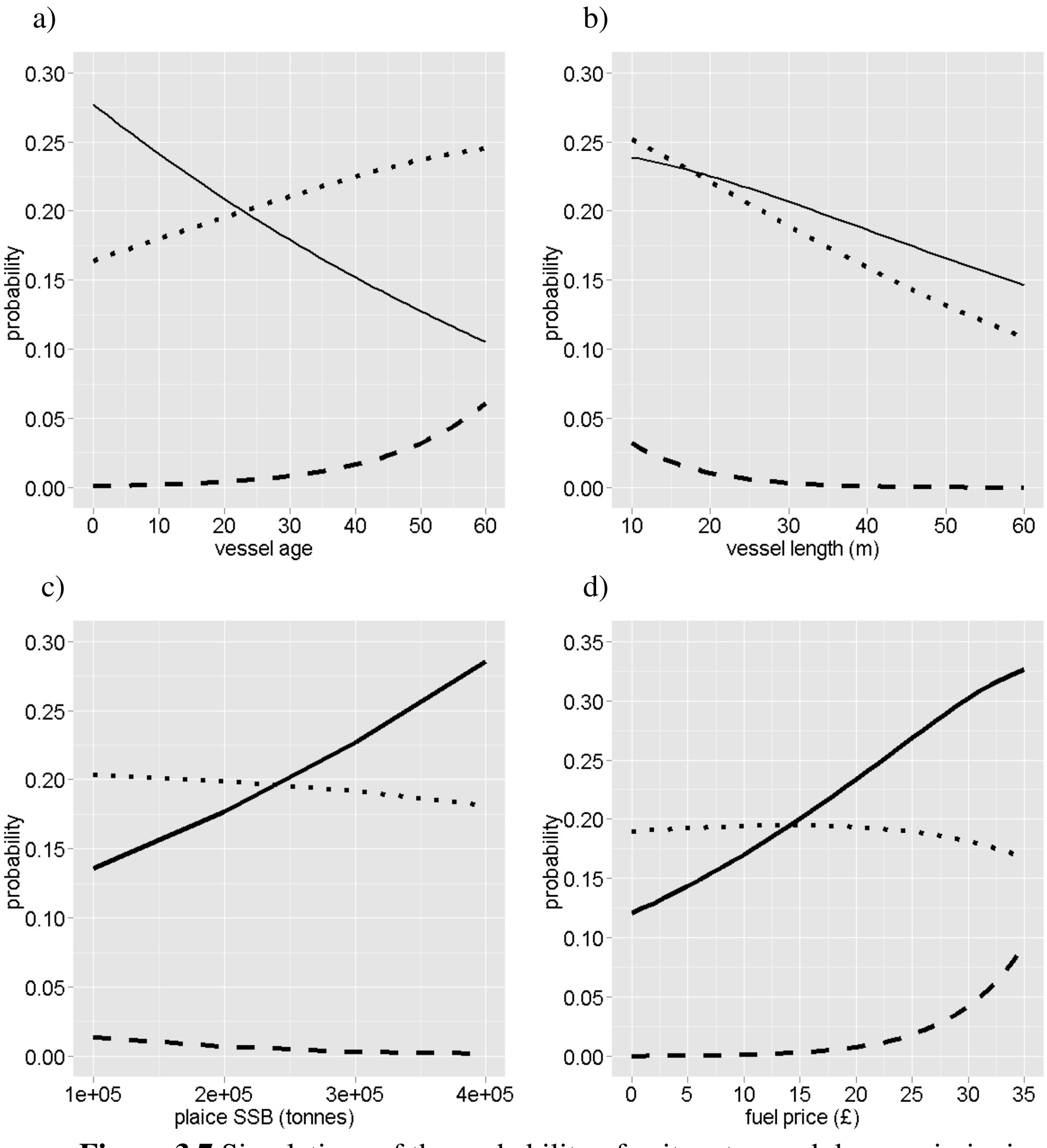


Figure 3.7 Simulations of the probability of exit, entry, and decommissioning decisions, i.e. 1 minus each probability equals the probability of staying in the fishery The solid lines represent the probability of entry vs. stay, the dotted lines the probability of exit vs. stay, and the dashed lines the probability of decommissioning vs. stay. (a) Vessel age, (b) vessel length, (c) plaice SSB, (d) fuel price, (e) decommissioning grant, (f) number of vessels, (g) plaice revenue, (h) anglerfish revenue.

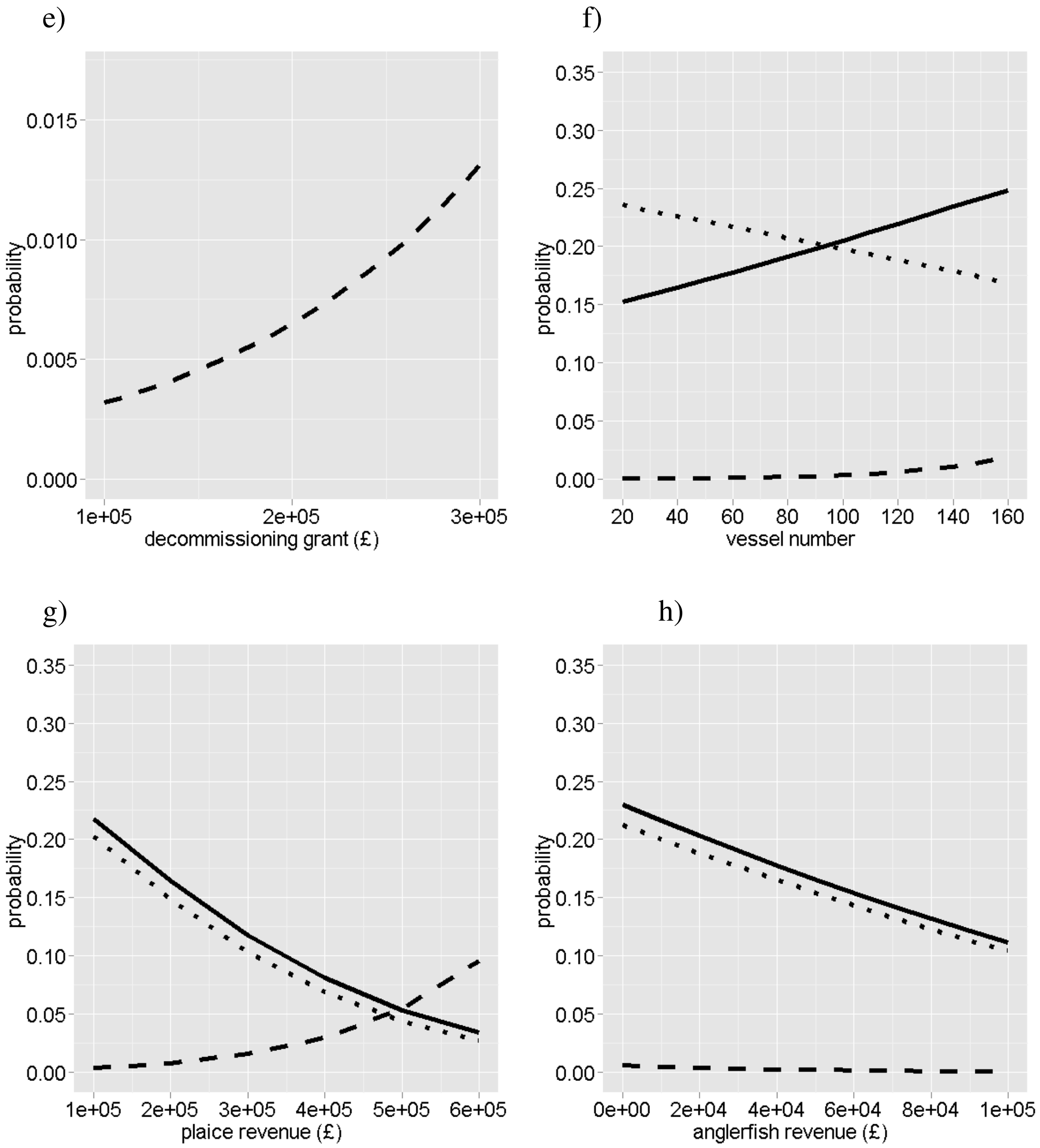


Figure 3.7 (continued).

In general the outcomes are as expected, and can be simply summarized as follows. Figure 3.7a supports the notion that an older vessel is less likely to enter a fishery than a newer vessel; decommissioning and exit probabilities increase with age. These predictions are plausible given the decommissioning schemes under the MAGP programmes of the 1990s. Figure 3.7b shows that smaller vessels have a greater probability of taking up a decommissioning scheme, consistent with the general patterns witnessed under the MAGPs.

The decision by the owners of such vessels to decommission may also suggest that the owners of smaller vessels find it easier to part with a vessel than stakeholders investing in bigger boats, where a group decision is required within a firm, and where financial considerations are more important. Figure 3.7c shows that with an increase in stock size of mature fish (SSB), it is more attractive for vessels to enter; on the other hand, the probability of exit is reasonably constant. However, it does seem that vessels are less likely to exit at lower levels of SSB, suggesting that the fishery is profitable at reduced plaice SSB and that the fleet switches to targeting other demersal species such as sole, anglerfish, cod, or turbot, or mixed combinations of these species. Figure 3.7d shows that fuel prices are important; in 2001, they accounted for 70% of running costs in the beam trawl fishery (Mardle *et al.,* 2005). At low and increasing costs, a vessel is more likely to enter until it appears to level off at £0.35 per litre of fuel.

The simulation also shows that, with an increase in fuel price, fishers are more likely to decommission their vessels. These are possibly the result of vessels being older and inefficient, and hence more costly to run. This highlights the importance of fuel costs and the potential effects of fuel subsidies. Fuel subsidies have a direct effect on fishing effort (Sumaila *et al.,* 2006). They are controversial, because they encourage wasteful, uneconomic fishing practices in already overcapitalized fisheries, and they maintain fishing effort even when stock levels decline. Figure 3.7e shows that the beam trawl flatfish fishery is of great value to fishers, and that a grant of £300 000 would only entice 1.3% of beam trawl fishers to decommission their vessel. The policy implications are that, given the overcapacity in the North Sea and the declining stocks, it is costly for a decommissioning scheme to become the most effective method of reducing capacity. Decommissioning is said to be only a short-term incentive for solving the underlying overcapacity issue, although some advocate that it is the only solution; however, it cannot solve the inherent incentive problem of over-investment and effort creep (Weninger and McConnell, 2000). In the longer term, therefore, one would expect a similar capacity/effort imbalance after a decommissioning scheme. Alternatively, if market forces dominate under a tradable quota system, the fleet will rationalize within a system of ITQs, where the race for fish is removed, allowing individuals to catch a set quantity and allowing investment and production strategies to be internally driven by market forces. As quotas decrease and fuel prices rise, fishing vessels may be forced to tie up and be sold, or to exit the fishery to operate in areas away from the North Sea (for most of the period of the study, beam trawl licences permitted fishing in other areas and quotas were not gear-specific). Redistributing overcapacity elsewhere or tying up, however, could cause social and economic problems to coastal communities that rely on the fishery for employment. In fact, the quasi-ITQ system in the UK (and the ITQ system in the Netherlands) has resulted in just that, a smaller English North Sea beam trawl fleet (now operated mainly by Dutch skippers) with traditional North Sea English fishing ports in decline.

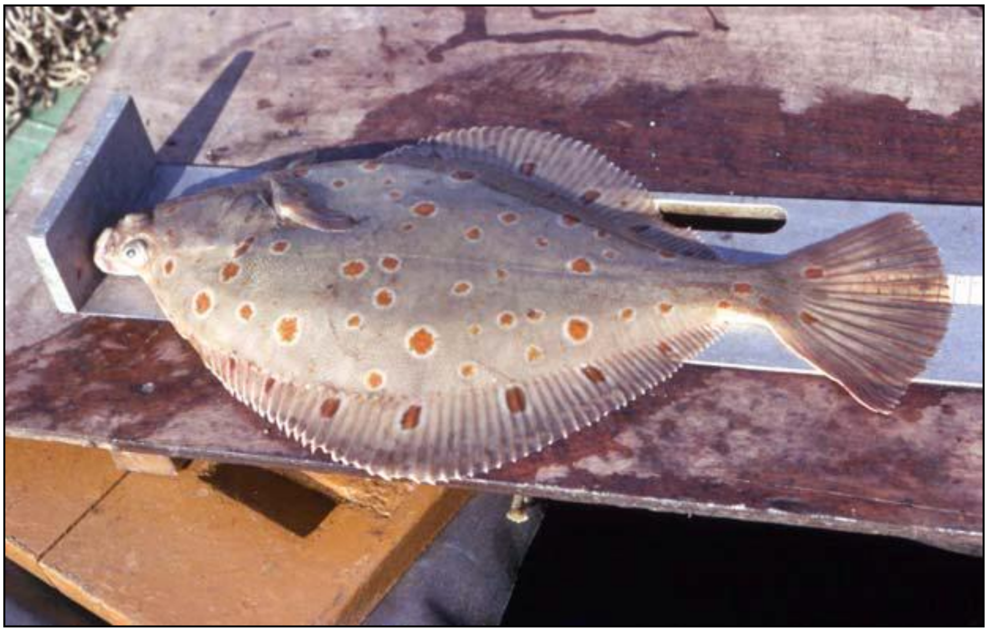
The results in Figure 3.7f demonstrate that the probability of entering the fishery increases with fleet size. However, Dasgupta and Heal (1979) state that ever-increasing numbers of competing fishers lead to externalities between them, i.e. the more one catches, the less is available to others, so that each operator believes that none of its competitors will adhere to a future conservation policy and in turn sees no benefit to pursue it personally. As fleet size expands, average landings and revenues per vessel decline along with catch per unit effort (cpue), the costs of fishing effort increase, and resource rents dissipate (Ward and Sutinen, 1994). Anecdotal information suggests that the fishery addressed here did just that, in that it expanded as plaice catch rates were high, followed by success that was short-lived as catch rates declined and fuel costs rose, and with the demise of the fishery reflected by dissipating profits.

Figures 3.7g and 3.7h show similar trends for both exit and entry, in that the less the revenue of plaice and anglerfish, the greater the probability of entry and exit (the results for sole are not shown). As noted above, cost factors (e.g. fuel costs) may have dominated during this period. In addition, plaice catches and revenue were relatively stable compared with such other covariates as fuel costs, which could adversely dominate in their explanatory power. The implications of a smaller fleet on fish stocks are yet to be evaluated. One method to explore the effects of alternative fleet management policies on fish stocks is management strategy evaluation (MSE; De Oliveira *et al.,* 2008). Under an MSE approach, the objective is to evaluate the management consequences of a strategy under alternative assumptions about stock dynamics, i.e. its robustness to uncertainty. A key element is to identify the relative impact of particular assumptions about the resource (e.g. stock–recruitment relationship, natural mortality) or fleet dynamics (e.g. the implementation of management regulations). The overall objective of fishery management is balancing the short- and long-term socio-economic needs of stakeholders while maintaining a healthy stock and ultimately rebuilding fisheries.

To conclude, we have discussed the implementation of a discrete choice model (specified as a RUM) in an attempt to explore and better understand English beam trawl fisher long-term investment behaviour. The results confirm the notion that vessel age, vessel length, stock status (plaice SSB), fuel cost, the availability of decommissioning grants, fleet size, and the revenues from target species are significant factors in determining fisher decision-making. The low model r^2 of 0.22 suggests that other factors not incorporated in the model play a role, e.g. the real economic viability of each vessel (knowledge of which is limited by the limited availability of cost-structure data), their ownership, and the investment portfolio of firms that own single and multiple vessels, as well as factors such as skipper skill and age, and/or the availability of a skilled crew. The model predictions were similar to the actual choices apart from the decision to decommission, possibly because relatively few beam trawlers took up the decommissioning schemes offered during the period investigated. UK decommissioning schemes were *ad hoc* in nature and spread across many sectors, not just the beam trawl fleet. Rather, it was a case where some owners took advantage of limited decommissioning grants when they were worst off financially (with low catch rates of plaice and high fuel costs), whereas others valued future catches, the value of the licence, and their capital investment higher than the value of decommissioning.

Future studies should, if feasible, include an investigation of other externalities than subsidies on fuel and decommissioning grants. Subsidies could include tax relief in the form of income support and unemployment insurance, capital support such as for vessel modernization (a new engine refit), minimum price, and processing and marketing subsidies. Such financial instruments could help in attaining profitability and influence future investment decisions by a fisher. In addition, it would be of interest to know whether the skippers who decommissioned reinvested in newer vessels, encouraged by the profits of the fishery. Regulations, policy and alternative fishery performance, pre-enter and post-exit revenues, and costs, if available, would further enrich such analyses. Overall, our analysis has provided greater insight into the use of econometric RUMs in interpreting fisher behaviour.

Chapter 4. Dynamic prediction of effort reallocation in mixed fisheries[3]



European plaice *Pleuronectes platessa* (Source: CEFAS).

[3] Tidd, A. N., Hutton, T., Kell, L. T., and Blanchard, J. L, (2012). Dynamic prediction of effort re-allocation in mixed fisheries. Fisheries Research, 125-126: 243–253.

Abstract

A discrete choice model is applied to determine how fishing effort is allocated spatially and temporally by the English and Welsh North Sea beam trawl fleet. Individual vessels can fish in five distinct areas, and the utility of fishing in an area depends on expected revenue measured as previous success (value per unit effort) and experience (past fishing effort allocation), as well as perceived costs (measured as distance to landing port weighted by fuel price). The model predicts fisher location choice, and the predictions are evaluated using iterative partial cross validation by fitting the model over a series of separate time-periods (nine separate time-periods). Results show the relative importance of the different drivers that change over time. They indicate that there are three main drivers throughout the study, past annual effort, past monthly effort in the year of fishing, and fuel price, largely reflecting the fact that previous practices where success was gained are learned (i.e. experience) and become habitual, and that seasonal variations also dominate behaviour in terms of the strong monthly trends and variable costs. In order to provide an indication of the model's predictive capabilities, a simulated closure of one of the study areas was undertaken (an area that mapped reasonably well with the North Sea cod 2001 partial closure of the North Sea for 10 weeks of that year). The predicted reallocation of effort was compared against realized/observed reallocation of effort, and there was good correlation at the trip level, with a maximum 10% misallocation of predicted effort for that year.

4.1 Introduction

It is becoming increasingly evident that fisheries management is not solely a biological issue. Fisheries science is an interdisciplinary field, and combining social, economic, and ecological information has proven to be increasingly important in achieving sustainable fisheries management (Mumford *et al.*, 2009). Of increasing importance to fisheries science and management is the ability to anticipate fisher behaviour in response to management regulation, in order to reduce implementation error, i.e. where the effects of management differ from that intended. An example of implementation error is where fishing effort is redistributed following a spatial closure to protect a stock (or cohort) in a way that was not anticipated by management.

Many factors influence a fisher's decision where and when to fish, including fish distribution, fuel price, regulations, their habits and experience, previous catch rates, market prices, and the proximity to landing ports. These factors can lead to differences in observed individual fisher behaviour and the way a group of fishers (a fleet) allocate their effort in time and space. Several studies have looked at behavioural aspects of the way fishers spatially allocate their effort (Rijnsdorp *et al.*, 2000; Hilborn *et al.*, 2005; Smith *et al.*, 2009). An important element influencing fisher behaviour is stock density, because fishers tend to have prior knowledge (Begossi, 2001) of resource distribution and habitat (Hilborn and Ledbetter, 1979; Gillis *et al.*, 1993; Pet-Soede *et al.*, 2001). Catch rates are related to stock density and will have a large impact on fisher behaviour (Eales and Wilen, 1986; Marchal *et al.*, 2006). This means that fishers will gravitate towards areas where catch rate is greatest, and gravity models have been specified and applied to model fishing vessel spatial distribution (e.g. Walters and Bonfil, 1999). Economic factors and management measures in the form of technical measures (size restrictions or gear restrictions; Bene and Tewfik, 2001), marine protected areas (MPAs), and spatial closures may also force fishers to search for new fishing grounds (Hutton *et al.*, 2004).

Over the past few years, much attention has been paid to predicting fisher location choice by applying random utility methodology and discrete choice models (Anderson *et al.*, 2010). Predicting fisher behaviour using discrete choice models has increased in popularity with the increasing availability of appropriate data (vessel-by-vessel trip data), because such models offer an opportunity to study individual behaviour at finer resolutions of time and space than other techniques (Coglan *et al.*, 2004; Hutton *et al.*, 2004). These models can be applied to theoretical policy scenarios, which can also be simulated. The key characteristics of discrete choice models or random utility models (RUMs) are that they model discrete decisions, and the assumption of homogeneity among individuals does not need to hold. As with other economics-based choice models, utility drives individual choice with a deterministic component and a stochastic error component (hence the name “random” utility model). Prior to implementation in fisheries behaviour models, discrete choice models were used in the travel industry to analyse the behaviour of consumers of transportation services and facilities (McFadden, 1974; Ben-Akiva and Lerman, 1985).

The behaviour of fishers can be studied in the short term (their tactics), for example on a trip-by-trip basis in terms of decisions where to fish and which species to target, or the long term (their strategies), i.e. choices made year by year where the availability of decommissioning grants, stock status, catch quotas, investment, and other key factors play a critical role in the decision of a fisher to invest in the fishing operation (Chapter 3). Models prior to the application of discrete choice models assumed the ocean to be a homogenous space in which fish are distributed uniformly and fishing locations are identical (e.g. Holland and Brazee, 1996; Smith and Wilen, 2003). Sanchirico and Wilen (1999) modelled behavioural dynamics, including both spatial and temporal aspects, under conditions of open access. The results of their analysis suggested that fishing effort across a system of interconnected spatial patches is driven by the bio-economic conditions in each patch, and the biological dispersal rates between patches. In patches where costs are high or the catchability and prices low (mix of low price species and/or cohorts), effort is driven away, and as it relocates, it affects the distribution and density of stocks (i.e. the local density and the potential for dispersal to nearest-neighbour patches) of other patches directly and indirectly. Incorporating economic variables (such as revenue and travel costs) into decision-maker behaviour is therefore important when analysing a resource that is distributed heterogeneously in space.

In this study, we investigate whether tactical behaviour by fishers is influenced by expected revenues, habitual seasonal fishing patterns, effort fluctuations, and changes in fuel costs, and whether there are dynamic changes in the relative importance of these drivers through time. Focus is on the English and Welsh North Sea beam trawl fleet, where there have been changes in both ownership and spatial management; as such, this study provides an opportunity to investigate the dynamics and drivers of fisher behaviour. Also of interest to this study is the fact that, during 2001, the European Commission implemented a temporary closure or MPA in the North Sea between mid-February and the end of April, to conserve spawning of North Sea cod (EC, 2001). As a regulatory management measure that impacted fishing effort, the 2001 closure of the North Sea covered most of Roundfish area 7, which beam trawlers frequent, and the remainder of which included a plaice box preventing trawlers >300 hp from entering (Figure 4.1a). This allowed us to evaluate the predictive power of the model and analysis, and among other factors the response of the fleet to a management measure. An earlier study also applied a discrete choice model to the same fleet using individual fishing trip data over the years 1999–2000. Previous knowledge or experience of fishing grounds (in 1999) was found to have a bearing on the decision to fish in a given area in 2000, and this information was then used to construct a simple effort redistribution model to simulate the implications of the 2001 closure (Hutton *et al.*, 2004). Although that study investigated detailed spatial location choice, there are limitations to such work for considering temporal changes in fisher behaviour. This is because of the short time-period of data and the type of discrete choice model used. Hutton *et al.* (2004) used a conditional logit model, a model often criticized when used for spatial policy analysis because of the Independence of Irrelevant Alternatives (IIA) it imposes, i.e. choices are assumed to be independent, and a change in one choice would not affect the relative choice set, which could have serious implications if used for a spatial policy analysis (Wilen *et al.*, 2002).

Here, focus is on the dynamic changes in tactical behaviour over a 12-year period. We introduce the use of a mixed model (relaxing the IIA assumption) and extend the set of explanatory variables investigated to a wider range of potential drivers (such as distance to landing port and separation of catch into their targeted components, plaice and sole). To understand better the drivers and dynamics of fisher location choice over space and time, we fit discrete choice models over different periods and investigate the effects of the various explanatory variables (which are proxies of expected revenue and costs perceived by fishers from past experience on monthly and annual time-scales). We then predict fisher location choice over separate periods to evaluate the model predictions, along with the versatility and robustness to potential changes in tactics. Finally, we develop a framework for investigating fisher location choice that can be used to reduce potential implementation error and scientific uncertainty and allow for the management system to be adjusted or adapted to what is learned.

4.2 Material and methods

4.2.1 English North Sea beam trawl fleet

English beam trawl vessels in the North Sea have traditionally caught mostly plaice in a directed beam trawl fishery using 120 mm mesh north of 56°N, and a mixture of plaice and sole using 80 mm beam trawls in the southern North Sea. In 2003, international landings of North Sea plaice amounted to 66 502 t, compared with a peak of 170 000 t in 1989. Some 42% of the total plaice international landings were reported by vessels from the Netherlands, the UK accounted for 21%, Denmark for 21%, and Belgium, Germany, France, and other countries the balance of 16% (ICES, 2007). In the English fishery, the high value of sole makes it one of the most important species targeted by inshore vessels using trawls and fixed nets. The fishery is mainly conducted from March to October, but sole are also taken as a target species by offshore beam trawlers, otter trawlers, and gillnetters. The English North Sea beam trawl fleet operated mainly out of east coast English ports until 2003, typically spending an average of 250 days at sea in trips lasting about six days (Hutton *et al.*, 2004) (see Figure 4.1b for effort distribution of the beam trawl fleet, 1997–2007).

Towards the end of 2002, the main English east coast beam trawl company ceased fishing because it could not fish profitably. This was largely due to a fuel crisis from late 2000, with high and rising fuel prices over several years along with declining catch rates of large plaice. That company and other operators claimed that they could not catch the fish for which they had quota entitlement, that prices for fish were poor, and that the fuel costs incurred by vessels having to travel long distances to catch the fish were too high (Hansard, 2002). Subsequently, the fishing vessels were sold to operators in the Netherlands, but they still maintained the English flag and quota allocations. Some vessels were leased initially in 2001, with formal transfer of ownership depending on vessel taking place from 2002 to 2005.

English beam trawl fishers generally choose to target both plaice and sole, but in recent years because of the shrinking fleet size and the transfer of ownership to fishers from the Netherlands, skippers generally targeted sole because of its high commercial value and short distance from port in the southern North Sea, generally in Roundfish area 6 (Figure 4.1a).

a)

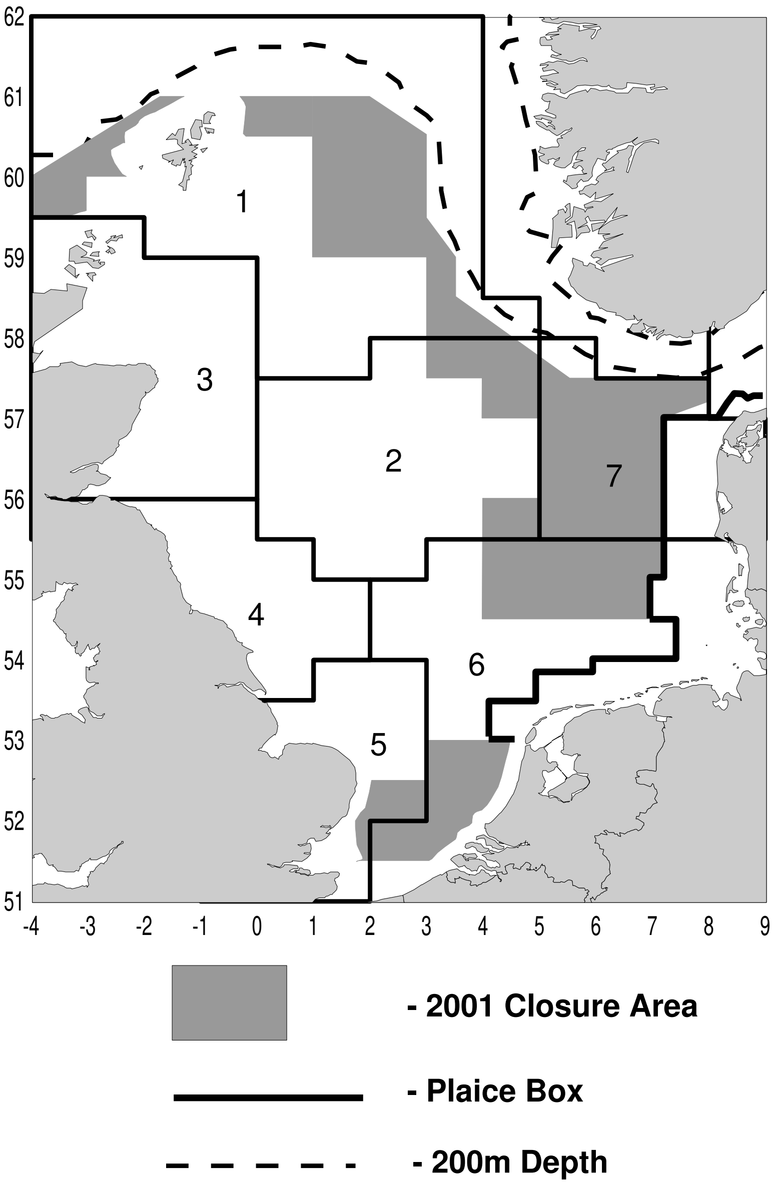


Figure 4.1 The study area showing (a) Roundfish areas, including the 2001 closure areas and the plaice box and (b) Total hours fished by the ≥10 m English beam trawl fleet operating in the study area.

b)

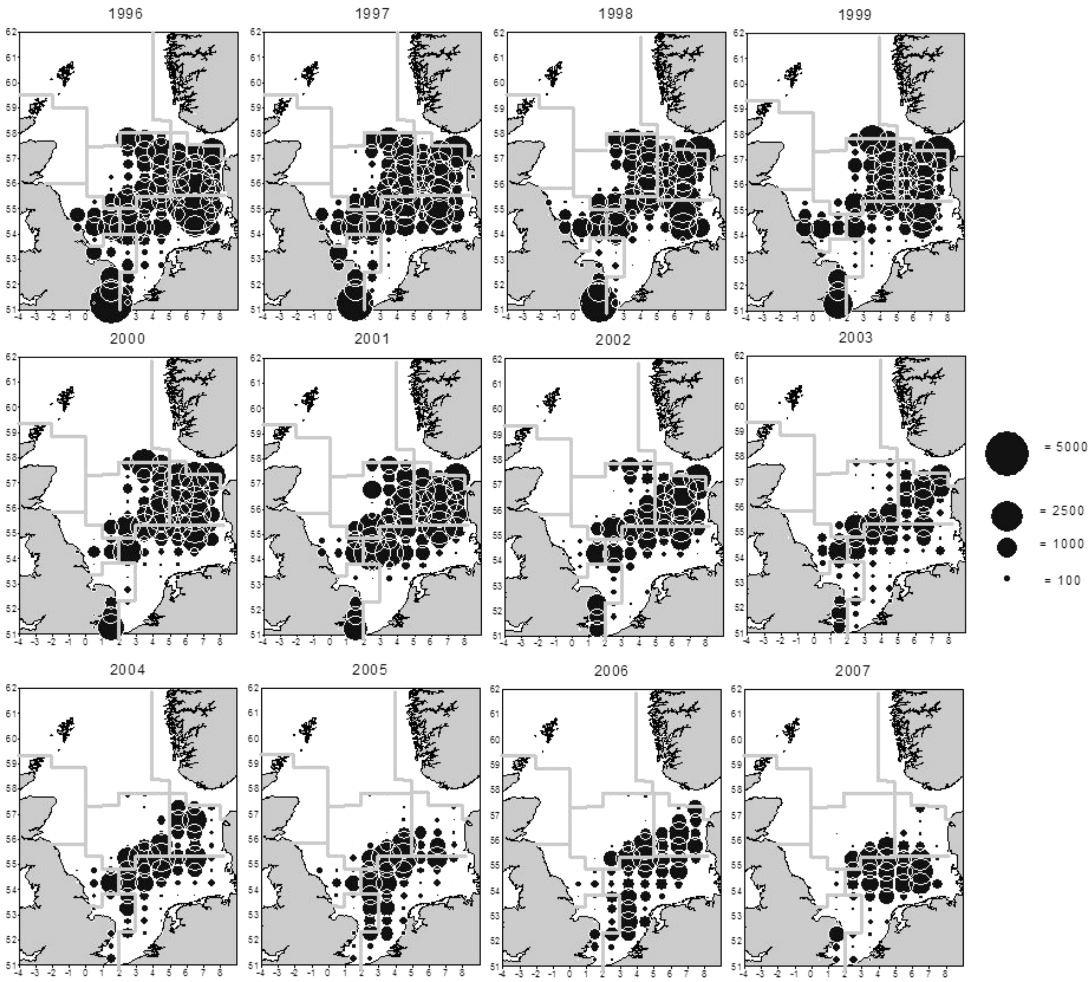


Figure 4.1 (continued) The study area showing (a) Roundfish areas, including the 2001 closure areas and the plaice box and (b) Total hours fished by the ≥10 m English beam trawl fleet operating in the study area.

4.2.2 Data

The areas in the study were chosen based on the International Bottom Trawl Survey (IBTS) and in particular the Netherlands beam trawl survey (BTS) which stratifies its sampling of sole and plaice to Roundfish areas (Figure 4.1a; ICES, 2009a). The fishery-independent survey results are used in the ICES North Sea demersal working group (WGNSSK) for assessing sole and plaice. These Roundfish areas also represent the main fishing grounds at a large spatial scale, and are independent, i.e. they are discrete choice decision units. Individual trip data for the commercial beam trawlers were collated for the years 1996–2007.

Roundfish areas 1 and 3 (see Figure 4.1a) were excluded from the study because English beam trawlers generally do not fish there. The number of trips decreased annually during the study period (Figure 4.2). The data collected for each vessel included species landed, hours fished, landed weight per ICES statistical rectangle (kg), month of fishing, year of fishing, and total value of the catch by species by vessel and trip. Within the EU, it is currently only a requirement for vessels >10 m long to submit logbooks, but the database also contains a subset of catch from <10 m vessels that historically reported their catches by means of logbooks.

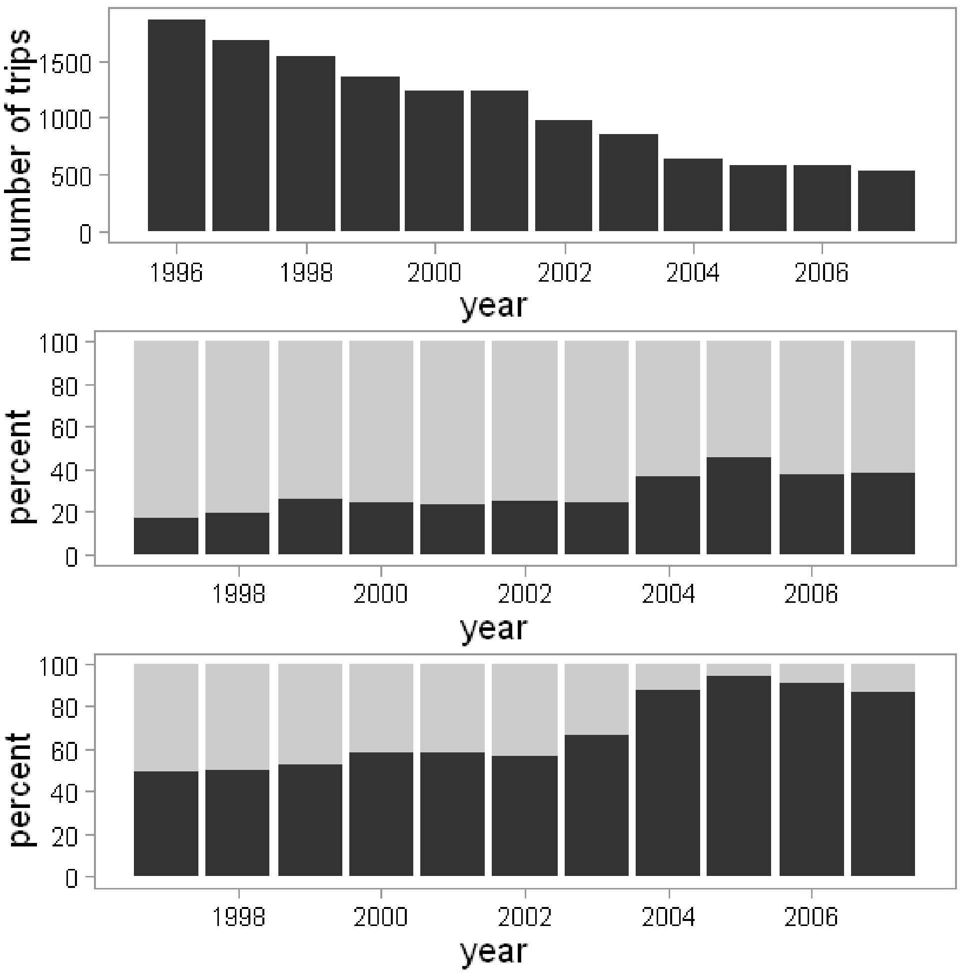


Figure 4.2 (top) Number of trips by registered English and Welsh beam trawlers during the study period. (middle) Approximate representation of the percentages of registered owned English and Welsh beam trawlers, black bars indicating foreign (excluding UK and Ireland) ownership. (bottom) Percentage of trips by English and Welsh beam trawlers to English or foreign (excluding UK and Ireland) landing ports, with black bars indicating foreign landing ports.

The methodology for the definition of fleets is based on the European Commission's data collection regulation (DCR; EC 2000). We use a method developed independently (see EC, 2006a), preceding the present data collection framework (DCF; EC, 2008c), which defines the beam trawl fleet based on its use of a beam trawl for >50% of a fishing trip.

The fleet activity or métier is determined by the fisher's tactic at a trip level, which is defined on the basis of the mix of target species. In other words, métiers are characterized on the basis of the outcome of a trip defined by the composition of the landings. That composition is calculated as a proportion of the total value of the catch, removing the differences in catch rates attributable to vessel capacity. Moreover, the proportions of the catches are based on economic value rather than weight, reflecting the notion that fishers are profit maximizers, so valuable species being targeted receive more weight in the analysis. In this study, the beam trawl métier that primarily targeted crustaceans (brown shrimp) was omitted, and a single demersal métier was defined (beam trawl demersal) and used in the analysis. This fleet targets the main flatfish stocks (plaice and sole) in the North Sea.

We anticipate there would have been changes in tactics attributable to changes in the availability of fish, prices, fuel costs, and whether skippers were re-employed. Unfortunately, there is no information available on ownership or personal information about the owners (and/or skippers), but just limited information on vessels registered to the UK and whether they record their landings under the UK flag. However, we do have detailed information on port landed, where they fish, and traditional landings data such as species landed, effort, and price paid. Anecdotally, a series of surveys on technological change were conducted on this fleet, and the results showed first-hand the switch in port (from a UK port to a port in the Netherlands) that occurred over the period of this study. The switch in port nationality for the large beam trawl vessels was characterized by a change in vessel ownership from UK-owned and operated to Netherlands-owned and operated (with lease agreements at first), and a change in the nationality of the skipper and the crew (Hutton, pers, comm.). Spatial location choice is discrete instead of continuous because it can be represented as 0–1 decision in the context of a choice model. The choices are planned *a priori* and influenced by seasonality, tradition, habit, belief, demand, fish habitat, and the spatial distribution of the target stocks.

4.2.3 Model description

Fishers gain economic benefit, i.e. a utility μ, from fishing, and have to make a choice of fishing location each trip based upon the potential catch rates (i.e. revenue), the cost of travelling to a location, and other preferences for a particular location (knowledge of fishing ground and weather). These will differ between locations, so the total utility μ_{njt} of fisher n for site j in trip t is

$$\mu_{njt} = \beta_n x_{njt} + \epsilon_{njt} \quad , \tag{4.1}$$

where $\beta_n x_{njt}$ are the vectors of coefficients and explanatory variables providing information on the known or observed component, and ϵ_{njt} is the random or unobservable component of each vessel's utility and, for simplicity, β_n is assumed to be homogenous among individual fishers (such that the vector β has the same length as the number of explanatory variables x). However, the conditional logit has often been criticized because it imposes an independence of irrelevant alternatives (IIA) property (Ben-Akiva and Lerman, 1985), especially for spatial models (Wilen *et al.*, 2002). The IIA property assumes that the random error component ϵ_{njt} is independent across choices for each decision-maker, and the unmeasured attributes of choice are assumed to be uncorrelated. This implies that a change in the choice set would not affect the relative probabilities. The probability ratio of any two choices depends on the attribute vectors of the respective choices, despite any single probability depending on the attributes of all choices.

The RUM used in this study is a mixed logit model (also known as a random parameters logit) (Hensher and Greene, 2003; Train, 2003) which relaxes the IIA property because it assumes heterogeneity among alternatives at the population level. It differs from the conditional logit (McFadden, 1974) in that β_n varies in a population across individuals. Instead of estimating β_n for all individuals, the mean $\bar{\beta}$ plus its standard deviation σ_n are used to represent the preference distribution in the population of fishers (Train, 1998). The mixed logit choice model takes the form of Equation (4.2) below, where $\bar{\beta}x_{njt}$ represents the observed utility and $\sigma_n x_{njt}$ the unobserved utility. One part of the error distribution (unobserved), therefore, is correlated over alternatives, and the other part, ϵ_{njt}, is independent and identically distributed (iid) over alternatives and individuals (McFadden, 1981; Maddala, 1983), and is written as

$$\mu_{njt} = \bar{\beta}x_{njt} + \sigma_n x_{njt} + \epsilon_{njt} \quad . \tag{4.2}$$

Within the mixed logit framework, β_n was assumed to follow a normal distribution, and for a given value of n (for simplicity disregarding *t*), the conditional probability of choice *j* across all other choices *k* = 1 to *J* is estimated by drawing random values β by simulation using

$$P_n(j) = \frac{\exp(\beta x_{nj})}{\sum_{k=1}^{J} \exp(\beta x_{nk})} \quad , \tag{4.3}$$

where β is a vector of coefficients that varies across individuals, and x_{nj} is a vector of the attributes of each choice that was made. All covariates met the normality assumption following log-transformation. In keeping with economic theory, distance is a proxy for cost, so enters the model with a negative sign, and expected revenues enter with a positive sign (Train, 1998; Ran *et al.*, 2011). The analysis was carried out in the SAS package PROC MDC (SAS, 1999) using quasi-Newton optimization and 100 Halton draws, and was re-run in the R mlogit package (R Development Core Team, 2008) to cross-validate results. The resulting lognormal coefficients of the mean, b, and standard deviation, s, for the log of β required back-transformation to provide correct interpretation (see Ran *et al.*, 2011), e.g. for ln(β), the median, mean and standard deviation can be calculated as follows: exp(b), exp[b + (s^2/2)], and exp[b + (s^2 /2)]$\sqrt{\exp(s^2) - 1}$.

4.2.4 Selection of explanatory variables

Fishing is a risky business, and predicting catches and revenues in advance is difficult, so experience and knowledge of fishing locations is important. Therefore, rather than using revenue and costs per trip as the utility (as measures of economic gross benefit or economic costs), we use value per unit effort (vpue). We assume that vpue is a proxy for net benefit (i.e. utility) and that targeting of a stock would be based on its vpue because fishers would attempt to target the most valuable species, and any reduction in vpue would indicate that a species had been depleted or the market and effort diverted to a less valuable species. The variable vpue can be computed from fishing in the same location in the same month of the previous year (i.e. lagged average vpue). The vpue had to be used because, although obtaining cost data for each decision unit is possible, we had no access to individual cost data. Moreover, accessing individual cost data is expensive in terms of research effort, and the economic data are anyway generally confidential in nature. In order to take account of strong spatial and temporal fluctuations and strong (or weak) year classes in the target species, a lagged average vpue was used on a monthly scale in the within-year of fishing as a proxy for the attractiveness of fishing in the same location as the previous month. This variable captures the within-year seasonal trends. Table 4.1 lists all the covariates estimated in the model. Not present in the skippers' logbooks was fuel consumption, so distances to the port of landing were weighted by marine monthly average diesel price per litre over the study years as a proxy for cost, because true trip costs were not available. The assumption is that before a skipper proceeds to the fishing grounds, he already has a good idea where he will land his fish in order to achieve the best return (Caddy and Carocci, 1999). Distant sites are expected to have better quality fish stocks, however, so the choice of how far to travel is a trade-off between higher travel costs to distant grounds and the expected better quality catch there. Distance was calculated using the Haversine formula (Sinnott, 1984), using the distance from the centre of the ICES statistical rectangle where a declared landing was taken to the port of landing for each trip in a particular month. A mean distance was then calculated by year, month and Roundfish area. The distances in our model are the average kilometres from fishing in the same location in the previous month of the same year, so they take account of the expected travel costs and the landing behaviour of the fleet. It was assumed that fishers would have received prior information of where to land, so reflecting better market prices for the distance travelled to land their catch (Mathiesen, 2003). Table 4.2 lists the average values for the chosen covariates for each spatial unit for 1997, to illustrate the scale of covariate values and differences from one area to another.

Table 4.1 Definition of the variables used in the random utility model (RUM).

Variable	Definition
plelagyr	Average vpue of plaice from fishing in the same location in the same month in the previous year.
Sollagyr	Average vpue of sole from fishing in the same location in the same month in the previous year.
timelagyr	Percentage effort spent in the location in the same month the previous year.
Plelagm	Average vpue of plaice from fishing in the same location the previous month in the actual year of fishing.
Sollagm	Average vpue of sole from fishing in the same location the previous month in the actual year of fishing.
timelagm	Percentage effort spent in the location in the previous month in the actual year of fishing.
Distcost	Average distance to port of landing from the same location the previous month in the actual year of fishing weighted by the fuel price*.

*Average marine fuel prices (£ per litre, excluding VAT and duty); source, DECC (UK Department of Energy and Climate Change).

Table 4.2 Mean values of the input variables for 1997 (as an example year) over all months.

Roundfish area	vpue sole (£ per h)	vpue plaice (£ per h)	Distance (km)	Trips (%)
2	3.2(96)*	143.8(24)	440.3(4)	18.3(36)
4	27.2(107)	73.2(29)	249.9(22)	6.9(41)
5	34.4(58)	30.5(75)	156.8(37)	29.1(24)
6	10.1(77)	123.6(13)	316.9(11)	29.7(23)
7	3.4(105)	138.6(34)	409.2(8)	15.9(36)

The coefficients of variation (*CV*) associated with the variables are given in parenthesis, showing variation for distance (as ports vary) and variation in the other variables attributable to individual differences for each decision unit and trip.
*Note the large variation, because sole catches are minimal in this area.

It is not unreasonable to assume that fishers are profit maximizers (Robinson and Pascoe, 1997), basing their decisions to fish in a certain location on catch rate, effort and essentially economic return. However, previous effort allocation (an average of the entire beam trawl fleet) also adds to experience and knowledge gained of a location and contributes to the utility of a choice. Fishers tend to choose the same areas, based on previous experience, and

apply habitual behaviour, which in this case is referred to as a habit variable. Therefore, the utility of the location choice is modelled by the observed choice of location last year (% effort spent) in the same month (i.e. lagged location). The explanatory variables within the model were calculated as a mean by year, month and area (i.e. for each trip in a particular month and ICES rectangle, a mean was calculated by year, month and Roundfish area) for the fleet, the result of which made the choice set for year, month and area. This set was merged with individual trip data by year, month and area, such that for every trip, the decision-maker had a choice. If the choice was made, the values took a value of 1 if chosen, or 0 otherwise.
The analysis was carried out in two steps. First, the RUM was fitted to the fishing trip dataset in nine time-windows (each three years long), each with a monthly time-step. These nine time-periods were 1997–1999, 1998–2000, 1999–2001, 2000–2002, 2001–2003, 2002–2004, 2003–2005, 2004–2006, and 2005–2007. Note that because lagged variables were used as explanatory variables (as an example, vpue in the same month of the previous year), data from the previous year (starting from 1996) were used to predict choice in the current year (i.e Lagged vpue for a particular month in year 1 = $-m$; lagged annual vpue in year–1 = m). These nine time-windows were used to evaluate whether alternative explanatory variables were apparent because of differing circumstances (economic or habit), or through changes in management, the populations being fished, or other factors.

The second step involved using the selected best models (based on each time-window) to predict future choice by fishers. Therefore, monthly time-series of predicted fisher location choice were projected over the periods corresponding to each of the above models (different cumulative time periods depending on the original model time-period): 1999–2007, 2000–2007, 2000–2007, 2001–2007, 2002–2007, 2003–2007, 2004–2007, and 2005–2007. Here we were attempting to get an indication of each model's predictive capability, at least partially.

We were also replicating a typical analysis that would have been performed by a researcher who would have cross-checked a model's predictive power by fitting over a time-period, predicting ahead one year, then later cross-checking predicted against observed values. Here, it is important to acknowledge that as tactics change over time, they result in differences in the significance of the explanatory variables, as noted above. This provides the rationale for the cross-validation as carried out. A likelihood ratio test was also conducted on the constrained model (log-likelihood under the null hypothesis) fits against an unconstrained model, to determine whether any model reduction was necessary (and to check the hypothesis that the random parameters are uncorrelated). This statistical test provides a comparison of the random effects model (null model) over its simpler form, a deterministic conditional logit model. The test describes how many more times likely one model is over the other. The resulting *p*-value indicates the significance (usually <0.05) of whether to reject the null model over the simpler model. The mixed logit was also tested for the IIA property using the Hausman test (Hausman, 1978). The assumption behind this test is to estimate the model with all the choice sets, then to reduce it to a small set of alternatives, and then to re-estimate. The resulting estimates should not change when the alternatives are removed, and the two models can be compared and tested for IIA. If IIA holds, the null model is said to be efficient, otherwise the model is said to be inconsistent and IIA does not hold.

4.3 Results

All statistical fits to the RUM were significantly better than null models (likelihood ratio test; Table 4.3), so the mixed model was considered the best model in terms of likelihood. The likelihood ratio tests suggested that all random coefficients were important additions to the model fits and clearly reject the hypothesis that the random parameters are uncorrelated. However, a direct comparison is not correct because of the degrees of freedom in the two models. Results from the Hausman test for IIA after Roundfish area 2 was removed from the data and re-estimated for all the fits showed that all models failed, giving a χ^2 value of between 0.006 and 0.02 and p = 0.99. As a test, the models were reduced to the simpler conditional model and the results indicated that it passed the IIA assumption, giving a χ^2 value of between 23.8 and 74.0 and $p < 0.05$, proving that the mixed model was the correct model to have used. The significant variables and their estimated coefficients for each of the models are listed in Table 4.4. Several variables had a significant influence on the utility and probability of location choice, including distance to landing port from fishing grounds, expected revenue of plaice and sole, and past habits on the same fishing grounds. In general, the coefficients of the estimated variables were consistent with expectations; a positive sign was observed for expected revenues and a negative one for expected costs (Table 4.4). The signs of the standard deviations in some instances are negative, but for estimation purposes they are free to take any sign, because the normal distribution is symmetrical around its mean, and the absolute value can be taken to estimate the variance. The estimated standard deviation of the coefficients in Table 4.4 show highly significant estimates of some of the drivers for location choice, indicating that the parameters (timelagyr, timelagm, sollagm) vary in the population of fishers.

Table 4.3. Results of the likelihood ratio test for each of the model fits, with d.f. representing the degrees of freedom for the constrained and unconstrained model, and d.f. Chisq the χ^2 value with the degrees of freedom equal to the difference in the number of degrees of freedom between the two models. Statistical significance at **, 5% level, and ***, 1% level.

Model	**1997–1999**	**1998–2000**	**1999–2001**	**2000–2002**	**2001–2003**
d.f.					
Unconstrained	35	35	35	35	35
Constrained	14	14	14	14	14
log-likelihood					
Unconstrained	–8881	–8063.3	–7307.1	–6481.5	–5766.5
Constrained	–8916.6	–8093.4	–7337.8	–6537.1	–5822.2
d.f. Chisq	–2169.9	–2160.3	–2161.3	–2111.3	–2111.5
	***	***	***	***	***

Model	**2002–2004**	**2003–2005**	**2004–2006**	**2005–2007**
d.f.				
Unconstrained	35	35	35	35
Constrained	14	14	14	14
log-likelihood				
Unconstrained	–4659.6	–3921.9	–3336.8	–2871.7
Constrained	–4706.0	–3944.1	–3349.6	–2892.4
d.f. Chisq	–2192.7	–2144.3	–2125.6	–2141.4
	***	**	**	**

Table 4.4 Estimated lognormal parameter estimates for each of the models. Parameters were (a) plelagyr: Average vpue of plaice from fishing in the same location in the same month in the previous year; (b) sollagyr: Average vpue of sole from fishing in the same location in the same month in the previous year; (c) timelagyr: Percentage effort spent in the location in the same month the previous year; (d) plelagm: Average vpue of plaice from fishing in the same location the previous month in the actual year of fishing; (e) sollagm: Average vpue of sole from fishing in the same location the previous month in the actual year of fishing; (f) timelagm: Percentage effort spent in the location in the previous month in the actual year of fishing; and (g) distcost: Average distance to port of landing from the same location the previous month in the actual year of fishing weighted by the fuel price.

Parameter	**1997–1999**		**1998–2000**		**1999–2001**		**2000–2002**		**2001–2003**		**2002–2004**		**2003–2005**		**2004–2006**		**2005–2007**	
sollagyr_M	0.0405		0.00548		0.0426	.	0.0167		0.0441	.	0.0242		0.0507	.	0.0209		0.0317	
sollagyr_S	–0.1256		–0.1484		0.00509		0.0554		0.0737		–0.0498		0.0289		–0.00656		0.00482	
plelagm_M	0.0714		0.0806		0.0304		0.1153	*	0.0745		0.0156		0.0534		0.058		0.0813	
plelagm_S	0.6663	***	0.0205		0.00497		0.00025		0.00116		–0.0125		–0.00408		0.0179		–0.00267	
sollagm_M	0.0302		0.0926	***	0.1318	***	0.1829	***	0.1799	***	0.0262		–0.0276		0.0519		0.118	**
sollagm_S	0.2156	*	–0.285	***	–0.2213	**	–0.2531	***	–0.3071	***	–0.0846		0.0054		–0.00556		0.00268	
timelagm_M	0.5504	***	0.4914	***	0.2646	***	0.3919	***	0.3469	***	0.5762	***	0.5442	***	0.5951	***	0.6391	***
timelagm_S	–0.2174		–0.0174		0.0204		0.1636		0.1585		–0.4889	***	0.5556	***	0.5686	***	–0.5554	***
plelagyr_M	0.1799	**	0.1476	***	0.17	***	0.1222	*	–0.00577		0.2069	***	0.2064	***	0.2413	***	0.0879	
plelagyr_S	0.7301	***	–0.0249		–0.00247		0.00179		–0.00173		0.0144		0.00071		0.00017		–0.0149	
timelagyr_M	0.323	***	0.3629	***	0.6843	***	0.5788	***	0.5621	***	0.4618	***	0.5575	***	0.5267	***	0.5222	***
timelagyr_S	0.5094	***	0.00322		–0.4376	***	0.5267	***	–0.4971	***	–0.4991	***	–0.3758	***	–0.5151	***	–0.499	***
distcost_M	–0.1382		–0.1512	*	–0.106		–0.2511	***	–0.2859	***	0.1174		0.1558		–0.151		–0.4989	**
distcost_S	0.3732		0.0122		0.00987		–0.0122		–0.0016		–0.0326		–0.00415		0.0051		–0.00022	

* Statistical significance at *, 10% level, **, 5% level, and ***, 1% level.
Parameters marked _M are the lognormal mean coefficients and _S are their between-population standard deviations.

Figure 4.3 provides a visual summary of the changing relative influences of different factors on fleet decisions, over the nine time-windows (representing short and long term). This represents a transition between changing tactics (in the short term) and changing strategies (in the long term). The results highlight the noteworthy pattern (shown by the cells shaded darker) that past monthly effort in the year of fishing (timelagm) and fishing in the same location as the same month the previous year (timelagyr) is common and dominant in every model fit, implying a positive tactic by the fishers to choose an area based on past effort. Another variable which has a positive influence over almost all nine time-periods fitted is the variable for past catch rates of plaice (plelagyr).

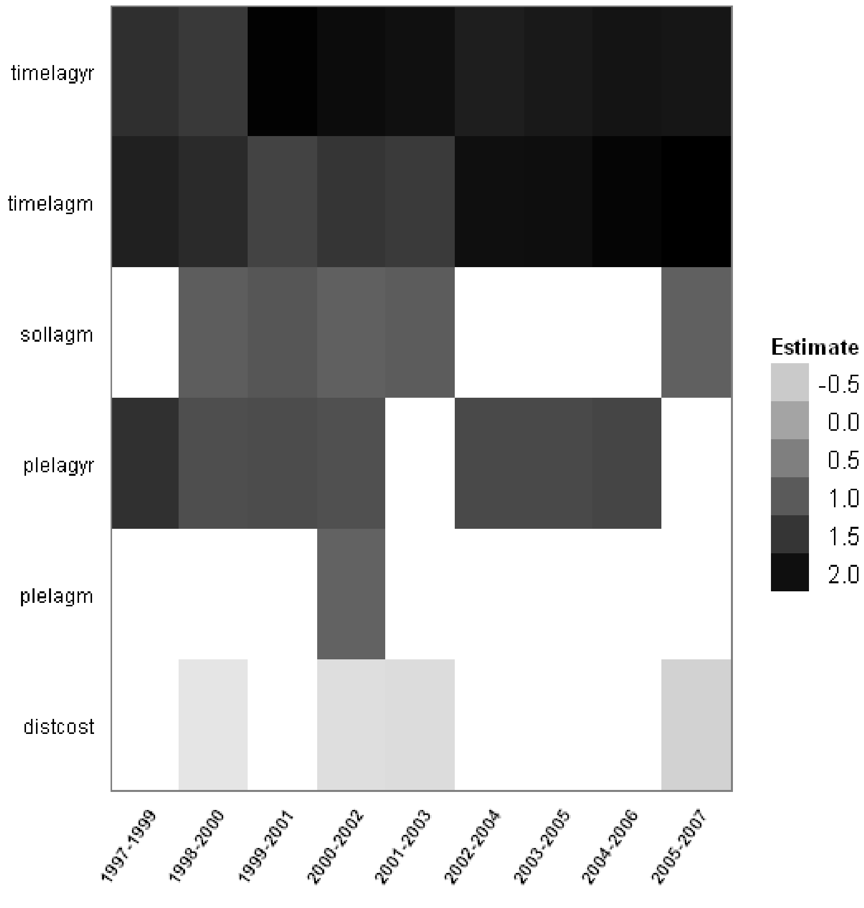


Figure 4.3 Heatmap of the transformed mean parameter estimates for each significant ($p < 0.05$) variable, showing the relative importance of the different variables over time (See Legend of Table 4.4 for explanation of variable names).

Model fits for the period 1997–1999 were more prominent in colour and showed that the fishers' tactics appeared to be based on past expected revenue of plaice. There is also substantial variation in the influence of the different variables across the model fits (a lack of homogeneity in the shaded cells across columns). This implies that fisher tactics were changeable across the different time-windows. For example, for the 1998–2003 fit, the expected revenue of sole from fishing in the same location in the same month of the same year (sollagm) had a noticeable influence. Conversely, the plaice coefficient was insignificant other than in the 2000–2002 fit. This was not consistent throughout all fits, because it was not until the fits of 2005–2007 did they reappear as significant, displaying an obvious change in tactics. The expected revenue of sole from fishing in the same location in the same month in the previous year (sollagyr) is a noticeable absentee from all fits, implying that it was not a significant factor in determining location choice. The distance proxy (distcost) displays significant negative coefficients in four of the year fits, suggesting that fishers were affected by changes in fuel prices. The lack of significance of the distcost coefficients in other years possibly suggests that distance travelled to fishing grounds is traded off against the value of the catch, such that the costs to reach the best fishing grounds are compensated for by better catch rates there. Interestingly, the observations of significance in fuel price, the gap in the significance of expected sole and plaice revenue (sollagm, plelagyr), and the different strengths of the habitual effort (timelagyr and timelagm), coincide with the change of ownership of the fishing vessels from the UK to the Netherlands (see Figure 4.2). Over the longer term (1997–2007), past annual and monthly effort (timelagyr, timelagm) were the most persistent driving factors influencing fisher choice (Figure 4.3).

Elasticities were calculated for plelagyr (the average vpue of plaice from fishing in the same location in the same month in the previous year), and distcost (the average distance to port of landing from the same location the previous month in the actual year of fishing weighted by the fuel price), for model fit 2000–2002; this fit was chosen because it yielded the most significant contribution of the coefficients. The effect of a 50% increase/decrease in value/cost was explored with respect to a change in the probability of location choice relative to the model predictions. A 50% decrease (50% increase) in plelagyr had a negligible effect on the predicted location choices throughout the time-series except in July 2007, when there was an 8% increase (4.8% decrease) towards the probability of fishing in Roundfish area 5 and <0.02% reductions (0.01% increases) in the probabilities of fishing in other areas. In contrast, distcost had a much more persistent and greater effect throughout the predicted time-series. A 50% decrease in distcost resulted in a 19% increase in the probability of choosing Roundfish area 5, and small reductions of ~0.04% for other areas. A 50% increase in distcost resulted in a 10% decrease in the probabilities of choosing Roundfish area 5, with small increases (~0.02%) towards selecting Roundfish areas 2, 4, 6 and 7.

4.3.1 Predicting future choice

The predictions for all model fits through time are presented in Figure 4.4 along with the observed percentage of trips in each Roundfish area (black line in the Figure). Predictions were computed using the estimated significant parameter estimates and the mean values of these variables at a monthly scale [Equation 4.3]. Overall, the models (shown as different colours of line) yielded good fits relative to the observed (black line) monthly time-series (Figure 4.4). The models predict the effort allocation in Roundfish areas 2 and 6, possibly because these are the main fishing grounds for plaice, and have expected good catch rates (Figure 4.4).

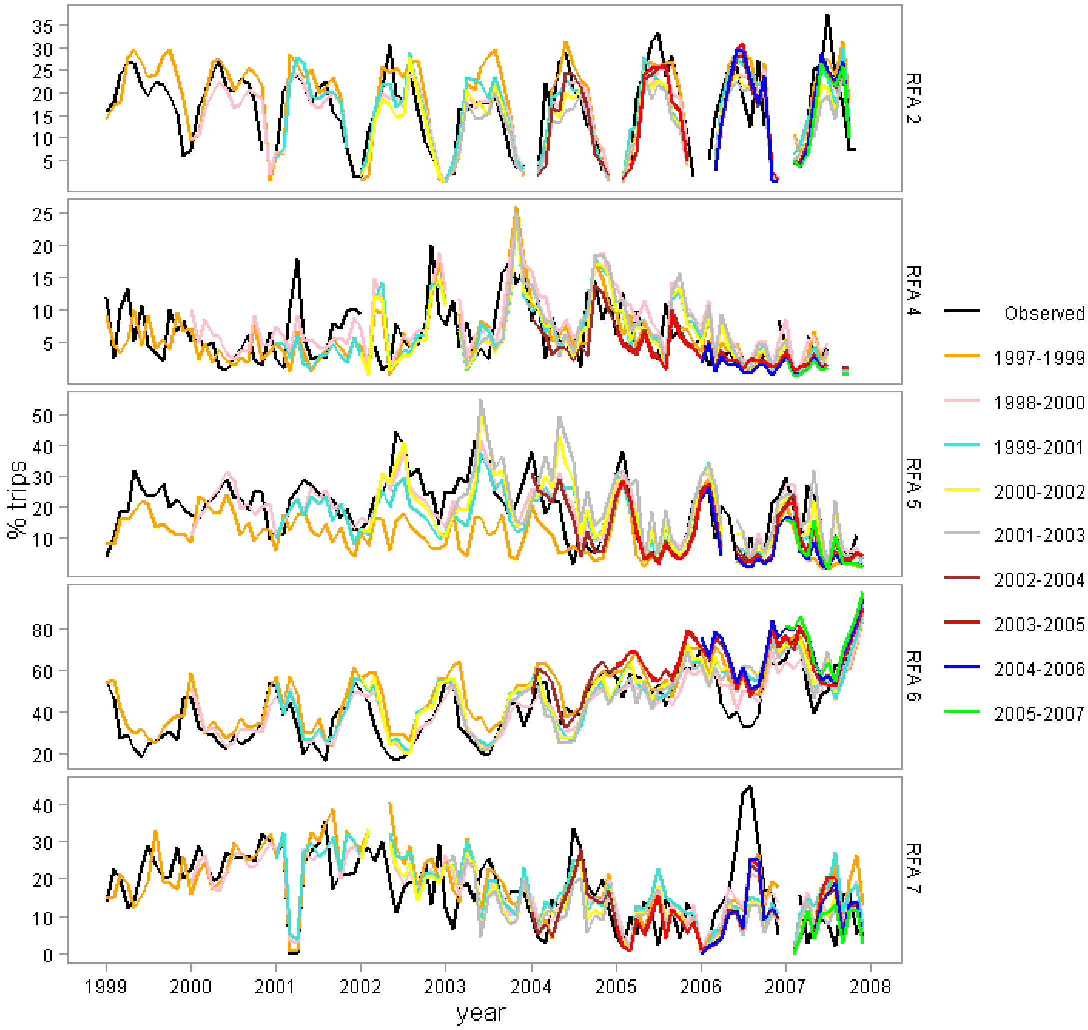


Figure 4.4 Plots of eight of the model predictions based on fits to the data, showing the relationship between the percentages of predicted and observed fishing trips, the black line representing the "perfect" fit and RFA meaning Roundfish area.

The model fit to data over the period 1998–2000 was used to predict effort reallocation during the closure in 2001. As Roundfish area 7 encompassed part of the study area, we simulated a closure by forcing all variables in the area to take a value of 0. Using the estimated model coefficients, the probabilities of different trip choices were predicted, then compared with actual trip choice to assess the degree of effort redistribution (Figure 4.5). The percentage of trips to Roundfish area 7 predicted to reallocate effort during the closure to Roundfish area 2 for the months March and April 2001 were 23 and 25%, respectively, compared with the observed percentages of 23 and 24% of trips (20 and 24% in 2000).

Roundfish area 4 showed predicted estimates of 4 and 9% compared with observation percentage allocations of effort of 10 and 18%[4] (5 and 5% in 2000), Roundfish area 5 showed predicted estimates of 20 and 24% compared with observations of 23 and 26% for percentage reallocation of effort (21 and 23% in 2000), and Roundfish area 6 showed predicted estimates of 53 and 42% compared with observations of 45 and 32% for percentage reallocation of effort (30 and 39% in 2000). The notable differences were in Roundfish areas 4 and 6 in April, for which there were 9 and 10% over- and underestimates of predicted vs. observed, respectively. Most of the predictions are, however, reasonable for the choices made during the closure period (Figure 4.5).

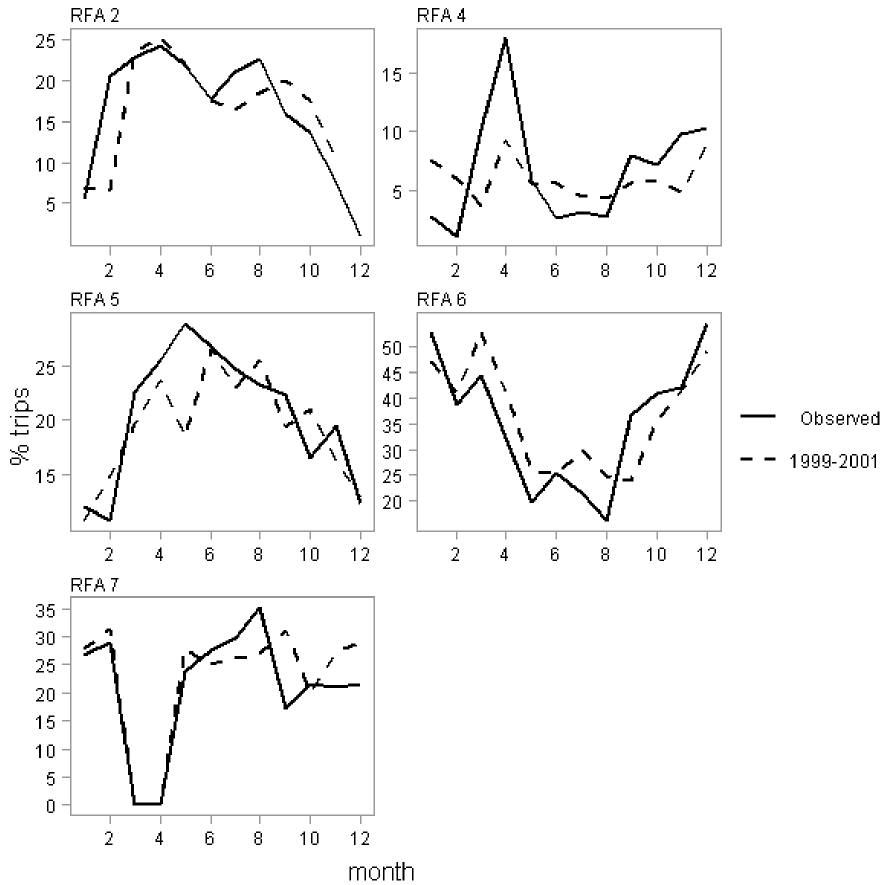


Figure 4.5 Model predictions from the 2001 closure simulation, based on closing Roundfish area (RFA) 7.

[4] The EU flag vessel legislation requires member states to have some economic link with its national fisheries communities. During the closure period, the economic link rules applied, so because the western part of the North Sea (RFA 4) was open rather than closed, foreign owned vessels having UK quota could land their catches in Grimsby.

4.4 Discussion

The work documented here has described a novel method of predicting choice of fishing location for the English North Sea beam trawl fleet fishing in the southern North Sea, using a mixed model. The model showed good fits relative to the observed monthly time-series (Figure 4.3) and predicted the general patterns of spatial change by fishers over time. Model variability in prediction is apparent in Roundfish areas 4, 5, and 7 compared with Roundfish areas 2 and 6, where the main plaice and sole grounds are respectively located. The model was also used to simulate part of the cod closure in 2001 (Figures 4.1a and 4.5), and showed good agreement with actual observations on a monthly time-scale.

One of the key findings from this study is that although fishers' tactics are driven by persistent long-term habits, there are also shorter-term subtleties driven by additional issues that can vary in their relative importance over time. The utility of fishing in a location (a distinct fishing area) depends on previous success measured as good catch rates in terms of economic vpue, as well as previous experience, in this case a measure of past fishing practice monthly and annually (the effort allocation variable; Hutton *et al.*, 2004). Therefore, the results of the RUM analysis here reveal some of the assumptions that could be expected *a priori* for location choice. Essentially, some previous knowledge or experience of a given area has the dominant bearing on the decision whether or not to fish there. In addition to past experience, we also found that cost (i.e. distance to port of landing and fuel prices) was an important driver of choice (see Abernethy *et al.*, 2010). The results of the analyses also revealed that fishers made their decisions based on past habitual behaviour/previous experience in combination with targeting for plaice (i.e. one-year lagged vpue), fuel price, and past monthly catch rates of sole. The heterogeneity in the selection of fishing grounds by fishers is attributable to individual variations in decision along with other unexplained factors. The mixed model handles this type of heterogeneity and makes it a useful tool for fisher choice modelling approaches.

Past and current failings of fishery management relate partly to uncertainty in the stock assessments and the management. These range from different sources of model error, through biased input data or process error, to implementation error (Peterman, 2004). Figure 4.4 is an example of the temporal and spatial variation or uncertainties of fishing patterns attributable to model error. To reduce these uncertainties, there is a need to improve understanding of the processes driving location choice, i.e. more-detailed economic (fuel, market prices), social (employment), biological (recruitment, spawning-stock biomass) and regulatory (quotas, technical measures) influences. Of course, many processes are complex and interrelated, and it is difficult to account for all the uncertainty, but each process needs to be understood better along with the sources of the uncertainty. This study progresses our understanding of the drivers of this fleet significantly in terms of the short-term choice of location both temporally and spatially, which appear to be largely driven by habit, but also by other subtle drivers. In an environment where change is the norm, fishers develop tactics and strategies to survive when faced with rising fuel costs, fluctuating stock levels, regulations, and market conditions (some of which can be observed in our study; see also Abernethy *et al.*, 2010). In a management context, it is important to understand fisher behaviour in the face of a changing environment so as to manage the system better (Hilborn, 1985; Fulton *et al.*, 2011). This is especially important when considering closed areas or marine protected areas, MPAs.

4.4.1 Conclusions and future work

To conclude, the implementation here of a discrete choice mixed model allowed us to explore and improve understanding of English and Welsh beam trawl fisher short-term tactical behaviour over a 12-year period. The results confirm the notion that expected revenues from target species, experience or habit, and fuel prices are significant factors in determining fisher decision-making. Some of the unobserved random components of the model causing heterogeneity in the selection of fishing grounds by fishers could be attributable to individual variations in decision-making along with other unexplained factors. For example, factors that we have not captured could include skipper skill, age, nationality and vessel attribute. Compiling data on these factors to investigate the influence fisher attributes would be a valuable aim of future work. Nevertheless, even without these, model predictions were similar to observed choices during the study period, and the simulated closure we modelled resulted in discrepancies of location choice of just 9 and 10%.

Future application of the fleet behaviour model taking account of implementation error within a Management Strategy Evaluation (MSE) framework could help evaluate future stock levels and the profitability of this fleet (Pilling *et al.*, 2008). The main factor that could contribute to this analysis would be the accuracy of predictions of location choice based on knowledge of the two main target species, bearing in mind the fact that fisheries have historically been managed on a stock-by-stock basis. Although several studies have been published on the North Sea sole and plaice fishery (Kell *et al.*, 1999, 2005; Ulrich *et al.*, 2007; Kraak *et al.*, 2008; Andersen *et al.*, 2010), the work reported here on the spatial dynamics of the fleet may complement future research effort, as it has in other MSE spatial studies (Pelletier and Mahévas, 2005; Bastardie *et al.*, 2010; Lehuta *et al.*, 2010). Such an analysis could provide an insight into mixed fishery management, because in the short term, an approach needs to be developed to resolve conflicting management advice for different species in the same fishery.

Chapter 5. Effective fishing effort indicators and their application to spatial management of mixed demersal fisheries[5]

Sole *Solea solea* (Source: CEFAS).

[5] Tidd, A.N., Effective fishing effort trip indicators and their use for efficient spatial management in mixed demersal fisheries. Fisheries Management and Ecology, doi: 10.1111/fme.12021.

Abstract

Since Common Fisheries Policy reform in 2002, there have been various proposals for designing effective input-management tools in the context of demersal multispecies and multimétier fisheries, to augment quota management. The relationship between fishing mortality and effort exerted by the English beam trawl fleet is investigated for two stocks of North Sea demersal fish, plaice, *Pleuronectes platessa* L., and sole, *Solea solea* (L.). Catchability was adjusted by accounting for targeting by this gear, seasonal and area effects, and individual vessel variation, using results from a generalized linear mixed effects model (GLMM) that included random effects (in this case, vessel). Descriptors were standardised in relation to distinct submétiers and their impact on both species. Fishing efficiency was calculated as the ratio between relative nominal landings per unit effort derived from the GLMM and survey indices from a standard survey vessel. Fishing efficiency for sole increased (+0.6% annually) and for plaice decreased (−6.2%), probably because of changes in targeting, fuel costs and regulations.

5.1 Introduction

The management of European mixed fisheries is primarily based on total allowable catches (TACs) along with effort restrictions (days-at-sea), technical measures (gear and/or mesh size regulations, size restrictions) and seasonal closures. The difficulties in managing fish stocks through TACs are widely recognised (Shepherd 2003; Beddington *et al.,* 2007). The main issue is that a TAC set to protect one species within a mixed fishery can have an undesired effect on another through increased discarding, or indirectly through foodweb interactions. Hence, a conservation policy cannot achieve its goal through this single management action. For example, a TAC for one species in a fishery may be exhausted earlier in the year than for another species taken by the same fleet/fishery (Vinther *et al.,* 2004). The fleet could then continue to fish the same grounds until it landed the TAC remaining for each target species, but any catch of a species for which the TACs were exhausted would have to be discarded. Discarding species that almost certainly die on return to the sea or the illegal retention of the catch leads to socially undesirable results (Copes 1986). Since the Common Fisheries Policy (CFP) was initially revised in 1992, fishing effort management schemes have had an increasing role as tools to control fishing mortality. Effort management differs from TACs in that controls on effort manage the input rather than the outputs specified by a TAC, although they both aim to limit fishing mortality.

In fisheries science, fishing effort (E) is an essential parameter in the assessment of fish stocks and their effective management. It is linked to fishing mortality (F) via the catchability (q) at age of a stock, a term that generally means the extent to which the stock is susceptible to fishing and that would be captured by one unit of effort. Catchability is therefore as important to managers as effort in assessing fish stocks and ultimately in supporting effective management. The relationship is assumed to be linear and takes the form $F=qE$. Fishing effort, however, is difficult to quantify because the sizes and types of vessels and gears differ. It is usually approximated by a metric of capacity, such as gross tonnage or engine power, with a measure of activity (e.g. days-at-sea or hours fished), and is therefore an aggregated measure of fisher behaviour in locating the greatest densities of marketable fish (Rijnsdorp *et al.,* 2006). Nevertheless, capacity has not always decreased at the same rate as stocks (Cunningham and Gréboval 2001), and as resources are depleted, fishers tend to redistribute their fishing effort across other fisheries, implement new technologies such as advanced fish-finding devices (Branch *et al.,* 2006), or participate in illegal, unreported and unregulated (IUU) fishing (Agnew *et al.,* 2009). Additionally, vessels and/or gears may be modified to circumvent regulations and/or to increase effective fishing power, in an attempt to continue harvesting at the most profitable level (Gréboval, 1988).

The efficiency of fishing vessels and hence catchability tends to increase over time because of factors such as fishing technology improvements. This increase, known generally as 'technological creep', can be quantified in relation to fishing mortality with constant nominal effort (E_n) and intensified effective effort (E_e). These relationships are important to fishery managers because they are crucial in reducing fishing mortality through effort control, and ignoring them could prove meaningless in limiting fishing mortality (Pauly *et al.,* 2002). Shepherd (2003) suggested that for a given amount of effort exerted, and because of variations in vessels and their activity, different effects on stocks can be generated. Therefore, it would be necessary to set effort limits at the individual level based on area fished and gear used. Standardized fishing effort has been interpreted in the literature, however, in different ways, and there is some contention within the fisheries scientific community as to what it actually means, and also as to how any problem should be addressed. Many authors have tackled it using statistical regression models (Maunder and Punt, 2004), where some dependent variable, e.g. catch per unit effort (cpue), is modelled as a function of plausible explanatory factors such as seasonal, temporal and gear characteristics (Hilborn and Walters 1992; Weninger and McConnell, 2000; Hinton and Maunder, 2003; Mahévas *et al.,* 2004; Piet and Jennings, 2005; Bishop 2006; Marchal, 2008). The parameters from such models are then used to estimate the value of the variable in question for any combination of seasonal, temporal and technical (e.g. gear) factors. Since the 2002 CFP reform, there have been various management and recovery plans, as well as some difficulties in designing relevant, efficient and effective management tools in the context of multispecies, multimétier fisheries. Hence, there is an increasing role for input management as part of ongoing CFP reform.

The aim of the present study was to evaluate the relationship between fishing mortality and nominal effort applied on two North Sea demersal stocks, plaice (*Pleuronectes platessa*) and sole (*Solea solea*), caught by the English beam trawl fleet, using an adaptation of the commonly used general linear model (GLM; Nelder and Wedderburn, 1972). Effort indicators for UK fleet capacity based on vessel capacity units (VCUs[6]) determined by vessel size and engine power, and hours fished were used rather than the more traditional metrics (e.g. kW and days at sea). Methods of standardising such descriptors in relation to submétiers and their impact on both species are suggested, allowing for potential changes and strategies in the fishery to be evaluated. The basis for the approach is to resolve potential conflicting spatial management advice for different species that can be taken in the same fishery, and which could be applied at an individual level, as suggested by Shepherd (2003). Multispecies fisheries are difficult to manage, so advice at the fleet or fishery level may be more effective than trying to balance and integrate single-species advice for a range of stocks (Vinther *et al.,* 2004). This means that altering effort controls or spatial regulations for one stock can have implications on others and the wider ecosystem.

5.2 Methods

English beam trawl vessels in the North Sea have traditionally caught plaice in a directed fishery using 120 mm mesh north of 56°N, and in a mixed fishery with sole, using 80 mm mesh, in the southern North Sea. In 2005, international landings of North Sea plaice amounted to 55 700 t, compared with a peak of 170 000 t in 1989. Reported international landings of plaice from the North Sea were dominated by the Netherlands (40%), followed by the UK (23%) and Denmark (20%), with Belgium, Germany, France, and other countries reporting the remaining 17% (ICES, 2007). In the English fishery, the high value of sole makes it one of the most important species targeted by inshore vessels operating trawls and fixed nets. The fishery is conducted mainly from March to October, but sole are also taken as a target species by offshore beam trawlers, otter trawlers and gillnetters. The English North

[6] A VCU is a unit used by the UK as part of fleet capacity management (see UK Fisheries Department 1988).

Sea beam trawl fleet until 2003 operated mainly out of east coast English ports, typically spending on average 250 d at sea in trips lasting about 6 d (Hutton *et al.,* 2004). Since 2002/2003 and the transfer of ownership to the Netherlands, however, skippers have generally targeted sole because of its greater commercial value and short distance from their Dutch home port.

5.2.1 Data

Individual trip data for the commercial beam trawlers were collated for the years 1997–2007 and examined by area. These areas were based on International Bottom Trawl Surveys (IBTS) and in particular the Netherlands beam trawl survey (BTS), which stratifies its sampling of sole and plaice by Roundfish areas (Figure 5.1; ICES, 2009a). Roundfish areas 1 and 3 were excluded from the study because English beam trawlers generally do not fish there. The data collected for each vessel and trip included species landed, hours fished, landed weight (kg) per ICES statistical rectangle, month, year, and total value of the catch by species. Within the EU, it is a requirement for vessels >10 m long to submit logbooks, but the database contained a subset of catch from <10 m vessels that historically reported their catches. Fleets were defined to align with those in the data collection regulation (DCR) of the European Commission (EC, 2000). A method was developed independently (see EC, 2006a), preceding the current data collection framework (DCF; EC, 2008) that defines the beam trawl fleet on the basis of its use of a beam trawl for >50% of a fishing trip. The fleet activity, or métier, is determined by a fisher's tactic at a trip level, and is defined on the basis of the mix of target species. In other words, métiers are characterized on the basis of the outcome of a trip and defined by gear, fishing grounds and composition of landings. The compositions of landings were calculated as a proportion of the total value of the catch, thus removing the differences in catch rate attributable to vessel capacity. Catch proportions were based on economic value rather than weight, reflecting the notion that fishers are profit maximisers, so valuable species received more importance in the analysis. In this study, the beam trawl métier that primarily targets crustaceans (brown shrimp) was omitted, and a single demersal métier was defined (demersal beam trawl) and used for analysis. The fleet targets the main commercial flatfish stocks (plaice and sole) in the North Sea.

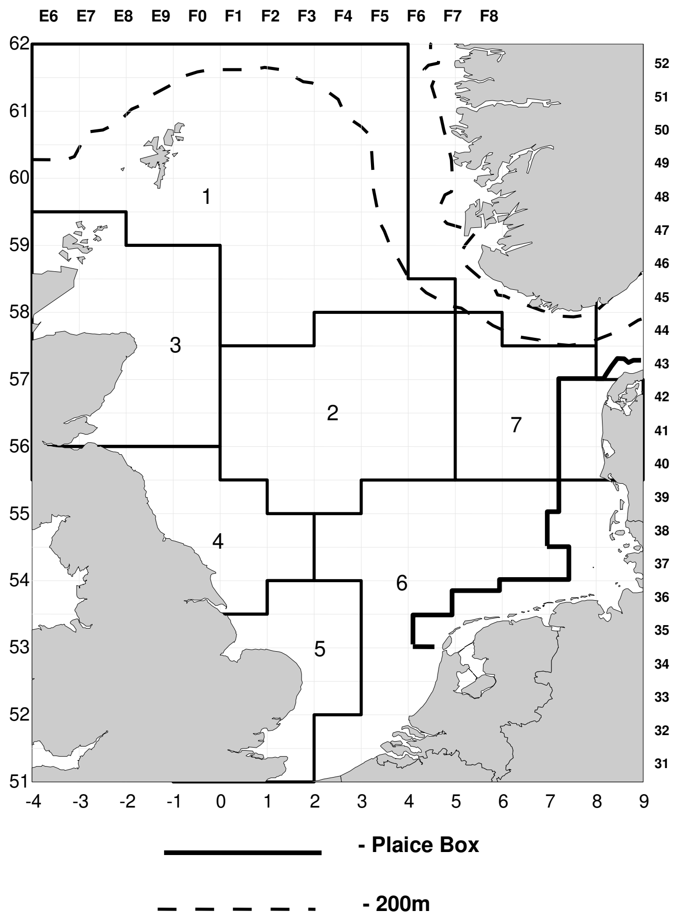


Figure 5.1 Map of the North Sea showing ICES statistical rectangles and roundfish areas (1−7), with the plaice box indicated by the heavy dark line (closed to beam trawlers of hp >300 for the whole year since 1994).

5.2.2 Exploratory analysis and covariates

Vessel landings per unit effort (lpue) were calculated from logbook-recorded landings as kg per h fished per vessel per trip per area (ICES statistical rectangle; Figure 5.1). Although haul-by-haul data are preferred for such analyses, logbook declarations are by day and by ICES statistical rectangle per trip. The underlying statistical distribution generating the data was also hypothesized to be of the form of a gamma distribution, but after examining the data, a lognormal distribution was investigated and normality tested using Q−Q plots. In keeping with other studies (e.g. Butterworth, 1996; Ortiz *et al.,* 2000; Ortiz and Arocha, 2004), zero lpue values were addressed by the addition of a positive constant of 1, because the logarithm of 1 is 0 (Ortiz and Arocha, 2004).

Vessel capacity units, i.e. overall length × breadth of vessel (both in m) + engine power (kW) × 0.45, was chosen simply because this metric is used in policy and combines characteristics recorded in the UK fleet register. Unfortunately, other potentially relevant covariates, such as the electronics used (e.g. global positioning systems, GPS, plotter software, fish finding equipment, seabed mapping and navigation systems), skipper and crew experience in the fishery, and specific technical characteristics of the gear, are not available from logbooks or fleet registers. These can be obtained only by face-to-face interviews with the skipper, and would also change over time. Year was included as a factor to capture temporal changes in technology or fluctuations in stock abundance. Month and area (ICES rectangle) were included to account for strategic/tactical effects (e.g. responding to seasonal changes in stock abundance). Vessel effect was considered to be an important factor and included, because it could be an indication of skipper/crew experience and gear characteristics (Mahévas *et al.,* 2011).

5.2.3 The model

Generalized linear mixed effects models (GLMMs) are used widely in ecological research (Bolker *et al.,* 2009), but less so in fisheries (Venables and Dichmont, 2004a). Nevertheless, the applications of GLMMs are beginning to be explored using catch and effort data (Bishop *et al.* 2004; Helser *et al.,* 2004; Baum and Blanchard, 2010; Tascheri *et al.,* 2010). A GLMM is a generalization of a GLM (Nelder and Wedderburn, 1972), such that the data are permitted to exhibit correlation and non-constant variance (Diggle *et al.*, 2002; Venables and Dichmont, 2004b). A GLMM therefore provides the flexibility of modelling not only the statistical means of data (as in the standard linear model) but also their variance and covariance. The term mixed model refers to the use of both fixed and random effects in the same analysis. The model is described formally as:

$$\eta_i = \sum_{a=1}^{f} \beta_a \chi_{ia} + \sum_{b=1}^{r} Z_{ib} \upsilon_{ib} \ , \tag{5.1}$$

where $\beta_a\chi_{ia}$ are the fixed effects as descriptors of lpue (η_i), and $Z_{ib}\upsilon_{ib}$ are the random effects made up of Z_{ib}, the levels of the random effects, and υ_{ib} is assumed to be distributed normally.

For comparison with the GLMM analyses, a basic GLM with temporal and vessel characteristic fixed effects was constructed as follows:

ln(lpue) ~ vcu + year + month + area + month*year + month*area . (5.2)

Variables were selected initially based on their importance as reported in a pan-European study by Mahévas *et al.* (2011) and their availability from logbooks: final selection was based on their statistical significance at a level of α of 0.05, following stepwise backward selection.

Two other alternative models with the same fixed effects as (2) but with different random effects assumptions were compared using GLMM methodology (1). Alternative regressors of fishing power were considered and for these analyses, vessel tonnage was replaced with VCU, which is highly correlated with the other technical characteristics of the vessel, and 'vessel' was not considered a fixed effect but rather treated as a random effect. Earlier studies explored the use of random effects of vessel and vessel−year interactions when standardizing catch and effort data in examining fishing power (Bishop *et al.,* 2004; Helser *et al.,* 2004). Based on those studies, the same method was applied in the choice of the variable 'vessel' to account for between-vessel variation, and 'vessel and year' to account for vessel variation over time, to capture increase or decrease in fishing power and skipper changes.

The model was developed to capture the variation within vessels and between times, to account for potential technical changes in fishing power over the study period. For example, older vessels in earlier years should have lower fishing power than vessels that joined the fishery later. Residual plots were plotted against predicted values and tested for normality using Q−Q plots. The GLMMs were then compared by inspecting the Akaike information criteria (AIC; Akaike, 1974). All model analysis was implemented by PROC GLIMMIX (SAS Institute Inc. 2006).

5.2.4 Relationship between fishing effort and fishing mortality

The link between F and nominal E can be characterised by the catchability coefficient q (which relates to biomass abundance, and is the fraction of fish caught by a defined unit of effort, see above); catchability also links population biomass abundance N to cpue as cpue = qN.

Following Mahévas *et al.* (2004, 2011), it was assumed that lpue can be represented as

$$\text{lpue} = \frac{\text{landings}}{\text{fishing time}} = aPEN, \tag{5.3}$$

where a represents the accessibility coefficient of the target population, and P the fishing power of the vessel targeting population N applying nominal fishing effort E (in this case hours fished represented by E_n). The product of aP is the catchability. The different factors characterising fishing effort estimated from the model can be used to calculate effective fishing effort E_e by adjusting nominal effort. The relationships between fishing mortality were investigated by plotting log-transformed partial ln(F) against log effort, ln(E_n) and ln(E_e) for all trips in the time period, and the r^2 values compared. Relative nominal and adjusted lpue and effort were calculated based on annual totals and averages of the totals for the period of the study. Fishing efficiency was calculated based on a method used by Marchal *et al.* (2002) and Engelhard (2008), the ratio of relative nominal lpue and survey stock assessment indices from a standard survey vessel that was used consistently throughout the time period of study (ICES, 2007) for each species by comparing start and end estimates weighted by the number of years to give average weighted increase or decrease.

5.2.5 Estimates of fishing mortality

Total international landings and estimated values of fishing mortality were obtained from ICES annual stock assessments (ICES, 2007) for North Sea sole and plaice. Partial fishing mortalities were calculated as

$$F_{ysvta} = F_{ys} \frac{\sum l_{ysvta}}{\sum l_{ys}}. \tag{5.4}$$

The subscripts l, y, s, v, t and a refer to landings, year, stock, vessel, trip and area, so F_{ys} is the total fishing mortality by year and stock (or mean F over selective ages 2−6 (for both stocks), $\sum l_{ys}$ the total international landed weight in kg per year and stock, $\sum l_{ysvta}$ the total landed weight in kg per year, stock, vessel, trip and area, and F_{ysvta} the partial fishing mortality by year, stock, vessel, trip and area.

5.2.6 Investigation of submétiers within a fleet using multivariate techniques

The aim here was to characterize the tactics of a trip based on the effective effort on sole and plaice, in order to give an indication of the operational activities of the vessels (i.e. grouping the vessels into similar subgroups linked to area, season, capacity and ultimately related to approximate fishing mortality) and to use the information as a tool or indicator for managing the mixed fishery. For the present study the Ward minimum variance clustering method was used, in which the distance between two clusters was the ANOVA sum of squares between two clusters added up over all variables (SAS Institute Inc. 1996). This method was preferred because it produces tighter clusters (Gauch, 1982). The Wards minimum variance method tends to join clusters with few observations, and is strongly biased towards producing clusters with roughly the same number of observations. A hierarchical agglomerative clustering (HAC) analysis was used to define subfleets.

5.3 Results

Results from the GLMM and GLM are presented in Table 5.1. Convergence was achieved for all models. The model containing the random effects to account for between-vessel variation and vessel variation between years and vessel variation over time had the lowest AIC and was considered the best model for both species (Pinheiro and Bates, 2000). The plots of residuals against predicted lpue did not show trends and the Q−Q plots followed the reference line, suggesting that the distribution was close to normal and that the correct error models were selected. Furthermore, plots of subject against fitted indicated that all the model outputs tracked the data well, with all values of r^2 >0.56 (Table 5.1).

Using parameter estimates from the descriptors of the GLMM to describe lpue, fishing effort was adjusted. Figure 5.2 shows the log of partial *F* vs. effort relationships (nominal and adjusted) for the two stocks. For sole and plaice, the r^2 values increased from 0.11 to 0.74 and 0.51 to 0.89, respectively, when effort was adjusted by the parameter estimates of the model. The implications of this are that there has been an improvement in the definition and modelling of metrics (effort, capacity and others) that defined the relationship between effort and capacity and *F*.

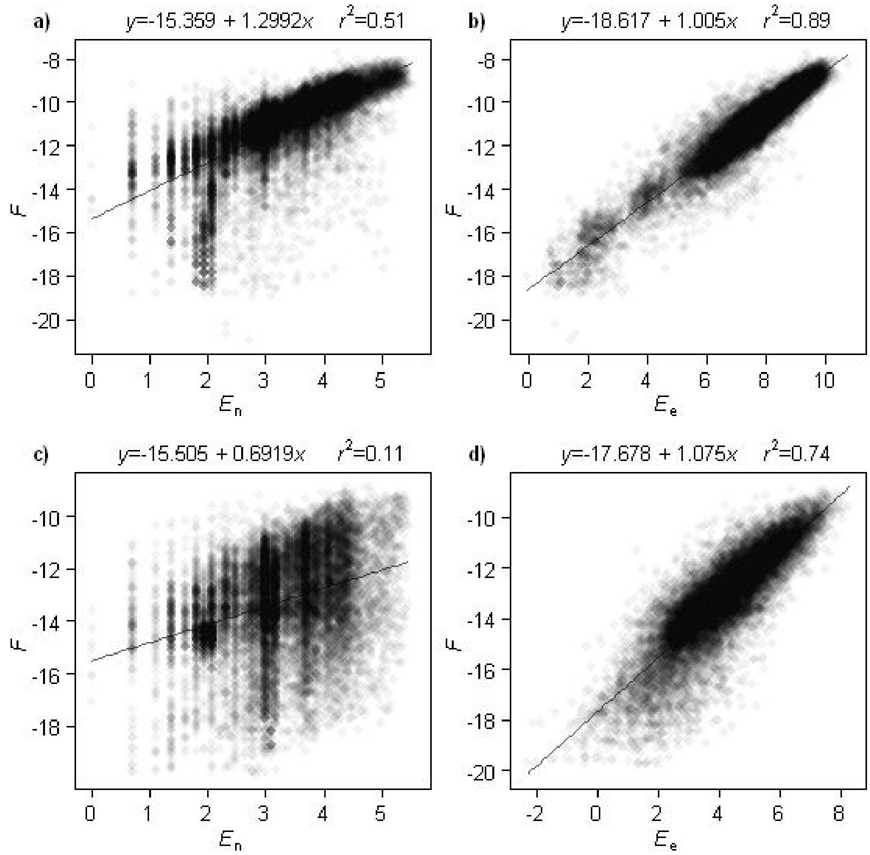


Figure 5.2 Relationships between fishing mortality (*F*) and [(a) and (c)] nominal effort (E_n) and [(b) and (d)] adjusted effort (E_e) for (left panels) plaice and (right panels) sole.

Table 5.1 Diagnostic statistics for the best models explaining plaice and sole lpue as a function of vessel and accessibility (year, month, area) characteristics. The best GLM model (i.e. without random effects) is shown as the basic model (models 1 and 4). GLMMs 2 and 5 have fixed effects equivalent to the basic model but also include the random effects of individual vessels. GLMMs 3 and 6 include the random effects of vessel*year interactions (interpreted as 'technological creep').

Model		**AIC**	**ΔAIC**	**Subject against Fitted (r^2)**	**d.f.**
	Plaice				
	GLM without random effects, and including main effects				
1	vcu + year + month + area + month*year + month*area	49 141.28	9 979.48	0.74	20 263
	GLMM, including random effects				
2	basic + vessel	42 483.61	3 322.09	0.80	20 124
3	basic + vessel + vessel*year	39 161.52	0	0.84	19 701
	Sole				
	GLM without random effects, and including main effects				
4	vcu + year + month + area + month*year + month*area	64 029.54	5 510.08	0.57	20 263
	GLMM, including random effects				
5	basic + vessel	60 032.64	1 513.18	0.62	20 124
6	basic + vessel + vessel*year	58 519.46	0	0.66	19 701

5.3.1 Fishing efficiency and year effects

Trends in effort (nominal and adjusted) and lpue (nominal and adjusted) over the study period (1997–2007) for the various stock/fleet combinations are displayed on Figure 5.3. For both stocks, there was a downward trend over time in both nominal and adjusted effort, but this trend appears to have stabilized for the final three years of the analysis. In terms of lpue, there was no trend for plaice, but there was an increase for sole over the final 5 years of the study period. Analysis of the percentage change in fishing efficiency resulted in an annual 6.2% decrease for plaice and a 0.6% increase for sole. These results coincided with the transfer of ownership to the Netherlands, where skippers generally target sole because of its greater commercial value and availability relatively close to port in the southern North Sea, vessels generally operating in Roundfish area 6 (Figure 5.1).

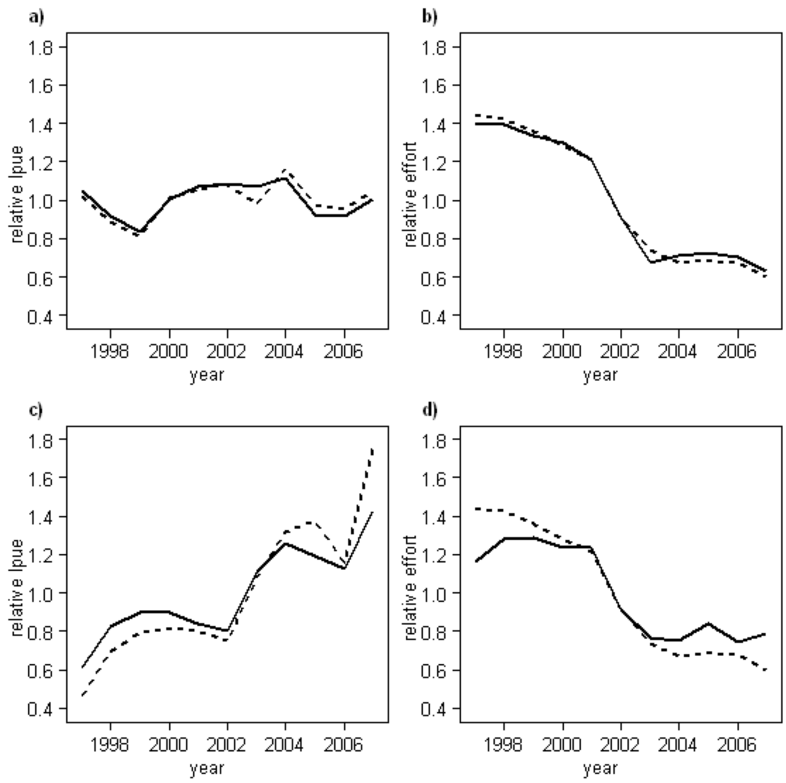


Figure 5.3 Relative [(a) and (c)] landings per unit effort, lpue, and [(b) and (d)] relative effort for (top panels) plaice and (bottom panels) sole for the English beam trawl fishery in the North Sea (1997–2007), with data for both nominal effort (E_n) (dashed line) and adjusted effort (E_e) (solid line) indicated.

5.3.2 Cluster analysis

The results of the Cluster Analysis pseudo F and cubic clustering criterion (ccc; SAS Institute Inc. 1983; not shown) revealed local peaks at three clusters, reinforced by a local low t^2 and a levelling of R^2 for these clusters, indicating three distinct submétiers (Figure 5.4). Exploratory analyses (Figures 5.5−5.7) showed interesting spatial and temporal patterns. Clusters 1 and 2, although close spatially (Figure 5.5), were distinguished seasonally (Figure 5.6) in terms of a decrease in effective effort on sole during the second quarter of the year for cluster 1. Cluster 3 was distinct, being mainly a sole fishery just off the English coast fished mainly by inshore vessels of smaller capacity (VCU).

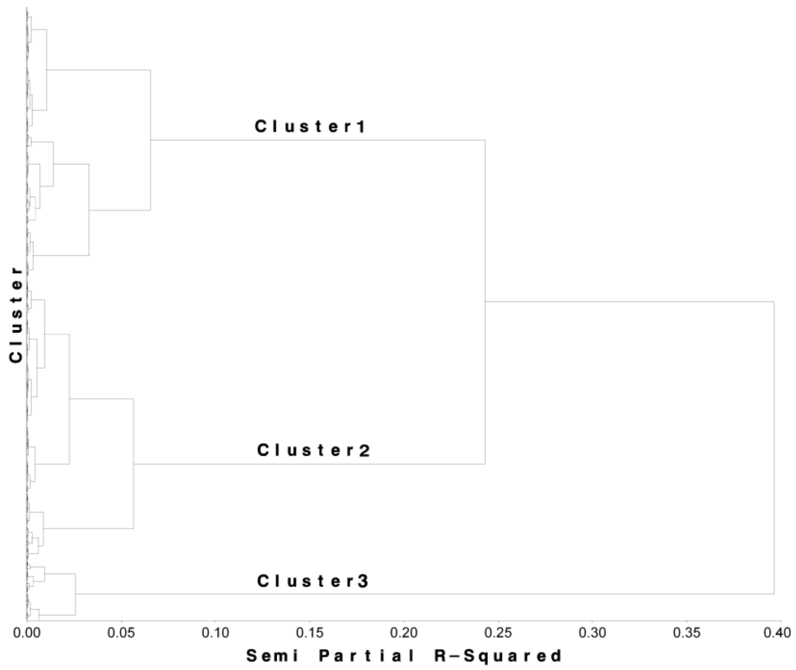


Figure 5.4 Dendrogram of the beam trawl fishing trips in the North Sea, based on effective effort profiles for sole and plaice.

5.3.3 Application of the analyses

As a demonstration of utility of the analysis in terms of management indicators, the effects of reducing fishing mortality on both stocks for a given reduction in mortality on one stock were estimated. Taking into account the relationships between effective effort and fishing mortality

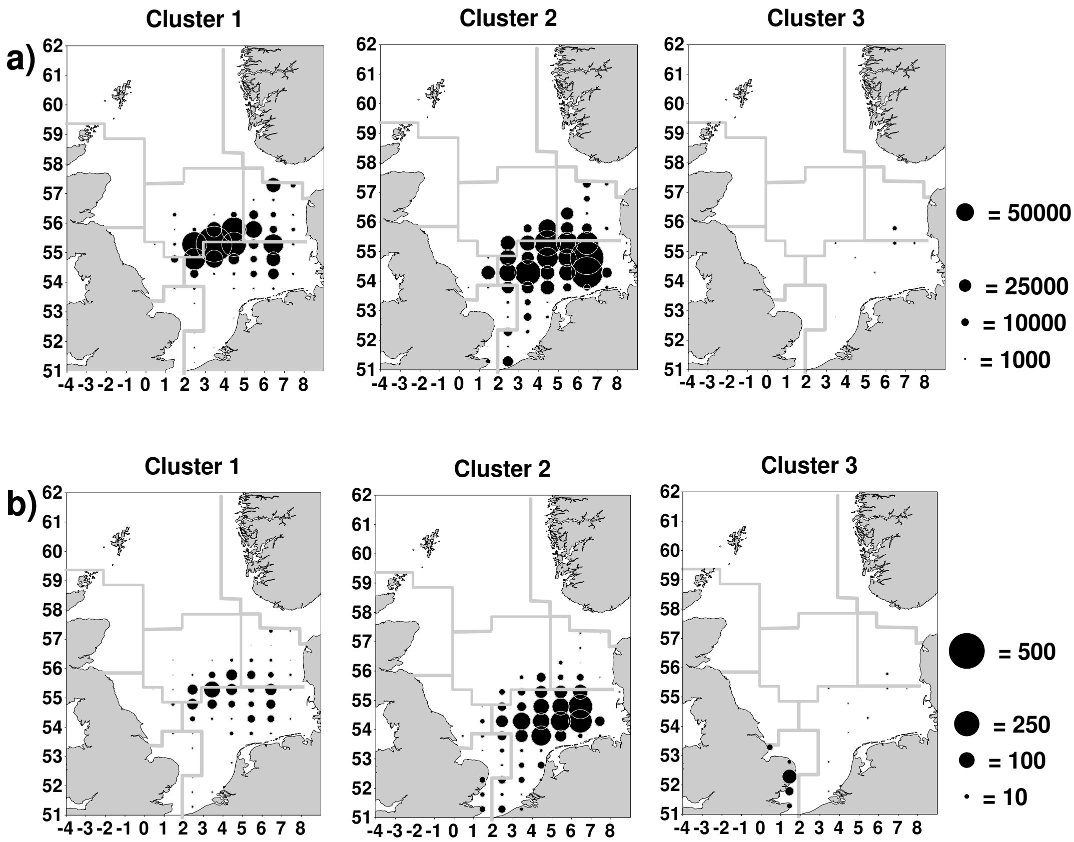


Figure 5.5 Total effective effort of (a) plaice and (b) sole by cluster for 2007.

for each submétier/cluster and the trends for each cluster over time, for each gear grouping and area, a simple management approach is presented to demonstrate application of the approach. Using the values produced from the cluster analysis, Figures 5.6 and 5.7 display where the main effort is in terms of trip numbers and effective effort by area and season. For example, if a manager wishes to reduce fishing mortality on plaice by 20% in the 1st quarter of the year in Roundfish area 6 (or in rectangles in this area) for cluster 2 (Figure 5.7a) and vessels with a VCU of 800−1099 (Figure 5.7b), the effective indicators provide a platform to control fishing mortality by reducing the hours fished. An example is described below.

Step 1 − Taking the example from above, in 2007 there were ~120 trips (Figure 5.7) exerting an average effective effort of 8 (Figure 5.6; plaice effective effort). A fishing efficiency decrease of 6.2% is applied in order to estimate the effective effort, which results in a new effective effort of 7.94 (e.g. exp (8) × 93.8%, then back-transformed).

Step 2 − Applying the effective effort from Step 1 (Figure 5.2b, using the equation from the plots) results in a fishing mortality on plaice of −10.641 in total, equating to exp (−10.641)*120 trips) and an estimate of *F* of 0.002869.

Step 3 − A 20% reduction results in an *F* value of 0.002295. The average per trip log-transformed gives an *F* value of −10.864, which results in an effective effort of 7.71 (Figure 5.2b) and a nominal effort of 3.46 (Figure 5.2a). The nominal effort back-transformed approximates to 32 h per trip, an overall reduction of 6 nominal hours fishing per trip per vessel from the original calculated nominal effort of 38 h based on −10.641 fishing mortality (Figure 5.2a).

Step 4 − To provide an indication of the effect on sole for a 20% reduction in plaice, a ratio of the start and end estimates of effective effort of plaice as calculated in the steps above (7.71/7.94) was applied to the mean effective effort on sole (Figure 5.6; sole effective effort 5.5, including 0.6% fishing efficiency increase), which was estimated at 5.73 and the associated *F* calculated to be −11.52 (Figure 5.2d). Applying the ratio, the resulting effective effort was 5.57 and the revised *F* −11.69, giving a total reduction of 15.8% in sole mortality and a total reduction of 720 h fishing based on a 6 h reduction *120 trips.

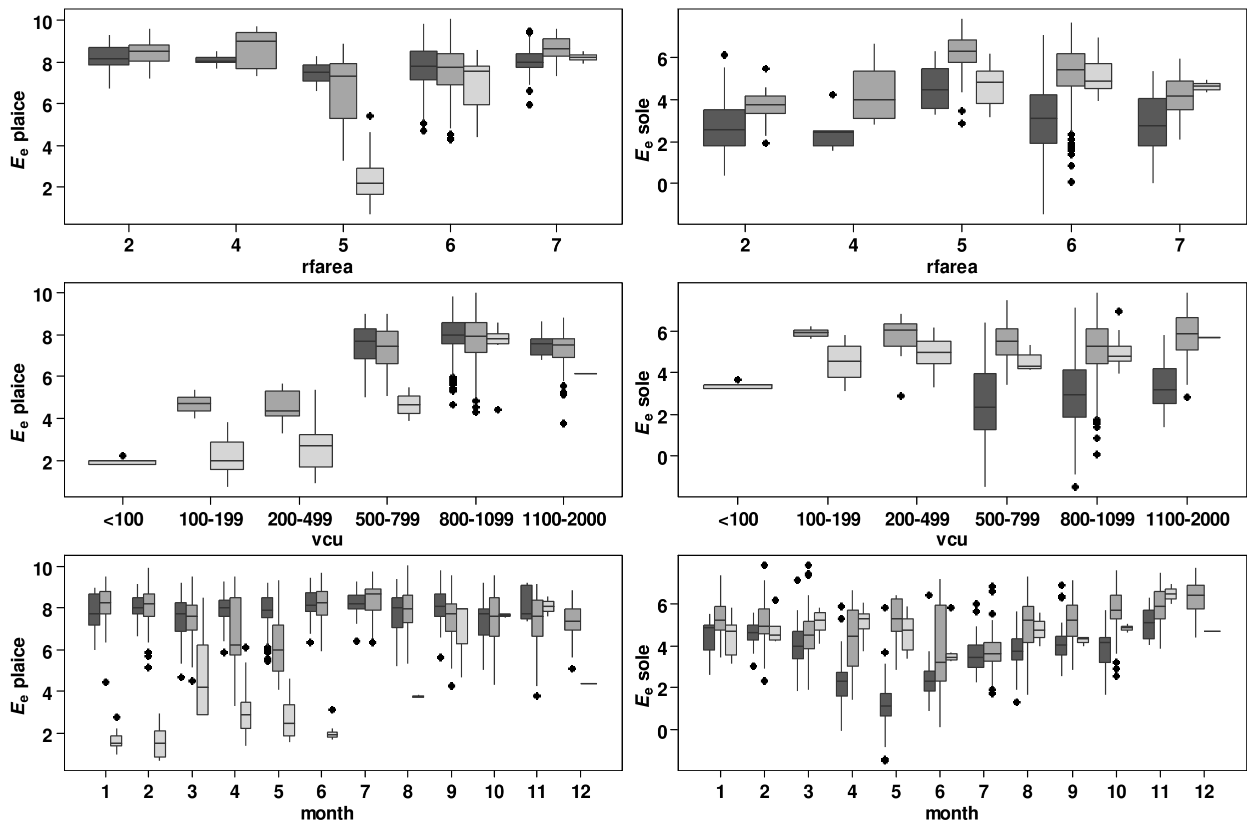


Figure 5.6 Effective effort indicators. Box and whisker plots of clusters vs. the different covariates (roundfish area, VCU and month) for plaice (left panels) and sole (right panels) in 2007. The horizontal line represents the mean, the box the 25th−75th percentiles, the whiskers the ranges of data, and the solid diamonds outliers, with Roundfish areas (rfarea) displayed because they demonstrate the approach more clearly than a series of ICES rectangles. (Cluster 1=dark, Cluster 2 =medium dark, Cluster 3= light).

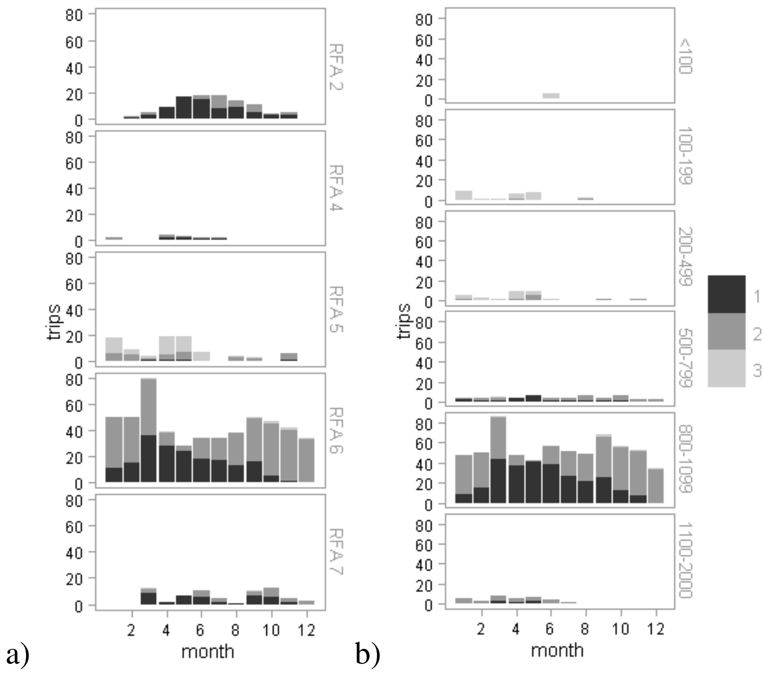


Figure 5.7 Number of trips by (a) Roundfish area (RFA) and (b) VCU by cluster (1−3) and month for 2007.

5.4 Discussion

The analysis here has provided an understanding of the relationships between some of the parameters that allow linkages to be drawn between capacity, effort and fishing mortality and of their use as indicators for spatial and temporal management of the North Sea flatfish beam trawl fishery. It also takes account of changes in capacity and fishing power. Limiting fishing through effort controls via spatial management requires an understanding of likely fisher response, and also an ability to predict the choice of fishing area or fishing activity (Vermard *et al.,* 2008; Chapter 4). Here, no attempt was made to predict the choice of fishing ground, but on the basis of fisheries seasonality, Chapter 4 provided a simplistic ecosystem approach (FAO, 2003) to manage a fleet's activity in a particular area (Daan, 2005), targeting sole and plaice. Bycatch species were not included in the model because of the lack of estimates of fishing mortality, nor were benthic habitats of conservation interest included.

A GLMM that included random effects (in this case, vessel) was applied to lpue as the dependent variable in order to explain the variance attributable to targeting by the gear, changes in efficiency, capacity, seasonal and area effects. This method was selected over the more traditional GLM because of the unbalanced dataset, i.e. not all vessels operated throughout the study period. As such, including the vessel as a random effect takes account of inter-vessel variation and variation between individual vessels over time; ignoring it could produce negatively biased lpue estimates. The model parameter estimates for sole and plaice were adjusted with nominal effort and fitted against *F*. Both adjustments resulted in improved relationships relative to *F* vs. nominal effort. Relative adjusted effort over the study period declined initially for both species, but stabilised towards the end of the study period, whereas relative adjusted lpue improved slightly for sole and increased the fishing efficiency for this species. Cluster analysis of individual trips, based on estimates of effective effort for sole and plaice, revealed three main submétiers within the fleet, which then made it possible to estimate spatially the effect on one stock of applying an effort or fishing mortality limit (including fishing efficiency).

The model relied on estimates of *F* from ICES working groups. If the *F* estimate was biased there would be variances in the F~E_e relationship. Landings are not always a direct proxy for fishing mortality, because of discarding, however, and discarding was not taken into account in these analyses because the information was not available for all fleet segments/submétiers. The quality of other data sources (e.g. VCUs derived from the fleet register), and the collection and databases of logbook information, cannot be assessed. The results from the *F*-reduction exercise underscore the difficulties in controlling fishing effort when managing a mixed fishery, because the nominal effort vs. mortality relationship for sole had a poor fit (Figure 5.2c). The analysis relied heavily on the plaice fit (Figure 5.2a), which provided a better indication of nominal effort exerted at a trip level. The effective effort indicators were based on means (Figure 5.6), although they showed the relative uncertainty or spread of the data associated with respect to each factor. However, such a spread of data for each factor is not uncommon, because fisher behaviour varies and leads to different values of effective effort. Managers applying effort limitation need to be aware of the variability in catchability by fishers in the same fishery acting on the same stock group.

The seasonal nature of the fishery was evident from the analyses (Figure 5.7). There was typically more effort at the start and end of the year in Roundfish area 6 for cluster 2, reflecting targeting of plaice then and corresponding to the seasonal migration of the fish from the central North Sea (Roundfish area 2) to the southern spawning grounds (Roundfish area 6; De Veen, 1978; Rijnsdorp and Pastoors, 1995; Hunter *et al.*, 2003), and greater effort in Roundfish area 5 in late spring and summer, possibly reflecting beam trawling for sole on their spawning grounds near the English east coast (Cluster 3; De Veen, 1976). Cluster 1 (in contrast to Cluster 2) was characterised by more effort farther north in Roundfish area 2 throughout summer, but this was not as prominent at the start or end of the year. The results support the findings of earlier studies of clear seasonal trends in beam trawl effort redistribution throughout the study period (Chapter 4).

One of the main assumptions here was that fishing VCU was a proxy for capacity, the rationale being that the unit is the basis of vessel-reduction programmes (Multi Annual Guidance Programmes; UK Fisheries Department 1988) in the UK. Vessel landing rates, i.e. nominal lpue values, were calculated as catch in kg per h fishing per vessel per trip per area. The importance of making management decisions on effort measured in hours, in theory, may provide a less crude measure that relates closely to actual fishing activity rather than the current days-at-sea restrictions applied to North Sea fleets. However, the current regulations are expressed in days-at-sea to simplify the process in terms of enforcement. Irrespective of potential changes in fishing tactics to maximise number of hours fished, increases in efficiency are evident for one stock (sole), whereas decreases in efficiency for plaice could indicate increased targeting of sole (Figure 5.3). More importantly, the slope of the regression in each case increased (see Figure 5.2). In practice, this implies that management that considers several factors (capacity, seasonal and area effects) that contribute to effective effort should be more effective in reducing fishing mortality than management based purely on nominal effort. The policy implications are such that adjusting effort such as days at sea (or h-at-sea) by capacity (and taking into account month and area effects) should result in greater than proportional decreases in fishing mortality. How viable it would be to adjust for such an approach through regulation and enforcement requires more study. Changes in catchability that arise when applying additional nominal effort or fishing efficiency are important to fisheries scientists, to monitor changes in F, and likewise, for a given F, the effective effort will be influenced by fishing efficiency and the nominal effort will need to be adjusted appropriately.

A key finding from the study was the switch in targeting and the changed fishing efficiency, an estimated 6.2% decrease in plaice and an estimated 0.6% increase in sole annually for averages calculated over the 11-year study period. The decrease in plaice efficiency is interesting because the concept of negative creep is becoming more evident especially as fuel prices rise. Increasing fuel costs in beam trawling (Abernethy *et al.,* 2010; Chapter 3) may well have influenced the distribution of the fleet in the southern North Sea, with less steaming time to ports in the Netherlands reducing operating costs to counteract fuel price increases. English beam trawlers generally target both plaice and sole, but in recent years, because of the shrinking fleet size and transfer of ownership to the Netherlands, sole has generally been targeted because of its greater commercial value and short distance from port in the southern North Sea, also perhaps contributing to the increase in efficiency and the decrease in catches of species targeted previously (Marchal *et al.,* 2003; Engelhard, 2008). Measures in 2007 to protect juvenile cod as part of the cod recovery plan were imposed on certain beam trawl gears; an 8% reduction in effort from 2006 was enforced, and this could have also contributed to the fleet fishing closer to port and the switch in target species (EC, 2006b, c).

Limiting and reducing the time a vessel spends fishing is possible in theory, but out of sight of regulatory enforcement, it used to be difficult to control. With the application of Vessel Monitoring Systems (VMS), however, it has become possible for regulatory authorities to monitor the activities and locations of commercial fishing vessels better, although there remain problems in identifying activity and there are anyway limitations in VMS data use (e.g. the time between satellite pings that monitor the vessels; data being collected only on vessels ≥15 m long within the UK; accurate matches with landings data by trip and ICES rectangle; and for scientific studies, confidentiality), which is why days-at-sea effort restrictions have been preferred in EU waters.

A spatial means of effort control to reduce fishing mortality and discards on cod and to encourage compliance introduced by the Scottish Government in 2008 after consultation with stakeholders was that of real time closures (RTCs). Fishers were rewarded with extra days at sea for avoiding areas where the lpue of cod was high. Currently, the threshold for enforcing a RTC is 40 cod per h fished; one catch exceeding this threshold triggers a closure. Early studies by Needle and Catarino (2011) using VMS data showed that vessels tended to move away from RTCs, but also that vessels returned to these areas shortly after the closure ended.

Overall, the conclusion on RTCs was that mortality on cod was reduced, but not sufficiently to influence future exploitation patterns. One can argue about the effectiveness of RTCs because they do not control effort, but rather displace it, so it is difficult to evaluate their effectiveness in the short term. Moreover, any benefits from RTCs may be partly negated by the increased days at sea allocated to participating vessels. On the positive side, the measures were developed with input from stakeholders, and compliance with respect to RTCs via VMS data was encouraging, with vessels moving away from the boundaries of the closed areas. With the emergence of electronic logbook data and closed circuit TV (CCTV) for on-board surveillance, monitoring of catches may improve and create a more-level playing field across sectors of the wider fishing industry. Other recent studies linking catches and effort in mixed demersal fisheries in the EU fisheries include Fcube, the Fleet and Fisheries Forecast (Maravelias *et al.*, 2011; Ulrich *et al.*, 2011). This useful application attempts to promote fleet and métier management to progress from the traditional single-species approach for routine advisory use. Ulrich *et al.* (2011) concluded that the current single-species management for North Sea cod (*Gadus morhua*) could not be achieved unless TACs and effort reduction for other species were applied. However, this study differed from Fcube by accounting for changes in fishing power, so can be applied at a finer regional scale.

This study has shown clear applications for input control for mixed fisheries management and has also complemented other research initiatives, such as recent catch quota trials (FVM 2009) undertaken by the UK, Denmark and Germany using remote electronic monitoring (REM). The inclusion of REM, personal information on skippers (Kirkley *et al.*, 1998; Squires and Kirkley, 1999), information on gear and technological changes (Marchal *et al.,* 2006) and precise time actually fishing should lead to more detailed estimates of effective fishing effort and relationships with fishing mortality at a finer resolution than the ICES rectangle. It will also be important for future studies to take account of other factors, ranging from non-target fish and wider ecosystem impacts to the social and economic implications of effort controls and their impacts on the different submétiers. The movement away from single-species management to the fleet-based management approach applying temporal, spatial and gear-specific control measures under the guidance of the DCF (Data Collection Framework) and future CFP could be used to evaluate alternative management strategies in conjunction with stakeholders, so could facilitate implementation and improve fisheries management, including perhaps fairer access to resources.

Chapter 6. Fishing for space

Scallops *Pecten maximus*

Abstract

Since 2008, the European Union has had objectives for spatial planning and regulation to deal with increasing human activities and pressures at sea. Integrating spatial planning with existing fisheries regulations has been difficult because of the spatial scale at which landings are reported and the fear among practitioners of conceding space to competing activities. To determine the extent that spatial competition influences choice of fishing grounds, a discrete choice model was applied to fine spatial resolution data obtained from the Vessel Monitoring System (VMS). We analyse the determinants of English and Welsh scallop-dredging fleet behaviour, including competing sectors operating in the eastern English Channel. Results show that aggregate activity and maritime traffic negatively impact the choice of fishers, and conversely that past success, expected revenues and fishing within the 12 nautical mile limit have a positive effect on their utility. The model has potential application for Marine Spatial Planning (MSP).

6.1 Introduction

As human pressures increase there is a need to balance competing demands for the natural resources that society is challenged to manage and conserve for future generations. Experience has shown that once humans have fully exploited a resource on land they look for alternatives at sea. The sea, traditionally seen as a common property resource, is confronted increasingly with competition for space by competing sectors, e.g. fisheries, oil and gas exploitation, aggregate extraction, wind energy, shipping and transport, recreation, dumping and the military. The spatial planning and regulation of the increasing human activities and pressures at sea are therefore becoming a concern, especially given that some resources are limited in space and quantity. If the limited resources are poorly regulated, there may be a race to exploit them, a situation commonly known as the “Tragedy of the Commons”

(Hardin, 1968).

Since 2008, the European Union has had objectives that place a responsibility on member states to achieve common principles. It is called the “Roadmap for spatial planning” (EC, 2008a) and falls under the Integrated Maritime Policy (IMP; EC, 2007), and is now generally referred to as Maritime Spatial Planning (MSP). The objectives of MSP are to manage anthropogenic activities in space and time, precluding or minimising conflicts between competing sectors without negatively impacting the ecosystem, operating within the Marine Strategy Framework Directive (MFSD; EC, 2008b) and covering human activities. MSP is therefore an integrated marine management plan to alleviate conflict and balance ecological, social and economic demands to achieve Good Environmental Status (GES) in EU waters. However, because sectors at sea can change rapidly and the complexities of natural systems are linked and inter-reliant, a management decision for one may affect others, and MSP needs to be treated as a process of continuous, adaptive management. Uncertainty associated with compliance to management measures and thus its effectiveness has been linked to lack of knowledge on the motivations associated with people. Traditional fisheries management treats fishers as fixed components with no consideration of their behaviour in terms of attitudes to fishing (i.e. spatial, temporal, social, ecological and economic) and individual aims (Salas and Gaertner, 2004; McKelvey 1983; Smith and McKelvey, 1986).

The EU’s Common Fisheries Policy (CFP) recognised the importance of these factors (EC, 2009a) and is now committed to both an ecosystem approach and more regional approach, whereby fleets and fisheries and their interactions are to be managed within smaller regional areas rather than the broad ecoregions used in the past. Given the importance of MSP, several writers have stressed the relevance of designing fleet-based spatial management in the commercial fisheries sector (Botsford *et al.*, 2007; Kraus *et al.*, 2009; Bastardie *et al.*, 2010), accounting for different fleet activities at a scale fine enough to feed into the MFSD. To date, integration has been difficult owing to the broad scale (ICES statistical rectangle ~900 nautical miles^2) at which some data (e.g. landings) are reported. With the emergence of Vessel Monitoring Systems (VMS) over the past decade, however, MSP is now potentially possible at a finer scale. Issues of data confidentiality between member states have hampered the process, though, and there is also a historic reluctance of fishers to provide accurate landings information for fear of conceding knowledge of profitable fishing grounds (NSRAC, 2005). Degnbol and Wilson (2008) suggested that fishers are concerned about data confidentiality, especially how the data they provide would be used and by which authority. For example, they especially raise concerns regarding how the data would be used against them by conservationists, as the data could potentially be used to identify productive fishing grounds as suitable for Marine Protected Areas (MPAs) or by fisheries managers to implement tighter enforcement constraints. In the light of the limited data availability and confidentiality, fisheries managers are looking now for alternative approaches to assist spatial planning, which will reduce implementation error i.e. where the effects of management differ from that intended (Peterman, 2004). One such approach involves anticipating fisher behaviour in response to regulation. Recent studies have applied random utility model (RUM) methodology (Vermard *et al.*, 2008; Andersen *et al.*, 2010; Chapter 4) to this issue, because such models offer an opportunity to study individual behaviour at a finer scale of space and time than previous approaches (Coglan *et al.*, 2004). Fisher behaviour cannot be predicted with certainty because of the many factors (see above) which influence where and when a fisher will operate. If managers can better anticipate fisher behaviour, then they may be able to reduce the unanticipated side-effects of management actions aimed both at the fishery sector and at other sectors.

The objective of the present study was to analyse and model the key determinants of where fishers choose to fish, building on retrospective time-series and including competition between a selection of key sectors of activity and understanding their interaction to these activities. The focus was the English and Welsh scallop-dredging fleet operating in the eastern English Channel (ICES Division VIId). That area also contains one of the busiest shipping lanes in the world, the route between the Atlantic Ocean and the North Sea, and there is a traffic separation scheme (TSS) in operation with 100 vessels in and 100 vessels out per day. It is perceived that such a concentration of vessels would have a negative impact on commercial fishers.

There are also several active marine aggregate extraction sites and fishers have expressed concerns about the accumulation of marine aggregate sites where licences are permitted and the effect of fishing pressure concentrating itself elsewhere for fear of gear damage and the sustainability of fish stocks (Cooper, 2005). In terms of fishing restrictions in the eastern Channel there is a 12 nautical mile belt of territorial water surrounding the base coastline that is sovereign waters, also local bylaws restrict beam trawlers of 300 hp or 70 grt from this area and as such restrict competition with the inshore fleet fishing for sole (Figure 6.1). This ruling also prevents fishing by any international fishing vessel, though the area can be used for safe passage. Most of the vessels operating in the region are small (<10 m) inshore boats that deploy gillnets, trawls, longlines, traps and pots, and target sole (*Solea solea*), plaice (*Pleuronectes platessa*), cod (*Gadus morhua*), bass (*Dicentrarchus labrax*) and some skates (Rajidae; Pawson, 1995).

A mixed RUM was developed to analyse the determinants of fisher behaviour at a fine scale (a trade-off between ICES rectangle and individual position) using English and Welsh VMS data, highlighting the effect of the key potential competing sectors on fishing behaviour. Suggestions are then made as to how the method can be used in integrated MSP in anticipation of the potential establishment of Special Areas of Conservation (SACs) in the area as part of UK commitments to the EU's Habitats Directive (EC, 1992).

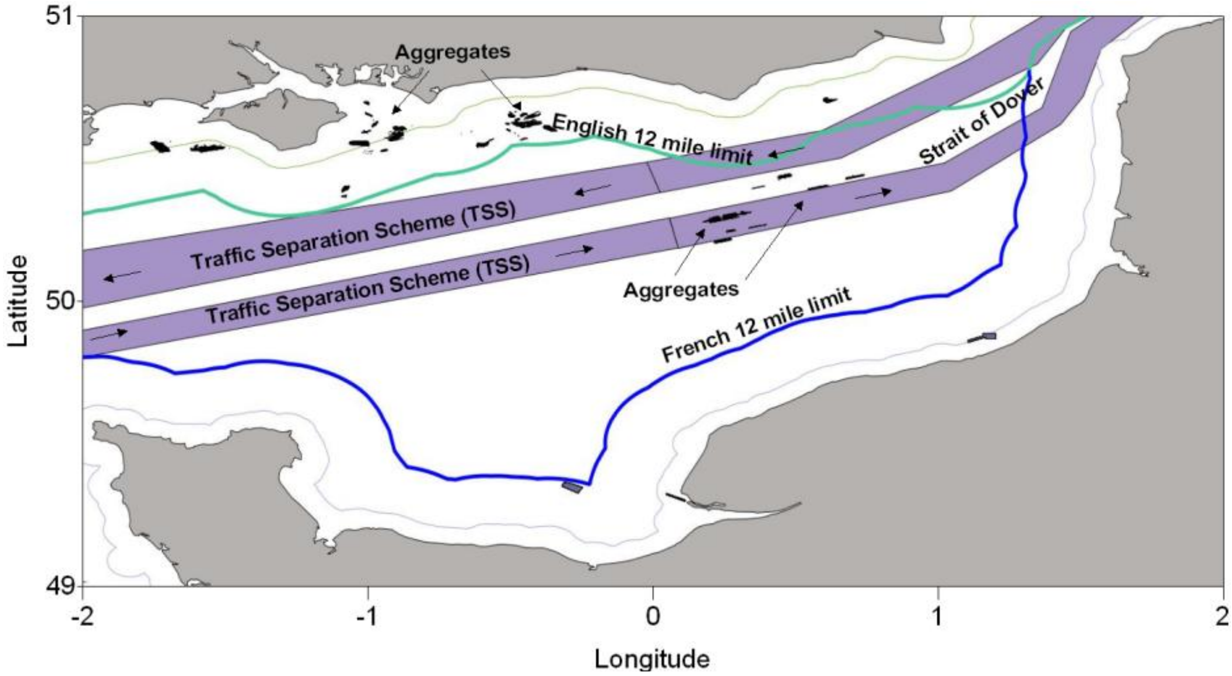


Figure 6.1. Competition among sectors within the English Channel.

6.2 Materials and methods

6.2.1The UK scallop fleet

The UK scallop (*Pecten maximus*) industry is one of the UKs most valuable fisheries and was valued at >£47 million (£13 million in the eastern English Channel) in 2009, employing >13000 people in the catching sector and 17 000 in the processing sector (Defra, 2011). Scallops are fished in one of three ways, dredging, trawling and hand-diving. Dredging is the most common method and consists of deploying a heavy metal frame with a chain mesh and a set of spring-loaded teeth pointed downwards to assist in raking out the scallops into the dredge's chain mesh. These dredges are connected to a beam, which in turn is connected to heavy warps that are towed over the seabed by a fishing vessel.

The UK scallop-dredging fleet is said to be nomadic in nature, moving around the UK coast to fish where scallop abundance is best and operating there until those grounds become economically non-viable. They then apparently move to other areas and only return to fished-out areas a few years later when stocks there have recovered (Defra, 2011). The eastern English Channel was traditionally a winter fishery because, following spawning in early summer, the scallops were in poor condition so unmarketable. In recent years, however, there has been an increasing trend in the number of vessels operating in this fishery as fishers in other fisheries have had to confront changes in regional management (e.g. more restricted fishing opportunities elsewhere, such as Cardigan Bay), market conditions and subsequently changed their own tactics and strategies in order simply to survive (Mangi *et al.,* 2011). This statement also applies particularly to UK whitefish vessels, for which economic performance has been hit by rising fuel costs and hence high running costs (Curtis *et al.*, 2006).

Scallop fishing is less fuel-intensive (in terms of search behaviour of the fleet) because fishers are chasing a high-value, stationary stock rather than one that is moving continually. There is also the additional pressure for summer fishing grounds for vessels to use, because the Irish Sea fishery is closed from June to October. This notwithstanding, there is discussion of a summer closed season in the Channel being imposed, as is the case in France. A further pressure over the past six or so years has been changes in Scottish fisheries which led to their largest scallopers (14 per side) being banned from Scottish waters, meaning that they can now work only south of the Scottish border (Howell *et al.* 2006).

Defra (2011) suggest that there has been a noticeable increase in fishing effort in the eastern

English Channel from 2008 to 2010 and this is predominantly from the larger ≥15 m long more powerful vessels due to the increase in scallop abundance resulting from heavy recruitment. The variability in landings resulting from fluctuations in recruitment, market demand, regulations and more recently fuel price are common features of scallop fisheries. Historically the consequences of which include variability in the number of vessels participating in the fishery due to there being no restrictions on licences or total scallop catches. In 1999 the number of vessels was at a high so regulatory authorities attempted to cap licences on vessels (≥10 m) (Brand, 2006). However it has been suggested that it had little impact on the fishing effort as there were more licences granted than there were boats fishing in the fishery (Brand, 2006). Nevertheless there are periods of temporal inactivity when stock abundance is low and the fleet move to other fishing grounds (Beukers-Stewart and Beukers-Stewart, 2009). Generally current management of scallop fisheries are controlled through minimum landing sizes and the numbers of dredges regulated by local sea fisheries committees as there are no catch limitations.

6.2.2 Data

The UK's Department for the Environment, Food and Rural Affairs (Defra) database for fishing activity and the fleet register were used to select commercial landing and vessel data from the English and Welsh fleet (excluding Scottish and Northern Irish due to confidentiality issues). Individual trip data for commercial scallopers were collated for the years 2005–2010. The data collected for each vessel included species landed, hours fished, landed weight per ICES statistical rectangle (kg), month of fishing, year of fishing and total value of the catch by species, vessel and trip. Within the EU, it is currently only a requirement for vessels >10 m long to submit logbooks, but the database also contains a subset of catch from <10 m vessels that historically reported their catches by means of logbooks.

Methodology for the definition of fleets was based on the European Commission's Data Collection Regulation (DCR; EC, 2000). A method was developed independently (see EC, 2006a), preceding the present Data Collection Framework (DCF; EC, 2008c), which defines the scallop-dredging fleet on the basis of its use of a scallop dredge for >50% of a fishing trip. It is assumed that dredge catches consist mainly of molluscs and that their tactics/métiers can be defined based on the proportional composition of mollusc value to the total value of landings, so removing the differences in catch rates attributable to vessel capacity.

VMS monitoring in the European Union (EC, 2003, 2009b) has been in place since 2000, initially for fishing vessels of ≥24 m long, post-2005 for vessels ≥15 m long, and in 2012 ≥12 m long. The data are designed to help regulatory authorities determine whether a vessel is rule-breaking by receiving a ping every 2 h giving position, course and speed. However it is not totally clear from VMS data whether the vessel is in port, steaming to and from fishing grounds, hauling, shooting or fishing. Over the past few years, authors such as Mills *et al.* (2007), Lee *et al.* (2010) and Hintzen *et al.* (2012) have described methods to determine fishing or steaming activities from unprocessed VMS data, methods that include removal of erroneous data, e.g. positions on land, unusually high speeds, positions close to port and duplicate records. No individual method in the scientific literature has been adopted as the definitive process or preferable to another, however, but for ease and accessibility, the data for the years 2005–2010 were processed in the manner described by Lee *et al.* (2010). Logbook data and VMS fishing records were selected, combined by vessel and ICES rectangle between departure and arrival dates, forming a detailed dataset of fishing activity. The ICES rectangle was further formatted into 200 (3' × 3')-pixel squares coded from 000 to 199, starting from top left and moving to bottom right, placing all the coordinates from the

VMS data into the pixels.

Marine diesel prices, excluding value-added tax (VAT) and duty, were obtained from the Department of Energy and Climate Change (DECC). Aggregate-extraction intensity data by month for the years 2005–2010 were obtained from the UK's Royal Haskoning and the French l'Institut Français de Recherche pour l'Exploitation de la Mer, Ifremer. In terms of shipping/transport, however, such data were not available at the time of the analysis, so marine traffic-separation zone data, obtained courtesy of Ifremer, and were used as a surrogate. Finally, UK and French 12-mile limits were added to the maritime activities dataset because it was considered that competition for space with the local inshore fleet would be an influencing factor. Having populated the dataset with all covariates, the dataset could be used in a mixed RUM to determine the key determinants of fisher behaviour in relation to the key competing sectors of activity as well as fishing specific covariates. It is likely that key competing sectors of activity as well as costs (i.e. fuel price) will negatively impact fishing specific operations (Figure 6.2), in contrast to expected revenues and past effort (knowledge or habit) largely influencing fishing operations. The scale of the analysis and variables selected are descried below.

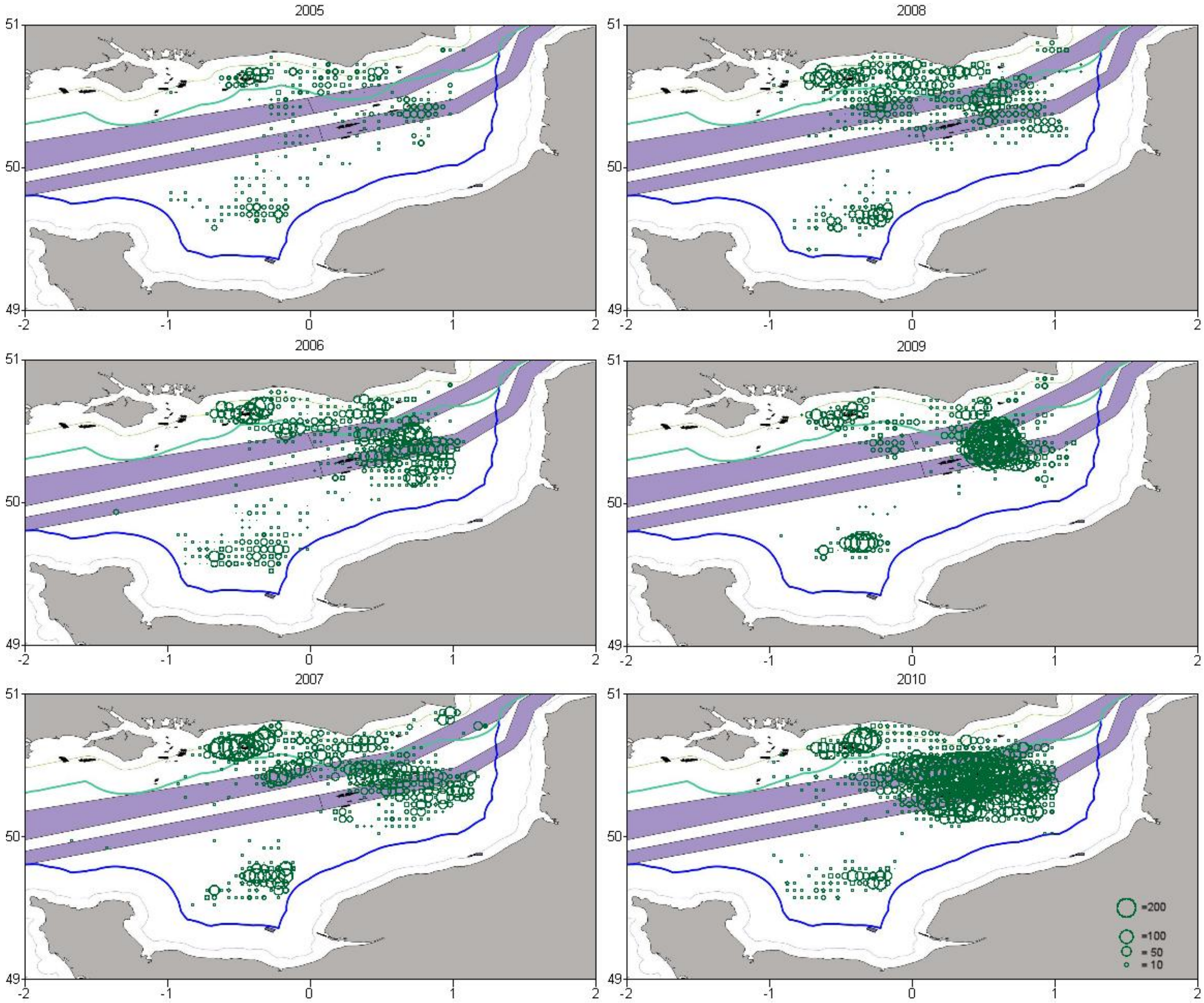


Figure 6.2. The eastern English Channel displaying scallop dredging effort in hours fished represented by green circles. (See Figure 6.1 for other activities).

6.2.3 The model

In keeping with the work of Chapter 4 describing the dynamics and drivers of fisher location choice, a mixed logit choice RUM was implemented because it relaxes the non-IIA (Independence of Irrelevant Alternatives) assumptions associated with preference heterogeneity among fishers. This approach is efficient in dealing with panel data for repeated individual choices, as is the case within this study. For a detailed explanation of mixed logit, see Hensher and Greene (2003) and Train (2003). Succinctly, the total utility μ_{njt} of fisher n for site j in trip t is

$$\mu_{njt} = \bar{\beta}x_{njt} + \sigma_n x_{njt} + \epsilon_{njt} . \tag{6.1}$$

where $\bar{\beta}x_{njt}$ represents the observed utility and $\sigma_n x_{njt}$ the unobserved utility due to heterogeneity, and ϵ_{njt} is the error distribution that is part-correlated and part independently and identically distributed (iid) over alternatives and individuals (McFadden, 1981; Maddala, 1983). The mean $\bar{\beta}$ plus its standard deviation σ_n are used to represent the preference distribution in the population of fishers (Train, 1998). All covariates met the normality assumption following log-transformation. Within the mixed logit framework, β_n was assumed to follow a normal distribution, and for a given value of n (for simplicity disregarding *t*), the conditional probability of choice *j* across all other choices k = 1 to *J* is estimated by drawing random values β by simulation using

$$P_n(j) = \frac{\exp(\beta x_{nj})}{\sum_{k=1}^{J} \exp(\beta x_{nk})} \quad , \tag{6.2}$$

where β is a vector of coefficients that varies across individuals, and x_{nj} is a vector of the attributes of each of the choices made. Cost data enter the model with a negative sign and revenues with a positive sign, as suggested in the economic literature (Train, 1998; Ran *et al.*, 2011). The analysis was carried out in the SAS package PROC MDC (SAS, 1999 Not in References) using quasi-Newton optimisation and 100 Halton draws.

6.2.4 The definition of choice set

When designing RUMs, fisheries scientists are confronted with the problem of creating a choice set, which covers the individual sites to which a fisher travels to fish. If sites are too small (individual latitude/longitude positions), there may not be sufficient site-specific information, but if they are too large, important site-specific information can be lost when aggregating, losing information valuable to policy-makers. Handling many variables with zero data in the choice set may cause problems of maximum likelihood estimation and result in model non-convergence.

Fishers have prior knowledge of resource distribution and habitat (Hilborn and Ledbetter, 1979; Gillis *et al.*, 1993; Pet-Soede *et al.*, 2001), and scallops are relatively static molluscs, suggesting that in future years, any choice set will be subject to relatively little or no change. On the basis of this assumption, the predetermined area making up the choice set for this study was based on the 2005–2010 effort distribution of scallop dredgers plotted from the VMS records (Figure 6.2). The investigative plots displayed effort coverage over a large area within the small number of ICES statistical rectangles (as previously ICES rectangles were considered too large for spatial planning purposes with pixels too finite). Therefore a trade-off was necessary and the pixels were grouped (25 pixels) into 8 subrectangles based on area (the 15' × 15' rectangles also used by Ifremer's Channel Groundfish Survey, CGFS). These areas were georeferenced into 45 subrectangles, so determining the choice set (Figure 6.3).

6.2.5 Variable selection

As with other economic/fisher behaviour studies, data on the costs of fishing trips are not always available because of the time and cost taken to collect such information, and the information is also likely to be confidential. Researchers therefore use a proxy of value per unit effort (vpue) rather than cost, which relates to the utility/net benefit of variations in stock density (Marchal *et al.*, 2007; Vermard *et al.*, 2008). Value per choice was calculated as a proportion of the total value (revenues from landings) per ICES rectangle based on effort derived from the VMS, and vpue was then computable. The average vpue by year and month and location choice was calculated for the fleet and lagged in two ways: lagged vpue for a particular month in year t = $-m$; lagged annual vpue in year t–1 = m_{t-1}, i.e. taking account of strong or weak temporal and spatial fluctuations. Habit, knowledge and experience of fishing locations influence fisher behaviour (Begossi, 2001). The past percentage of a particular vessel's scallop trips to a fishing location as a percentage of the fleet total elsewhere was used as the habit/experience variable and to track the seasonal nature of the fishery, as in Holland and Sutinen (1999). These variables were lagged in the same method as explained above.

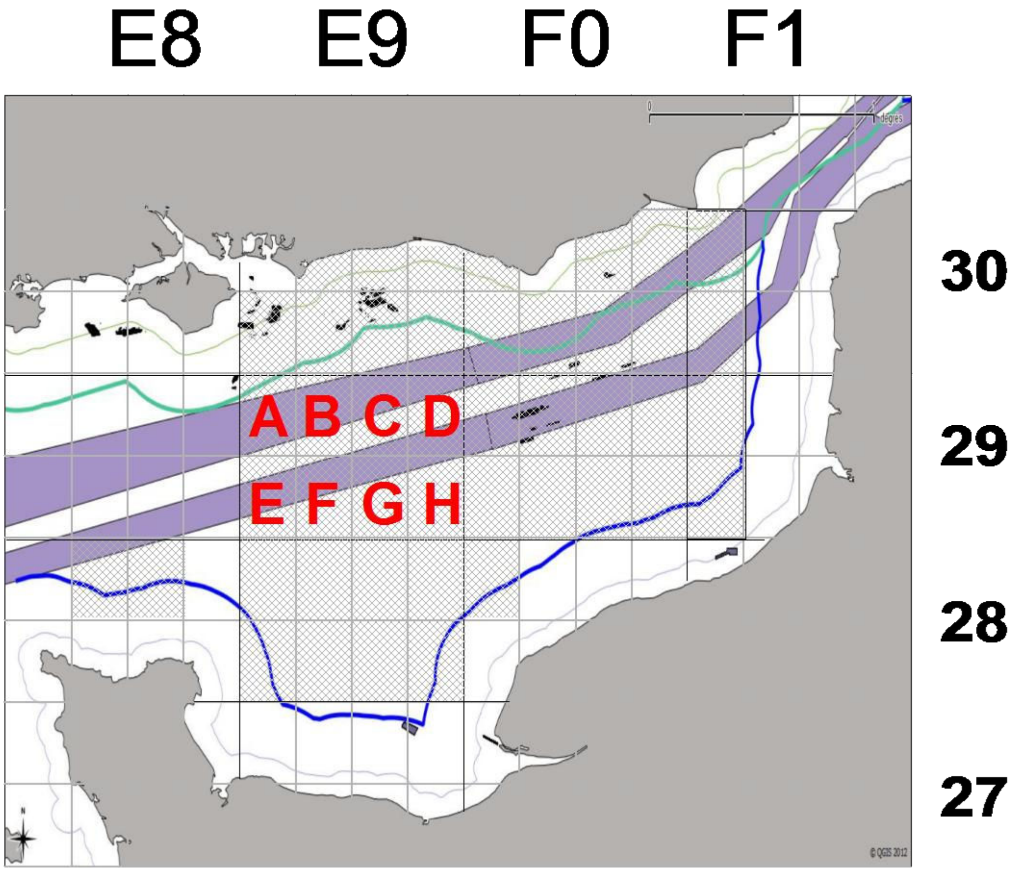


Figure 6.3. The eastern English Channel with ICES rectangles overlaid and the choice set represented by the hatching geo-referenced by ICES rectangle and the eight sub-rectangles within.

Fishers are assumed to maximise their returns (Robinson and Pascoe, 1997), so depending on weather and other factors, they trade off travel costs against the quality of the fishing grounds. An increase in distance linearly relates to an increase in fuel costs and hence time and energy, so removes the potential for participating in other activities, e.g. fishing closer to shore or non-fishing activities (Daw, 2008). Therefore, in terms of accounting for the expected travel costs and the landing behaviour of the fleet, a proxy for cost was calculated based on the average fleet distance to landing port from VMS fishing locations, calculated using the Haversine formula (Sinnott, 1984), weighted by mean average fuel price, lagged by a month in each year of fishing as a measure of perceived costs. Landing port was used because of the nomadic behaviour of this fleet; it was assumed that the fishers would have prior knowledge of seasonal market prices in the proximity of fishing locations.

Aggregate activity enters the model as the average percentage coverage per choice by year and month, but because of inconsistencies between French commercial aggregate data expressed at a daily scale and English intensities at a monthly scale, daily scale records could not be used. The traffic separation zone data and the 12-mile limit (as a proxy for the inshore fleet and internationally restricted zone) were treated as a spatial constraint (as above). One might assume that the greater the percentage coverage of a restriction, the greater the negative impact on site choice and that site preference would then be elsewhere due to activities that would be a nuisance to fishing. The variable selection set described above was merged with individual scallop trip data by year, month and choice, such that for every trip, the decision-maker had a choice of the specified 45-subrectangles. If the choice was made, the values took a value of 1 if selected or 0 otherwise. The definitions of the variables are listed in Table 6.1.

Table 6.1. Definition of variables used in the RUM to model fisher location choice for the 45 ICES sub-rectangles in the eastern English Channel as defined in Figure 6.3.

Variable	Definition
effyr	Percentage of trips to the location in the same month the previous year (taking account of trips by the scallop fleet fishing in other areas outside of the eastern English channel
vpueyr	Average vpue of scallop from fishing in the same location in the same month in the previous year.
vpuem	Average vpue of scallop from fishing in the same location the previous month in the actual year of fishing.
effm	Percentage of trips to the location in the previous month in actual year of fishing. (taking account of trips by the scallop fleet fishing in other areas outside of the eastern English channel).
cost	Average distance to port of landing from the same location the previous month in the actual year of fishing multiplied by the fuel price.
aggregate	Average % coverage of area occupied by aggregate activity.
traffic	Average % coverage of area occupied by marine traffic lanes.
limit	Average % coverage of area occupied by 12 mile limit.

6.3 Results

The results from the mixed model showed a McFadden's pseudo-R^2 of 0.21, suggesting an excellent fit (McFadden, 1979). Theoretically, the range for McFadden's pseudo-R^2 is between 0 and 1, but the general rule of thumb is that any value from 0.2 to 0.4 suggests an excellent fit as shown in an earlier study by Domenich and Mcfadden (1975) in which they compared ordinary least squares (OLS) R^2 of 0.7–0.9 with the above pseudo-R^2 range.

Pseudo-R^2 method differs from a traditional R^2 where the parameter estimates were not calculated to minimise variance via (OLS) goodness of fit process, instead they are calculated via maximum likelihood iterative process and the low values between 0.2–0.4 are considered to be acceptable (McFadden, 1979). The goodness of fit determined by the likelihood ratio test was also conducted on the constrained model (log-likelihood under the null hypothesis) fits against an unconstrained model, the resulting p-value of <0.05 and likelihood ratio of 4833.5 demonstrating that the mixed model was better in terms of likelihood compared to the conditional model.

Observations from the parameter estimates showed some key features, in terms of significance and direction of the signs. Holland and Sutinen (1999) suggested that the direction of the sign of the coefficient in terms of profit or revenue is an indicator of average risk preference in terms of variability, suggesting as an example that if fish aggregations are not present at certain times of the year, fishers would not go to an area; as such there would be an increase in variability in profit or revenue and the coefficient would be negative. Conversely one may view a positive sign and a small coefficient of variation as showing that fishers are risk-averse and fish in locations of past success or experience.

The estimated coefficients from the mixed model on the 3019 observations available are presented in Table 6.2. All coefficients were statistically significant ($p < 0.01$) except the coefficient for percentage of trips to the location in the same month the previous year (effyr_M, $p > 0.1$) and the average distance to port of landing from the same location the previous month in the actual year of fishing weighted by the fuel price which was marginally significant (cost_M, $p < 0.1$). The estimated standard deviations of the estimates were not significantly different from the mean indicating that the parameters do not vary in the population of fishers for past expected revenues (vpueyr_S, vpuem_S), percentage of trips to the location in the same month the previous year (effyr_S), average distance to port of landing from the same location the previous month in the actual year of fishing weighted by the fuel price (cost_S) and average percentage coverage of area occupied by marine traffic (traffic_S). Conversely, the percentage of trips to the location in the previous month in the actual year of fishing (effm_S), the average percentage coverage of area occupied by aggregate activity (aggregate_S) and the average percentage coverage of area occupied by marine protected area or 12-mile limit (limit_S) did not vary, perhaps related to variations in characteristics of the fishers not captured in the model. The signs of the standard deviations in some instances are negative, but for estimation purposes they are free to take any sign, because the normal distribution is symmetrical around its mean, and the absolute value can be taken to estimate the variance.

The effort distribution maps in Figure 6.2 show the interactions of the scallop dredges with the traffic separation scheme (TSS) and the aggregates and fisheries within the 12-mile limit. Coupled with the model outputs, these results display some notable features. In general the mean coefficients show the signs one would expect (Table 6.2). The positive sign on the coefficient for the 12 mile limit (limit_M) shows that fishers benefit from fishing within this zone as there are several hotspots with high scallop catch rates (Figure 6.2). Conversely the negative signs on the mean coefficients for aggregates (aggregate_M) and the TSS (traffic_M) imply that these sectors impede fishing operations. However, in every year of the study there was a large amount of fishing effort in these areas, even more so in 2010 within the TSS. Perhaps that result is a trade-off in terms of larger expected revenues in these areas. Expected revenues (vpueyr_M and vpuem_M) show positive signs which clearly demonstrate that revenue has a significant influence on the tactics of fishers in contrast to the cost proxy (cost_M) which was negative as expected. Past effort variables (effm_M and effyr_M), which were included to depict habit or knowledge of past success of fishing grounds, have positive coefficients, suggesting they are important drivers in determining fisher location choice.

Table 6.2. Estimated lognormal parameter estimates, the dependent variable took a value of 1 if a choice was made or 0 otherwise.

Parameter	**d.f**	**Estimate**	**Standard Error**	
vpueyr_M	1	0.0467	0.0114	***
vpueyr_S	1	–0.0006	0.2878	
vpuem_M	1	0.098	0.0222	***
vpuem_S	1	0.00384	0.2737	
effyr_M	1	0.0535	0.0301	*
effyr_S	1	0.0184	0.6357	
cost_M	1	–0.0294	0.0161	*
cost_S	1	0.00397	0.1623	
effm_M	1	0.7134	0.0309	***
effm_S	1	–0.2527	0.0884	***
aggregate_M	1	–0.0957	0.013	***
aggregate_S	1	0.2023	0.0528	***
limit_M	1	0.7528	0.1184	***
limit_S	1	–0.7213	0.1031	***
traffic_M	1	–0.1405	0.0465	***
traffic_S	1	0.00126	0.5333	

* Statistical significance at *, 10% level, **, 5% level, and ***, 1% level.
Parameters marked _M are the lognormal mean coefficients and _S are their between-population standard deviations.

To test the sensitivity to different variables the mean choice probabilities were calculated from the model output and then compared with mean choice probabilities after re-running the model under alternative scenarios where each variable was doubled/halved one at a time. The differences in probability of location choice, under each of these scenarios, show that the magnitude of the effect on location choice (Figures 6.3–6.5) and how sensitive the variables are to changes i.e. how the variables that penalise fishing operations (e.g. aggregate extraction, marine traffic, and fuel costs) affect fishers, in contrast to expected revenue which should encourage fishing operations.

In terms of aggregate extraction, fishers responded to a decrease (halving the coefficient, i.e. half the level from –0.0957 to –0.04785) in % area covered which resulted in a difference in probability of +0.019, +0.012, and +0.011 in the areas associated with aggregate extraction, 30E9G, 29F0C, and 30E9F, respectively (Figures 6.4–6.5). Doubling the effect, increasing the size of the site resulted in fishers moving out of the areas of aggregate extraction, notably to 30E9G, 30E9F and 29F0C with a change of probability of –0.019, –0.012 and –0.012 respectively. There was a small increase in probability into 29F0D, the adjacent sub-rectangle to 29F0C. Most of the main scallop grounds are in marine traffic areas and therefore one would expect that with a decrease in area occupied by traffic lanes there would be less competition for space and fishers would move into these areas.

Maritime traffic, however, surprisingly showed little effect, apart from in 30E9G (an area that contains aggregate sites and a small section of traffic lane). In this sub-rectangle, doubling the coefficient of maritime traffic resulted in fishing effort being displaced out of the area (probability reduction of –0.015), whereas halving the coefficient led to an increase in predicted effort in 30E9G (+0.006). An explanation could be the risk adverse nature of this fleet, as 30E9G is an inshore sub-rectangle (steady amount of fishing effort throughout the time series, Figure 6.2) and despite being relatively close to land (i.e. less distance to travel to land their catch and hence less fuel consumption), vessels may seek to reduce spatial competion with increased disruption from maritime traffic. However, expected fuel cost did not show large significant differences in probabilities of site choice when increased or decreased. Figure 6.4 and Figure 6.5 suggest that with a halving of the fuel price, fishers move to areas where the concentration of fishers and expected revenue is at its highest (areas 29F0A, 29F0B, 29F0C and 29F0D), resulting in a trade-off with expected costs and expected revenue (net benefits).

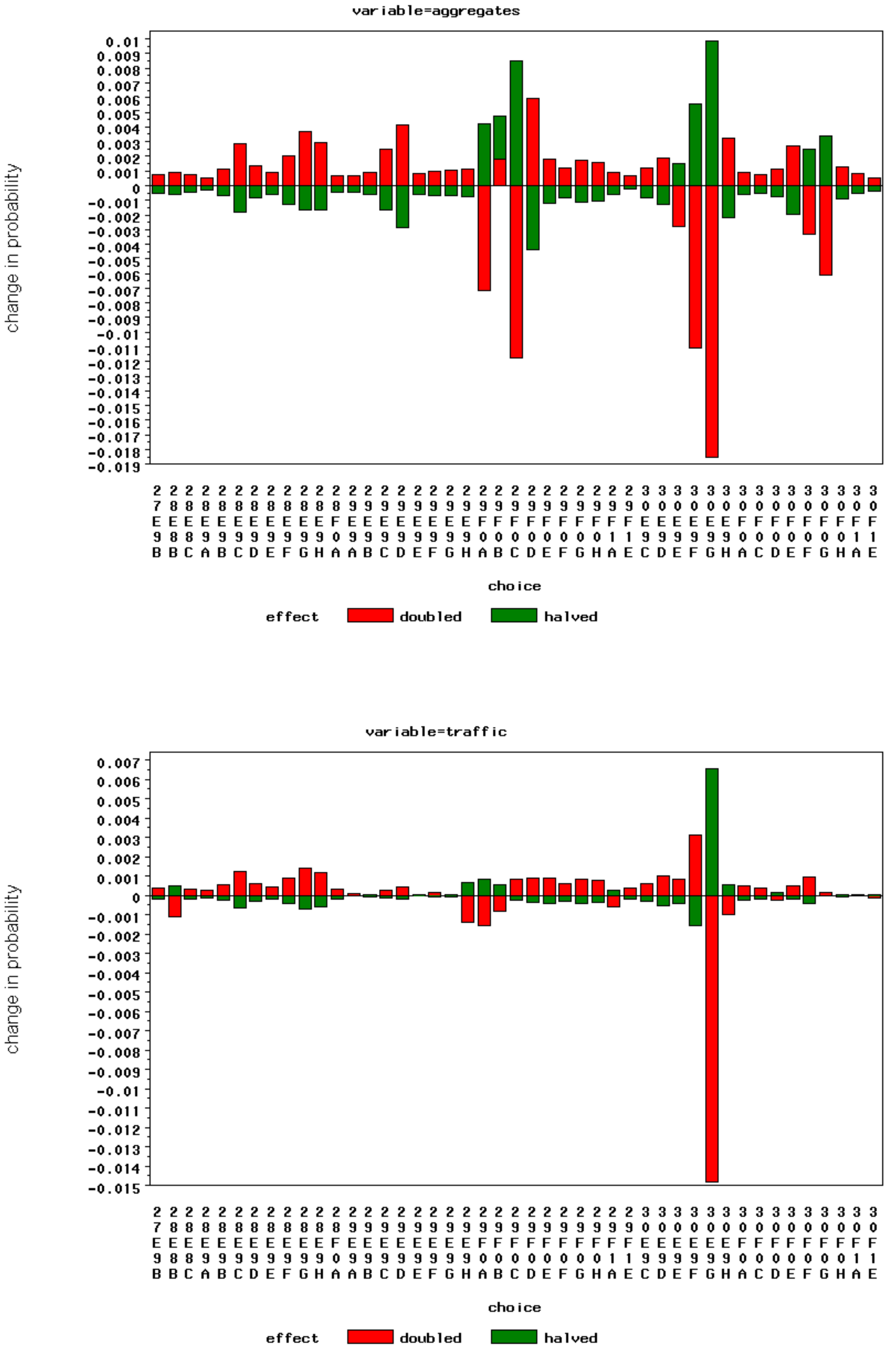
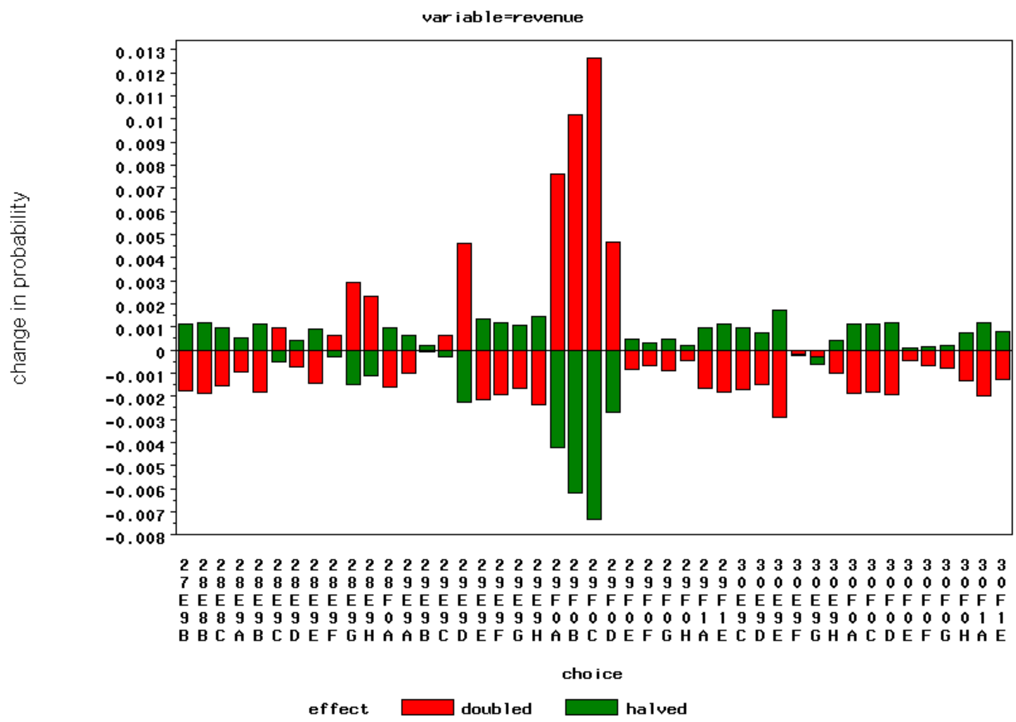


Figure 6.4. The difference in mean choice (sub-rectangles) probabilities from the benchmark model and one under alternative conditions (twice the level (doubled) and half the level (halved), for a selection of the variables)



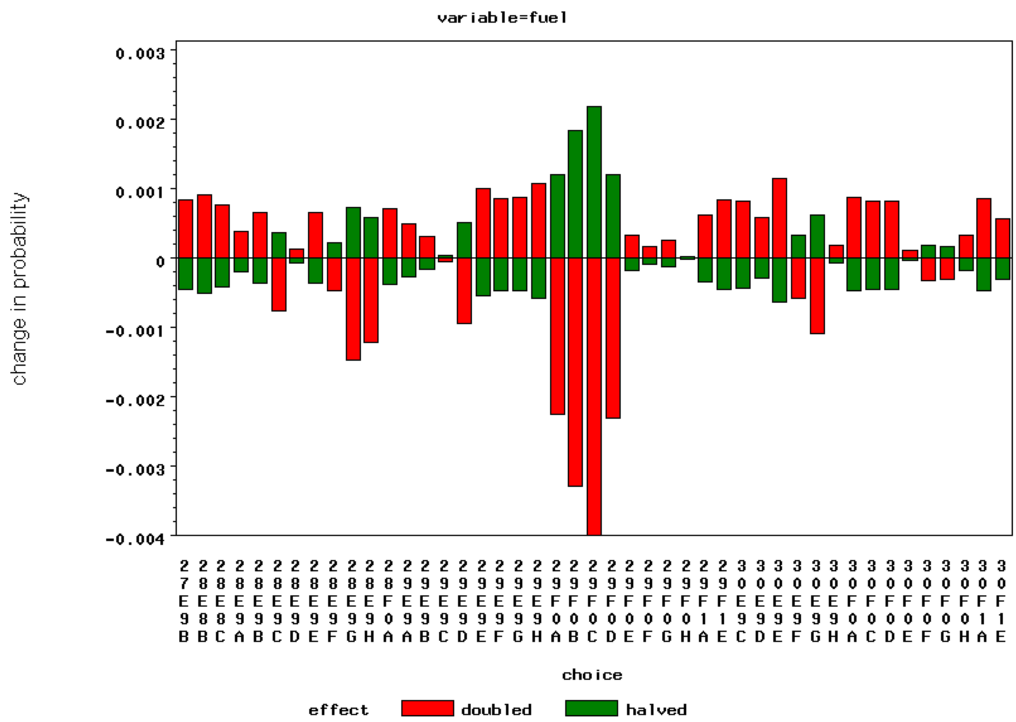


Figure 6.4 (continued). The difference in mean choice (sub-rectangles) probabilities from the benchmark model and one under alternative conditions (twice the level (doubled) and half the level (halved), for a selection of the variables).

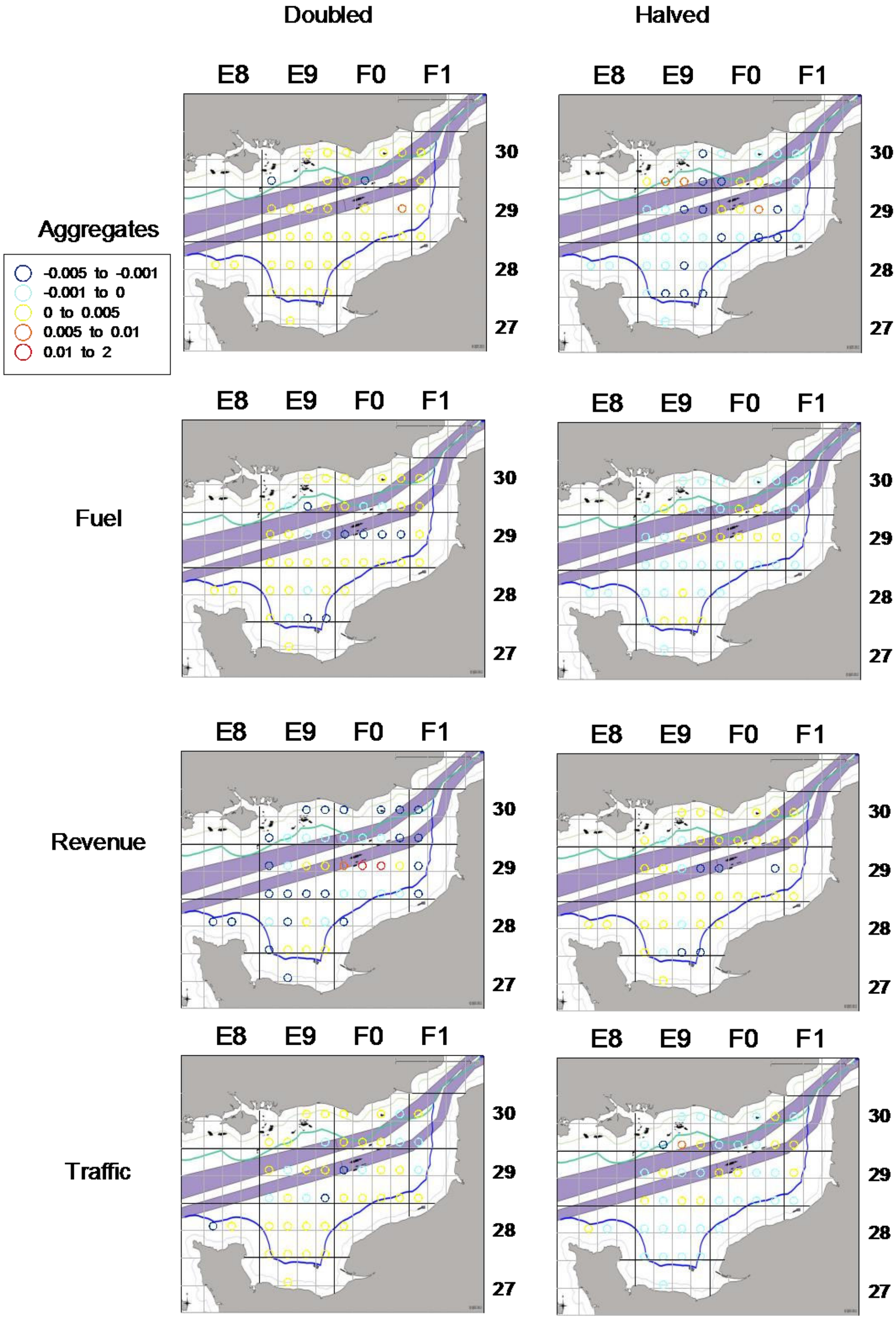


Figure 6.5 Changes in probabilities when halving or doubling the effects of each variable in contrast to the benchmark model.

6.4 Discussion

It is widely recognised that decision-makers and managers now require an ecosystem-based approach to address current interlinked problems for social well-being (FAO, 2003b). Since the Earth Summit in Rio de Janeiro in 1992 there have been pressures from environmental organisations, increased public and political interest and a concurrent implementation of directives and policies to improve management of human activities on a regional basis by different stakeholders. MSP requires the balancing of multiple objectives, e.g. fisheries managers need to understand the implications of effort displacement from closing an area and the unforeseen consequences of their management actions (e.g. effects on other marine life, economic implications and effects on other maritime sectors).

Several authors have stressed the importance of anticipating fisher behaviour in response to management regulation, in order to reduce implementation error (Dugan and Davis, 1993; Allison *et al.*, 1998; Fulton *et al.*, 2011). Here, a mixed RUM was applied at fine-scale resolution to assess the key determinants of scallop fisher behaviour in the eastern English Channel, so that if a regulation or new activity, emerging pressures as well as potential hazards were present, fishing effort re-allocation could potentially be predicted.

A key finding was that past success in a location within the previous month was a predictor of continued fishing in that location. I interpret this as a proxy for habit, knowledge or experience as in other studies (Holland and Sutinen, 1999; Salas and Gaertner, 2004; Andersen and Christensen, 2006). Similarly, the expected utility of visiting one fishing site rather than another in terms of marginal revenue, expressed as vpue, was significant as expected (Ran *et al.*, 2011). This is more apparent for the vpue in the previous month, rather than in the same month the year before, potentially capturing either seasonality or relatively short term temporal correlations in stock abundance (see Table 6.2). Surprisingly, perceived fuel costs were not a major driver in choice of fishing grounds, possibly because of the proximity of grounds to landing ports in the eastern English Channel. The location being inside the 12-mile limits was strongly positively correlated with it being chosen, possibly because grounds within these limits are close to landing ports. This may also perhaps explain the weak significance of the fuel-cost coefficient.

Competition for space with other <15 m vessels does not seem to affect this fleet (the 12-mile limit as a proxy for the inshore fleet which protects the inshore fleet from large beam trawls and international fishers), and the scallop fleet does benefit from the exclusion of an international fleet that is banned from operating within the 12-mile limits and as such have less competition. Nevertheless competition from the national fleet could become an issue if the fleet was squeezed into a small enough space, for example by spatial closures. Of policy importance are the effects of the commercial marine environment and associated maritime activities on the behaviour of the scallop fleet; if these are better understood then the additions of other sectors or addition of other potential aggregate site plans and their implications to this fleet can be assessed in terms of potential effort re-allocation. The areas occupied by aggregate extraction sites are less chosen than expected from their other attributes, confirming the assumption that the aggregate industry does impact scallop fishing, which takes place in large areas where aggregate licences have been granted since 2005 (Vanstaen *et al.*, 2007). This is contrary to Desprez and Lafite's (2012) findings for sole, which suggests that aggregate extraction can have a positive effect on the catchability of sole by beam trawlers and hence on profitability. Perhaps, increased turbidity increases sole catchability (by reducing visual cues for escape and/or fish being disturbed from the seabed) or the dispersal of food into the water column encourages sole to move away from the bottom to feed.

The existence of the TSS in one of the busiest shipping lanes in the world is a management attempt to alleviate maritime accidents which can also impede fishing. The output from the model suggests that the presence of a TSS significantly reduces the probability of a fisher choosing a location, suggesting that the policy is having the desired effect of separating fishing from other activities, though at the cost of reduced ability to choose areas of potential high profitability. Nowadays, policy makers require information on predictions of potential shares of each alternative chosen by the fishers, and the analysis shows that changing a particular preference parameter it is possible to calculate choice probabilities under alternative policies. For example, an increase in aggregate activity and the likely choices of fishers in response to this, or a levy on fuel price and the likely effects of effort displacement would have a high chance of displacing effort to local inshore waters (Figure 6.4). The results from the sensitivity analysis (Figure 6.4) show that the fleet trades off lower fuel cost by going further off shore with the expectation of the reward of higher returns, and when costs are higher they fish more inshore.

The fleet is also affected by maritime traffic, fishing further inshore under increased traffic and surprisingly moving into one specific area out of the way of any potential dangers. This may be because the majority of the traffic lanes are home to the main scallop fishing grounds and the specific location they relocate to inshore has the next best expected catch rates and lower costs. This is also apparent for the competition with the aggregate sites, which are located in the heart of scallop fishing grounds. Any reduction of the space taken up by aggregate extraction, especially inshore, shows an increase in effort allocation to those locations. An important point from Figure 6.5 is that if one of the parameters that disadvantages fishers (e.g. increasing the traffic lanes – doubling the effect) is altered, then effectively the competition for space increases and the fishery spreads out, and as such fishers ‘fish for space’. This could mean that a reduction in the total space occupied by the vessels could be interpreted as a direct measure of competition within the fleet as well as a response to other sectors. Further investigation would be necessary to prove or disprove this theory, along with the inclusion of international fishing fleets. Overall, the model describes the nomadic behaviour of the fleet, i.e. in-year behaviour with respect to habit, expected revenue, proximity to landing ports and competition from other maritime sectors.

6.5 Conclusions and future work

The Eastern English Channel is a shared resource and there is increasing competition for space and new challenges for novel management approaches by understanding all or some of the interactions between sectors. In parallel to this work progress is being made on several dynamic processes (e.g. larvae distribution, consequences of aggregate extraction on benthic communities and fishing interactions) that will be implemented into a bio – economic mixed fishery model and a complex ecosystem holistic model using the ATLANTIS (http://www.csiro.au/en/Organisation-Structure/Divisions/Marine--Atmospheric-Research/Atlantis-ecosystem-model.aspx) framework as part of EU VECTORS project (http://www.marine-vectors.eu/). Different management strategies can be performed and their outcomes assessed.

To my knowledge, no other study has used a mixed RUM at fine resolution to assess key determinants of human behaviour in relation to different maritime sectors and as a possible tool for MSP. The results are promising and lay the foundations for future work which could include including Marine Conservation Zones (MCZs) and using information on the movement of shipping traffic from Automatic Identification System (AIS) data which include vessel position and movement. Final decisions on where MCZs will be enforced in the

English Channel are still work in progress, so it was not appropriate here to incorporate simulated closure and effort displacement evaluated using Equation (6.2). Nevertheless, the principle outlined and the approach taken could already be applied to other fleets, as RUMs offer the capacity to model individual behaviour at fine spatial and temporal scales, which is needed for policy decisions (Smith, 2002). Further work could include evaluating trade-offs with both socio-economic and conservation objectives using efficient and effective spatial planning tools such as Marxan and MinPatch, as performed in a study by Wallace (2012) whereby cost layers were introduced in order to evaluate trade-offs. However Wallace (2012) did not incorporate fisher behaviour and the author stresses the importance to include this in any future analysis. Nevertheless, before such use for policy, the predictive ability of these models does need to be evaluated using a form of cross-validation (see Chapter 4).

Chapter 7. Discussion

Sunset at sea (Source: Jim Ellis)

7.1 Introduction

A primary aim of fishery management is to balance the fishing opportunities for all sectors of a wider fishing community (so ensuring the cultural and economic viability of coastal communities, maintaining fisher knowledge and providing fish products for wider society) with the need to maintain fish stocks in a healthy state and, of increasing concern to managers, the need to reduce fishing impacts on the ecosystem while also considering the needs of various other maritime industries (e.g. offshore renewable energy, transport, leisure, recreational fishing, aggregate supply).

Although some regulations (e.g. mesh size) are relatively easy to enforce and are applied across the relevant fleets (a key issue, as most fishers want a "level playing field"), others may not be particularly effective. For example, fisheries management within the European Union has been based mainly on setting TACs (and allocating quota nationally) as a means of ensuring appropriate levels of fishing mortality on the main commercial species (based on what are traditionally single-species stock assessments). Such an approach is of course best suited to highly selective fisheries exploiting a discrete stock, whereas in reality most fisheries exploit a range of stocks over varying spatial scales. Hence, issues such as misreporting, illegal fishing and discarding have compromised the quota system, especially in mixed fisheries.

Management of fisheries in the past has treated the fishers as fixed components with no consideration of their individual behaviour and goals when fishing. However, to paraphrase Newton's third law of physics, "for every *fishery management* action, there is an equal and opposite *fisher* reaction". The omission of fleet behaviour parameters from management systems that rely solely on biological assessments can lead to overconfidence in the likely effectiveness of proposed management actions. For instance, early models of Marine Protected Areas (MPAs) assumed that fishing effort would be displaced uniformly to other areas or simply dissipate (Wilen *et al.*, 2002). However, Dinmore *et al.* (2003) showed that the North Sea cod closure of 2001, which was developed without considering the behaviour of the fleet, simply resulted in effort displacement along the boundaries of the MPA, causing negative impacts on the ecosystem (including the benthic communities). Moreover, Yew and Heaps (1996) showed the benefits of incorporating a model of fleet behaviour into the management process by scenario-testing the outcomes of the current policy of limited licence entry to reduce fishing effort. The conclusion of that study was that the intended policy would not achieve its desired effect because fishers could potentially circumvent the regulation by fishing for more days. Other studies have shown that using models to assess a fleet's responses to management measures can provide essential information on fleet dynamics that can be used then to inform the management decision-making process (Pelletier and Mahévas, 2005; Bastardie *et al.*, 2010; Lehuta *et al.*, 2010).

The long-term motivation behind this thesis is that ignoring fleet and fisher behaviour in fishery management decision-making will undermine the overall value and likely success of fishery management, so the study was designed to provide new tools to assist the incorporation of behaviour into management-relevant modelling. The overarching research objective covered in the various chapters was to investigate the various factors that affect fishing mortality, and how more detailed information on fleet structure and dynamics (including effort and capacity) can improve our knowledge of the relative contributions of different components of a fleet to fishing mortality of the main target species.

7.2 Scientific contribution of the thesis

This thesis has resulted in three original peer-reviewed publications to date, and was intended to describe and quantify the links between fishing capacity, fishing mortality and fishing effort by considering various spatial, temporal, social, ecological and economic factors. I have demonstrated in all of the chapters that understanding the behaviour and dynamics of fishers and their fleets is hugely important for managing fisheries. Specifically, I have contributed scientifically to understanding of (a) how fishers invest/disinvest in capacity by implementing a multinomial logit model (Chapter 3); (b) how fishers utilise and allocate effort by using a mixed discrete choice model with partial cross validation techniques (Chapter 4); (c) how to link capacity and fishing effort to mortality in mixed fisheries using a combination of multivariate approaches (Chapter 5); and (d) how the key determinants of fisher behaviour, including competition for space from other important maritime sectors, can be modelled using a mixed discrete choice model to determine its utility as a tool for marine spatial planning (Chapter 6).

In terms of linkages across the work, Chapter 4 gave a good indication of the drivers determining how effort is allocated; these same drivers were then applied in Chapter 6 at a finer scale and produced similar results. Further, predicting location choice in multispecies fisheries and linking capacity to the resources, Chapter 5 builds on the findings of Chapter 4. For example, external factors or management regulations (input or output) that affected plaice could result in a changed catchability for sole. A similar approach might be applied in the methodology described in Chapter 6, and the implications of introducing a Marine Conservation Zone could then be assessed in terms of effort displacement and mortality induced on the resource assessed. Although that was not done within the framework of the present work, the results from the study are promising and lay the foundations for future work. Finally, in Chapter 3, I showed that there was no single significant variable driving investment in the study fleet; several economic variables can affect investment. Each of the sections below considers the main contributions of each chapter in the wider context of the research, limitations and future directions.

7.3 Exit and entry of fishing vessels: an evaluation of factors affecting investment decisions in the North Sea English beam trawl fleet

In Chapter 3, the strategic decision-making behaviour of fishers when investing (or dis-investing) in the English North Sea beam trawl fishery was examined using a discrete choice modelling approach. It was assumed that the decision to enter into, remain in or exit from a fishery depended on anticipated future profit. Decommissioning grants have been offered by the European Commission as a voluntary incentive to reduce the capacity of member states' fishing fleets, so are additional strategic choices available to fishers. By integrating available cost data, decommission grants and other factors that were likely to influence future anticipated benefits or losses, it was possible to predict whether operators chose to enter, remain, exit or decommission. Important factors considered in the analysis included future revenues and operating costs (e.g. potential fuel price increases), stock status of the main target species and the impact of management measures (e.g. total allowable catches, TACs), and total fleet size (i.e. congestion/overcapacity). The model provided a strategic planning tool that can be used to help develop management plans to align fleet capacity with fishing opportunities.

The results indicated, as expected, that older vessels were more likely to leave than newer ones. This is to be expected as older vessels are generally subject to replacement by newer ones, resulting in a gradually increasing efficiency of the fleet. The results also indicated that the larger the fleet, the more vessels entered, and the smaller the fleet, the fewer the number of vessels that exited, reflecting an historic trend in the sense that when a fleet experiences increased fishing opportunities (large catches of plaice and large revenues) more vessels enter. The number of active vessels has reduced gradually to very low levels in the past few years, with just a few larger, new vessels remaining, and fewer vessels are now exiting from the small fleet (specifically because they have acquired the quota of others who have left).

The stock status of sole was also an important factor determining investment, with a reduced sole spawning-stock biomass (SSB) yielding a greater probability of exit (and an increasing plaice revenue giving a greater probability of entry). The impacts on the long-term management of the fishery are such that in the medium to long term, unsustainable management of the stocks of sole and plaice (with consequent declines in stock biomass) will provide less opportunities and the fleet would then either decline in size or diversify (in terms of fishing areas and/or species).

The results for revenue unsurprisingly indicated that vessels with lower revenues had a greater probability of exiting the fleet. The implication of vessels with low revenues (or low value per unit effort) departing the fisheries must surely have an impact on overall fleet efficiency. Decommissioning programmes during the period of the study did not entice beam trawlers to decommission at a fast rate, and just 32 vessels out of more than 700 vessels in the whole UK fleet were persuaded to withdraw (Nautilus, 1997). This result was possibly attributable to quota and licence restrictions. Surprisingly, fuel price had a very small effect, perhaps partly because prices in the 1990s were relatively stable and that it was only from 2000 that there were large hikes in the price, up to £0.60 per litre in 2008. The model could have adjusted for this and rendered the effect small. It was assumed that the performance of a vessel in its first year of entry to the fishery achieves the expectation surrounding the entry decision. It was also assumed that the decision to exit, remain or decommission was based on the most recent year's performance, and that a decommissioning programme was based on scrapping a vessel with a 100% grant.

Misreporting practices or any other falsification of technical data used in the analyses will potentially bias the results and the predictive power of models applied for policy. As a suggested way of improving future models, data on pre-entry and post-exit performance related to revenues in other fisheries would be useful, along with background information on the origin of the vessel or its future location and use (Quillérou and Guyader, 2012; Van Putten *et al.,* 2012). As an example, as vessels are decommissioned or exit a particular fishery, what actually exits and what are the implications for the previously targeted stock(s)? Vessels may have been involved in many fishing activities, for example a vessel may engage in pot fisheries for crustaceans or whelks for part of the year, but be involved in fishing for demersal stocks (with otter trawl, longline or gillnet) for most of the year. Furthermore, there may be impacts on other stocks, for example the spatial distribution of fishing effort may change as some vessels exit a particular fishery and impact other stocks. This will result in fishing mortality changes on different ages of the new target stock and by-catch, and the subsequent discard levels. As some of the fleet exit others do not, it still results in the most profitable remaining with money from buybacks or government grants used for more investment (Chapter 3), i.e. new powerful engines, technological advancements (see Chapter 5) resulting in greater impacts on different stocks. Social changes also happen, as those that do not exit the fleet can be bought up by other national or international fishers, which is what happened to a large portion of the beam trawler fleet transferring to the Netherlands (Chapter 3). This can have implications in terms of changes in targeting (fishers like to fish nearer their homeports) and hence fishing mortality.

7.4 Dynamic prediction of effort re-allocation in mixed fisheries

In Chapter 4, a discrete choice model was developed to determine how fishing effort was allocated spatially by the English North Sea Beam trawl fleet. Individual vessels could fish in five distinct areas. The utility of fishing in an area depended on previous success, measured in this case as high catch rates (here, revenue-based i.e. value per unit effort, vpue), and experience, measured as past fishing effort allocation, and perceived costs based on fuel prices weighted by distances to landing port. Both lagged vpue and lagged effort were included as explanatory factors. The models were evaluated using iterative partial cross-validation by fitting the model over a series of separate time-periods (nine in total) to show changes in the drivers over time, because there were changes in both ownership and spatial management. The model can be used to predict potential changes in effort allocation under various management strategies for spatial control.

The utility of fishing in a location (the distinct fishing area) depended on previous success measured as high catch rates (in terms of economic vpue) as well as previous experience, in this case a measure of past fishing practice (effort allocation), as well as perceived costs (measured as distance to landing port weighted by fuel price). Therefore, the results of the RUM (random utility model) analysis showed some of the assumptions that could be made *a priori* for location choice.

Essentially, previous knowledge or experience of a given fishing rectangle had a bearing on the decision to fish there. The RUM analysis included only choice-specific variables (i.e. factors that varied for alternative decisions), but this was adequate because the beam trawl fleet is relatively homogenous in terms of size of vessel and the target species and their habitat. The model can be used to predict potential changes in effort allocation under various effort and catch control management strategies. The predictions may also help policy-makers understand fleet dynamics and the impact that regulations may be having on the fleet.

The methodology for the definition of fleets was based upon the Data Collection Regulation (DCR) of the European Commission (EC, 2006a), which defined the beam trawl fleet based on its use of a beam trawl for >50% of the time during a fishing trip. The fleet activity or métier is then defined as the fisher's tactic at a trip level, expressed as the group of targeted species. In this study the beam trawl métier that targeted primarily crustaceans (i.e. shrimps) was omitted, and only the demersal métier was used in the analysis. Métiers are characterised as an outcome of a trip based on the landing composition, which itself is calculated as a fraction of the total monetary catch. This removed the differences in catch rate attributable to vessel capacity. Moreover, the fractions of the catches were based on economic value rather than weight, reflecting the view that fishers are profit maximisers and that less-common high-value species being targeted have more weight in the analysis. It was assumed that stock abundance was relatively constant over each time-step (month), so stock effects were excluded.

For the purpose of this application, because the set of areas from which choices for beam trawlers targeting flatfish grounds, which can be defined by depth and sediment and seasonal availability (De Veen, 1976, 1978; Rijnsdorp and Pastoors, 1995; Hunter *et al.,* 2003) are quite specific, one may expect relative stability of selection for some years hence. One of the main issues in this analysis was the aggregated spatial scale of the choices, but this chapter focused on detailed temporal dynamics because it is more difficult to present fine-scale spatial and temporal dynamics. A management strategy evaluation, MSE, model and framework (a model is currently being developed to analyse the behavioural response of fishers to adjustments of TACs and effort levels to achieve a set of fishery management objectives) and also to investigate the implications on other species taken as by-catch in the same fishery (Romero *et al.,* 2013; Pascoe *et al.,* 2013). Future work could usefully involve research into how much stock assessment and associated projections better account for changes in fleet dynamics. What is the utility of routine stock assessments versus an occasional full MSE? Stock assessments aren't always spatial, yet there is a clear need to appreciate fleet dynamics in the formulation of management advice and in gauging the efficacy of potential management measures. Given the recent upsurge in spatial management, work to investigate how fleets may respond to different types of spatial closure in space and time, and the implications of the closure on the target species, bycatch species and wider marine ecosystem is required. For example, what are the likely impacts on catches, discarding levels, population structure and habitats, and what are the socio-economic impacts?

7.5 Effective fishing effort trip indicators and their use for efficient spatial management in mixed demersal fisheries

In Chapter 5, specific fleet landing profiles attributable to alternative fishing strategies of beam trawlers were analysed to define distinct fleet activity. The relationship between fishing mortality and effort exerted by the English beam trawl fleet was investigated for two stocks of North Sea demersal fish, plaice and sole. Catchability was adjusted by accounting for specific targeting of the gear, changes in efficiency, seasonal and area effects and individual vessel variation. This was undertaken on the basis of results from a GLMM, a mixed effects general linear model (GLM) that included random effects (in this case, vessel).

A mixed linear model is a generalization of the standard linear model used in a GLM, the generalization being that the data are permitted to exhibit correlation and non-constant variability. The GLMM therefore provided the flexibility of modelling not only the means of the data (as in the standard linear model), but also their variances and covariances. The method can be applied in cases where fishing fleet surveys are not collated and detailed vessel/skipper specific information is not available for a sample (or the whole population). It was possible to distinguish between effort measures such as days-at-sea and hours fished (this model) and to refit the relationship between fishing mortality and nominal effort (h). Changes in fishing efficiency were calculated, and descriptors standardised in relation to distinct submétiers and their impact on both target species.

The implications of setting management decisions on effort measured in hours are discussed in the context of recent regulations (days-at-sea restrictions) that are having an impact on North Sea fleets. Irrespective of potential changes in fishing tactics maximising the number of hours fished, increases in efficiency were evident for one stock (sole), whereas decreases in efficiency for plaice could indicate increased targeting of sole. More importantly, the slope of the regression in each case increased. In practice, this implies that management that considers several factors (capacity, seasonal and area effects) that contribute to effective effort should be more effective in reducing fishing mortality than management actions based purely on nominal effort. The policy implications are such that adjusting effort such as days-at-sea (or h at sea) by capacity (and taking into account month and area effects) should result in greater than proportional decreases in fishing mortality.

The reliability of the relationship between fishing mortality and fishing effort depends on the precision of both the measurement of the effort and the estimate of the mortality. In both cases, adjusting fishing effort (by taking into account significant explanatory factors) led to a substantial gain in the precision of the relationship between fishing mortality and fishing effort. How viable it is to adjust for these by regulation (and the implementation thereof) requires further assessment. A key finding of the study was the switch in targeting and the changed fishing efficiency, an estimated 6.2% decrease in plaice and an estimated 0.6% increase in sole annually for averages calculated over the 11-year study period, probably because of changes in targeting, fuel costs and regulations.

In the study, one of the main assumptions is that a fishing vessel capacity unit (VCU) is a proxy for capacity. The rationale for this is that up to now, VCUs have been the basis of vessel reduction programmes in the UK. Landing rates, i.e. nominal landings per unit effort (lpue) was calculated as catch in kg per hours fished per vessel per trip per area. The results suggest that a VCU may provide a reasonable approximation of fishing capacity for the beam trawl fleet.

The models for this study were heavily reliant on estimates of fishing mortality (*F*) drawn from ICES working groups. If the estimate is biased, however, there will be variances in the *F*~fishing effort relationship. There are issues relating to how lpue models deal with zero catches when effort is recorded as non-zero, so adjustments have to be made to account for this by including a small catch (the method commonly used). Traditionally, the method has been applied by scientists to yield the most normal distribution of residuals when using a log-transformed dependent variable and a normal error model, but there may be other reasonable approaches to the problem.

Physical characteristics that inform on VCU may have been misreported to or not updated in the fleet register, in which case raising implications not obvious in the model results. Another issue that would bear further investigation in future is to determine whether sales slip data, which contain nominal landings and effort associated with logbook landings, have been input correctly at port offices or misreported originally by fishers.

The study has shown clear applications for input control for mixed fisheries management and has also complemented other research initiatives such as recent catch quota trials (FVM, 2009) undertaken by the UK, Denmark and Germany using remote electronic monitoring (REM). Future work may need to account for other factors, ranging from non-target fish and wider ecosystem impacts to the social and economic implications of effort controls (Cheilari *et al.,* 2013) and their impacts on different sub-métiers (ICES, 2009b).

7.6 Fishing for space

Based on the methods described in Chapter 4, a method was developed using fine spatial resolution using Vessel Monitoring System (VMS) data and a discrete choice model to analyse the determinants of English and Welsh scallop-dredging fleet behaviour, including those of competing sectors operating in the eastern English Channel.

Results showed significantly that aggregate activity and maritime traffic have a negative impact on the utility of fishers, and that past success, expected revenue and fishing within the 12-mile limit have a positive effect on their utility. Based on the results, the model showed promising application for Marine Spatial Planning (MSP). One of the main assumptions was the way in which value per choice is calculated, as the proportion of the total value per ICES rectangle based on effort derived from VMS. The definition of a vessel's activity derived from VMS was formulated from another study (Lee *et al.,* 2010). One of the issues raised in this study was not having access to raw/unprocessed maritime traffic data, so a proxy had to be used in the form of percentage coverage of maritime traffic lanes. Further, the vessels used were >15 m which, by law have to supply position and activity information. Regrettably, fishing data for scallopers <15 m or for international fishing vessel are not readily available yet, so future work may well benefit from following up on these components of the scallop fleet. The results from the study are nevertheless promising and lay the foundations for innovative future work that could include real Marine Conservation Zones (MCZs) and the economic implications fishers potentially face from their introduction (Van de Geer *et al.,* 2013), shipping traffic from Automatic Identification System (AIS) data, including vessel position and movement. It would also be potentially possible to identify fishing vessels at a finer temporal and spatial resolution and as such would validate the effort estimates obtained from VMS data and give an accurate indication of compliance. This approach could the be applied to the log book effort estimates at a broader scale to give an indication of effort mis-reporting across management units which is in turn vital for the successful management of fish stocks.

7.7 Overall conclusions and future directions

7.7.1 Model validation

The overarching objective of this study was to use RUM to explain and predict fisher behaviour (e.g. Chapters 3, 4 and 6). This requires variables that can be used to explain the observed historic patterns in fisher behaviour and then predict future responses. If future conditions have not been seen in the historic observations then future predictions will be based on extrapolations. For example fuel prices have continued to increase and current management is aiming to recover stocks to levels that will support MSY (a level which has not been seen in the past). Therefore future costs and catch rates will be greater than those used to fit the RUM. In Chapter 3 for example, the model fitted the observations well showing the main factors that contribute to fisher decision-making; however the r^2 used to determine its predictive performance was low (0.22); nevertheless r^2 is not necessary a good guide to how well a model predicts as there is a possibility to over fit data by adding too many degrees of freedom and increase r^2.

If a model is to be used to inform the management process and support policy decisions, model validation is essential in checking that the model addresses the problems posed and describes the system being modelled accurately. Scientific advice provided to managers based on exploring the potential outcomes of alternative management strategies, without model validation, may result in erroneous decision-making down the line. A model may provide the best fit (describing how well the model fits the observations) but it may have poor predictive power, i.e. it could be unable to predict future events based on system changes in the past not captured in the fit, so a fleet's response cannot be predicted with certainty. It should be noted that models only form part of the inputs of a manager's decision-making toolbox (Grant, 1986; Pitelka and Pitelka 1993). Limits to the range of the data may limit a models' ability to predict because of observations not seen in the fit, and predictions will then need to be based on extrapolation. Further, correlations in the data may mean that different models fit the data equally well but have poor predictive power. For instance, if fuel prices increase while plaice catches decline, there may be a linear relationship (collinearity) between the two variables. Choice of variables may depend on whether they can be used as control variables by managers to influence costs or catch rates, e.g taxes or TACs and area or seasonal closures. However some variables such as stock abundance are not directly controllable as they are the result of complex ecological processes. Collinearity means that changing one of the variables results in a change in another, and that it is no longer possible to predict the effect of a single variable such as fuel price when analysing the marginal effects of the explanatory variables. An additional issue currently is that goodness of fit diagnostics for a collinear model can appear to be a highly significant. However, the problems with collinearity are not with the model fit but with the parameters. Collinearity can lead to erroneous parameter estimation in statistical models (Weisberg, 1985). If predictions are performed within the range of the data this may not be a problem, but extrapolating outside the range of the data for a collinear model may be risky. Alternative models may equally be supported by the data, in which case selecting multiple models and comparing their predictions may be beneficial.

Model choice is not necessarily about selecting the best model, because the recognition that there may be several equally good explanatory models is important in developing a better understanding of system dynamics. Also, simple models will help in generalisation, i.e. they will work in scenarios different from those in which the model was developed and tested. Principal Component Analysis (PCA) can help alleviate the problems of multi-collinearity in the dataset by converting correlated variables into uncorrelated components. It performs this by identifying directions in the data and places them in components with the greatest variation and uses linear combinations of the variables to describe the component. These linear projections provide a platform on which to base outputs from correlation matrices in supporting variable selection for the model(s).

Future work will be to evaluate whether fisher behaviour can be predicted based on past behaviour and whether these models can be used to model changes in fishing effort in response to management, economics and changes in stocks and to evaluate whether it is actually possible based on the data availability, i.e. what I currently have and what is needed.

One option for future work to better assess the performance of a model when there are sufficient data, and one that is widely adopted by other researchers to assess the true error of a model, is a statistical process called cross-validation (Kohavi, 1995). Cross validation is becoming a more widely used technique adopted by ecologists (although it seems to be less well used in fisheries science) used to provide defensible hypothesis about processes and conclusions (e.g. Boyce *et al.,* 2002). The basic theory behind cross-validation is to split the data by removing a portion to build a model (the training set), then using the remainder of the data (the test set) to test the performance of the training set model by computing the mean square error. The procedure is repeated k times by randomly partitioning different portions of the data in turn and predicting the test set $k-1$. Each model is then assessed on the different subsets of the data it predicts and an average proportion predicted is compared with the observed data from each test set. A confusion matrix can then be created to assess model performance, including a statistical test of index of agreement called a weighted kappa score between classifications of observed vs. predicted for each model. The weighted kappa takes account of data anomalies such as class skew (specificity and sensitivity). The model with the highest score for kappa would be considered the best predictor. Sensitivity and specificity confidence interval score can be assessed, confidence intervals calculated and the trade-offs observed through plotting receiver operator curves (ROCs).

In terms of my analyses, I evaluated my choice predictions using iterative partial cross validation by fitting the model over a series of separate time-periods and comparing with the observed choices (Chapter 4), although this can be a time consuming process. Indeed, Apostalaki *et al.* (2008) emphasized this common problem faced by scientists and stated that validation testing imposed delays on the inclusion of new knowledge, and the reluctance to include new knowledge due to time constraints. Nevertheless on-going research will factor in more time for cross validation and quality control checks and the possible implementation of new knowledge as it arises.

7.7.2 Qualitative survey

Several authors have stressed the importance of including fisher knowledge in management decision-making (McGoodwin, 2006; Menzies and Butler, 2006). Fisher knowledge was absent from this study because there was no time or resource to conduct a qualitative survey, so assumptions had to be made in the models constructed. Most fisher behaviour analyses based on decision theory has been constructed via theoretical economic theory and/or knowledge that's been published (Abernethy *et al.,* 2007). However, there are often issues of trust with respect to data confidentiality and how the data may be used by governing authorities (Degnbol and Wilson, 2008). Surveys also have to be conducted in a particular way, because respondents can influence the direction of the interview. An interview should not guide the interviewee, but for a thorough overview, an interview is best conducted using a semi-structured process (Bernard, 1994). A qualitative study would give a better understanding of fisher knowledge and the external factors influencing fisher behaviour, and it would also give the opportunity to potentially validate many of the assumptions made in this thesis. More importantly having this knowledge would increase the success of proposed management strategies in terms of willingness to comply and respect (Dimech *et al.,* 2009).

7.7.3 Management strategy evaluation (MSE)

Improved understanding of the key sources of uncertainty is required for fishery management to be effective. One method to determine the effectiveness of management plans is to use a simulation approach known as management strategy evaluation (MSE; Kirkwood, 1997; Butterworth and Punt, 1999; Sainsbury *et al.*, 2000; Kell *et al.*, 2007, Butterworth *et al.,* 2010). New tools are currently being developed to evaluate methods for providing scientific advice to fishery managers, including MSE, in collaboration with stakeholders (e.g. the EU-funded project JAKFISH: https://www.surfgroepen.nl/sites/jakfish/default.aspx). The process is known as participatory modelling, and scientists and stakeholders develop flexible, transparent models to enhance common understanding of biological and fishery management issues, so reducing the risks and consequences of implementing different management plans and also involving fishers more in the management process. Early MSE focused on target species (Kell *et al.*, 2007), but recent MSE have focussed on the broader impacts of fishing, e.g. ecosystem-based fisheries management, with management strategies being evaluated based on a set of environmental indicators (e.g. Pikitch *et al.*, 2004, on the EU-funded project IMAGE, http://ec.europa.eu/research/fp6/ssp/image_en.htm). Ecosystems are complex, dynamic and poorly understood, so predicting the results of any management plan at an ecosystem level is highly uncertain. MSE methods rely on simulation testing to assess the consequences of a range of management options and to evaluate each performance measure across a range of objectives, requiring the use of an operating model (OM; see Figure 7.1) to simulate the actual system to be managed and to evaluate the performance of alternative candidate management procedures (MPs) to be applied in practice.

Testing an MP should essentially be a blind experiment where information about the system is limited to the data available to the stock assessment. Performance statistics based on the OM (e.g. yield, probability of stock collapse) are then used to evaluate the performance of management against its objectives. MSE allows for testing the robustness of different management strategies to a lack of knowledge and/or data, both being major problems in providing advice under current advisory frameworks. Under the MSE approach, the objective is no longer to come up with the single answer, but to evaluate the consequences of different management strategies under alternative assumptions about overall system dynamics, i.e. its robustness to uncertainty. An important aspect of MSE is that the management outcomes from the harvest control rule (HCR) are fed back into the operating model so that their influence on the simulated stock, and hence on future simulated fisheries data, is propagated through the stock's dynamics. Traditional stock assessment has mainly considered just uncertainty in the observation process (e.g. recruitment). Uncertainty about the actual dynamics (i.e. model uncertainty) can have an even greater impact on whether management objectives are achieved (Punt, 2008). However, the effects of a HCR can be quite different from those intended because of the response of fishers to economic incentives and, as such, HCRs are generally poorly equipped to represent human welfare and MSEs tend not to represent implementation error well (Milner-Gulland, 2011).

A current challenge is to characterise and communicate uncertainty involving a range of stakeholders and to integrate ecosystem and economic models more fully into MSEs. Incorporating fleet behaviour into an MSE framework would reveal the benefits of fleet behaviour models, and this would be a necessary step forward in the use of MSE as an advisory tool (Venables *et al.*, 2009).

In future there would be value in attempting to incorporate the behavioural model developed in this thesis into a MSE, to evaluate retrospectively the observed behavioural response of a fleet to management measures implemented against model prediction, then to use the results to quantify the uncertainty in the predicted response to reveal the magnitude of the implementation error associated with model prediction. However, the bigger question is likely to be what should be in an MSE? For instance, what or how many assumptions about the fleet need to be made in terms of what is realistic to include, given the data and history of technical changes, and of course what is the cost of collecting more data if that is deemed necessary. Also, what is practical to manage, and can we manage at the level of a métier?

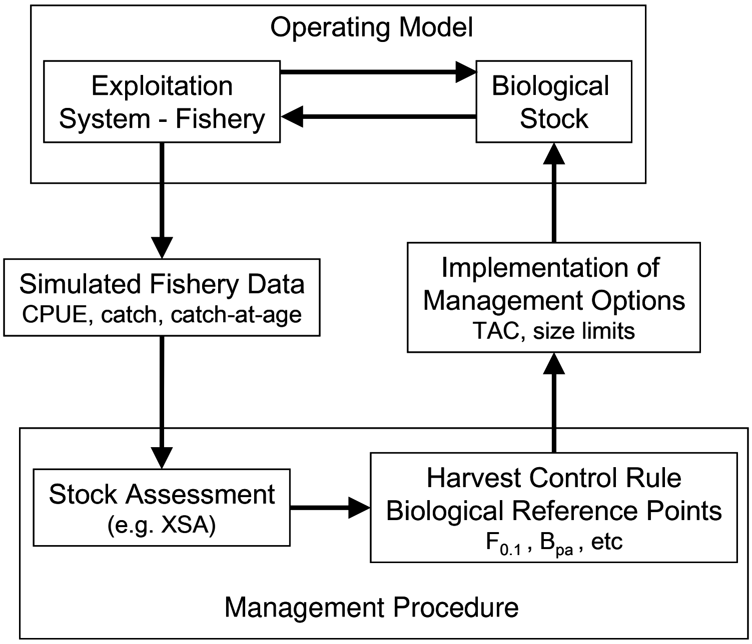


Figure 7.1 Conceptual framework of the MSE (see text above) that includes (i) an operating model that represents alternative plausible hypotheses about stock and fishery dynamics; (ii) an observation error model that describes how simulated fishery data are sampled from the operating model and (iii) a management procedure or management strategy that estimates stock status from the pseudo-data and generates management outcomes. Note that in this case implementation error is represented in the implementation of management options box.

Fisheries management has progressed over the course of the 20th century, but given the large proportion of stocks that are depleted or over exploited (FAO, 2010), the threat to many coastal communities, and the increasing number of marine species that have been lost or listed as endangered (Dulvy *et al.,* 2003), there is a clear need for improved management. As the European Community and other nation states move towards EBFM, in order to balance food production and security with wider ecosystem concerns, the types of model developed here will be of increasing importance in developing robust management plans which properly account for fisher and fleet behaviour.

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