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The impact of environmental regulation on human interaction with marine and coastal ecosystems

A thesis submitted in fulfilment of the degree of Doctor of Philosophy

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by

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Abstract

The primary objective of this thesis is to investigate the impact of environmental regulations on stakeholder interaction with marine and coastal ecosystems as well as provide a thorough treatment of the contrasts in stakeholder and scientific attitudes toward ecosystem management. With respect to stakeholder interaction with marine and coastal ecosystems, the focus is mainly on the behavioural response of fishermen to regulatory changes in fishery policy, but an emphasis is also placed on more diverse stakeholder groups in ecologically important and remote coastal areas, especially as this relates to contrasting ideologies on ecosystem management. The analysis is carried out in three separately framed research papers, each applying separate empirical methodologies.

The first paper considers the impact of changes to species quotas on a fishing fleet's harvesting behaviour in a multi-species fishery by approximating the fleet's objective function using the expected utility hypothesis and portfolio theory. This approach allows the impact of quota changes on the composition of theoretical species harvest portfolios to proxy for the impact of quota changes on fleet's actual retained species catch. The paper contributes to the literature by applying a relatively novel methodology in fisheries research (portfolio theory) to fisheries, using this approach to estimate the impact of quota changes on fleet behaviour. The results indicate that the objective function of the fleets under study were well approximated using the expected utility hypothesis and portfolio theory. Results also suggested that large quota changes lead to significant amounts of displaced effort to be redirected toward the harvest of alternative species in a multi-species fishery, or indeed, in neighbouring fisheries.

The second paper demonstrates an application of Biome Portfolio Analysis (BPA); a novel adaptation of financial portfolio theory for ecosystem management. This is a spatially orientated framework which uses Geographical Information Systems (GIS) and surveys local stakeholders and agencies on their attitudes toward ecosystem risks and benefits across space and habitat types and uses this information to assist in coastal management decisions. The paper's contribution to the literature is its use of survey techniques to quantify the extent of the 'attitude gap' between scientists and local shareholders with respect to the optimal management of coastal and marine areas. The results showed that BPA was a useful tool for coastal managers to use when wishing to assess the main sources of ecosystem services and threats in a given area, particularly if needing to do so across interest groups. Furthermore, the results empirically supported earlier commentaries in the literature which stressed the degree of incoherence in the EU principles of Integrated Coastal Zone Management (ICZM).

Paper 3 assesses the impact of changes to multispecies quota constraints on the fishing location choice of demersal otter trawlers in Irish waters. The paper uses spatial mapping of fishing locations in Irish waters and records of retained species catches within each area to create a spatial picture of demersal otter trawl fishing intensity around Ireland. The paper's contribution of this paper to the literature is that it combines Vessel Monitoring System (VMS) fisheries data and econometric choice modelling methods to assess the spatial response of fishermen to policy changes. Overall, results indicate that quota changes have a significant impact on the fishing location choice, displacing effort and redirecting it to alternative fishing sites

Table of Contents

List of Figures

List of Tables

Abbreviations

ARIMA: Auto-Regressive Integrated Moving Average

BPA: Biome Portfolio Analysis

CBD: Convention on Biological Diversity

CFP: Common Fisheries Policy

CL: Conditional Logit

CPUE: Cost Per Unity of Effort

CSO: Central Statistics Office

DG-MARE Directorate General of Fisheries and Maritime Affairs

EAP: Environmental Actions Programme

EBFM: Ecosystem Based Fisheries Management

ED: Electoral District

EC: European Community

EEC: European Economic Community

EU: European Union

FCS: Favourable Conservation Status

GDP: Gross Domestic Product

GES: Good Environmental Status

GIS: Geographical Information Systems

HMS: High Migration Species

HBD: Habitats and Birds Directives

ICES: International Council for Exploration of the Sea

ICZM: Integrated Coastal Zone Management

IFIS: Integrated Fisheries Information System

IIA: Independence of Irrelevant Alternatives

IID: Independently and Identically Distributed

IM: Integrated Management

IMP: Integrated Maritime Policy

INFOMAR: Integrated Mapping For the Sustainable Development of Ireland's Marine Resource

MI: Marine Institute (Ireland)

MNL: Multinomial Logit

MPA: Marine Protected Area

MS: Member State

MSFD: Marine Strategy Framework Directive

MSP: Marine Spatial Policy

MSY: Maximum Sustainable Yield

MWGI: Marine Work Group Ireland

NMFS: National Marine Fisheries Service

NUTS: Nomenclature of territorial units for statistics

NWWRAC: North Western Waters Regional Advisory Council

OECD: Organisation for Economic Co-operation and Development

OSPAR: Convention for Protection of the Marine Environment of the North-East Atlantic

PT: Portfolio Theory

RAC: Regional Advisory Council

RNL: Repeated Nested Logit

RPL: Random Parameter Logit

RUM: Random Utility Model

SAC: Special Area of Conservation

SFPA: Sea Fisheries Protection Authority (Ireland)

STECF: Scientific, Technical and Economic Committee for Fisheries

TAC: Total Allowable Catch

TCM: Technical Conservation Measures

UN: United Nations

VMS: Vessel Monitoring Systems

WFD: Water Strategy Framework Directive

1. Introduction

The primary objective of this thesis is to investigate the impact of environmental regulations on stakeholder interaction with marine and coastal ecosystems, primarily with respect to the fishing industry. A further objective is to provide a detailed understanding of the gap between stakeholder and scientific attitudes to management of coastal and marine ecosystems and their derivative services. Beyond establishing a context for this thesis and detailing the types of services that marine and coastal ecosystems provide society, this chapter is concerned with providing justification for the overall research questions of this thesis. Section 1.1 provides a description of ecosystems services. Section 1.2 compares the contrasting origins of ecosystem management and fisheries management. Section 1.3 provides an overview of the research objectives and how these objectives have been achieved through three separate empirical papers. Section 1.4 outlines the structure of the thesis. Thesis outputs such as papers and public presentations are provided in section 1.5.

1.1. Context: Ecosystem Services

According to de Groot et al. (2002), 'early references to the concept of ecosystem functions, services and their economic value date back to the mid-1960s and early 1970s (e.g. King, 1966; Helliwell, 1969; Hueting, 1970; Odum and Odum, 1972)'. Economic policy and analyses often focus on services for which there is an immediately apparent economic market; for example food provisioning, fossil fuel production, timber harvesting etc. However the flow of benefits provided to society by natural capital extends far beyond ecosystem outputs intended for economic markets. The millennium ecosystem assessment (MA, 2005) built on previous research into ecosystem services (e.g. Costanza et al. 1997; Daily, 1997; Daily et al. 2000; de Groot et al. 2002) and involved the work of 1300 scientists around the world to define what is now one of the most widely used classification of ecosystem services. Within this classification there exist four categories of ecosystem service: provisioning services, regulating services, cultural services and supporting services, each of which is broken down into further subsets of services shown in Table 1.1[1]

Table 1.1: Millennium assessment classification of ecosystem services

Provision Services	**Regulating Services**	**Cultural Services**
Products obtained from ecosystems	***Benefits obtained from regulation of ecosystem services***	***Nonmaterial benefits obtained from ecosystems***
> Food	> Climate regulation	> Aesthetic
> Fresh water	>Disease regulation	> Recreation and ecotourism
> Fuelwood	> Water regulation	> Aesthetic
> Fibre	>Water purification	> Inspirational
> Biochemicals	>Pollination	> Educational
> Genetic resources		> Sense of place
		> Cultural heritage
Supporting Services		
Services necessary for the production of all other ecosystem services		
> Soil formation	> Nutrient cycling	> Primary production

Source: From MA (2005).

Provisioning services account for physical products obtained from ecosystems like fish, fresh water and fuel-wood, regulating services play a climatic and environmental regulatory role and cultural services provide a host of non-material benefits. In turn, all three of these categories are reliant on supporting services, which differ from provisioning, regulating and cultural services in that their impacts on people are either indirect (and operate through their effect on regulating services) or occur over a very long time. For example while humans do not directly use soil formation services, changes to this service would indirectly affect society through an impact on the provisioning service of food. Of particular note is the fact that more

[1] The full description of the breakdown of ecosystem services within each of the four main categories in the classification is shown in Appendix A.

immediate provisioning services are often better understood by the public, for example those in coastal zones and near-shore marine areas, but a vast array of ecosystem services come from less accessible areas, such as the deep sea, as documented by Armstrong et al. (2012).

Any discussion of ecosystem services must also consider the role of biodiversity, for the two are inextricably linked; ecosystems are the product of interaction between the Earth's various life forms, physical geographies and atmosphere. The UN convention on biological diversity (CBD) defines an ecosystem as a 'dynamic complex of plant, animal and micro-organism communities and their non-living environment, interacting as a functional unit' (MA, 2005). Biodiversity is defined as 'the variability among living organisms from all sources including, *inter alia*, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems' (MA, 2005). Biodiversity is thus a structural feature of ecosystems and the variability of ecosystems is an element of biodiversity. The strength of this link for different ecosystems is critical in understanding how anthropogenic impacts on biodiversity affect ecosystem functioning and the supply of ecosystem services to society (Loreau et al, 2002).

Since its conceptual evolution in the 1960s and 1970s until the term was first officially used by Ehrlich and Mooney (1983), ecosystem services has emerged as a substantial area of interdisciplinary research. De Groot et al. (2002) describe an 'almost exponential growth in publications on the benefits of natural ecosystems to human society'. In the inaugural publication of *Ecosystem Services,* Costanza and Kubiszewski (2012) claim it is now a 'well-defined and active enough field of research to warrant its own academic journal', having, as of January 2011, generated over 2400 papers since the 1990s. As a newly emerging area of research, a great many papers on this issue have grappled both philosophically and methodologically with the valuation of ecosystem services (e.g. Daily 1997; Daily et al. 1997; Postel et al. 1997; Loomis et al. 2000; Costanza 2000; Fisher and Turner 2009). Many others have attempted to quantify the impact of human activities on ecosystem health, functioning and services by addressing losses in biodiversity (e.g. Worm et al. 2006; Balvanera et al. 2006; Nelson et al. 2009; Luck et al. 2003). Of critical importance for these latter studies is defining a measure of biodiversity change that can be translated into a measureable change in the supply of ecosystem services. While evidence arising out of such studies is established, it also incomplete; yet there is little doubt that the rate at which human activities are causing losses to biodiversity and ecosystem services is unprecedented and unsustainable:

'60% of the ecosystem services… …are being degraded or used unsustainably, including fresh water, capture fisheries, air and water purification, and the regulation of regional and local climate, natural hazards and pests. The full cost of the loss and degradation of these ecosystem services are difficult to measure, but available evidence demonstrates that they are substantial and growing'.

- MA (2005)

To stress the extent of the unsustainability of the current human use of the earth's natural resources, below are a few of the findings of the Millennium Assessment:

- More land was converted to cropland in the 30 years after 1950 than in the 150 years between 1700 and 1850 and cultivated systems now account for 25% of the Earth's terrestrial surface;
- 20% of the Earth's coral reefs have been lost and a further 20% have been degraded in in last several decades of the 20^{th} Century;

- Water withdrawals from lakes and rivers has doubled since 1960 and 70% of this is for agricultural use;
- Since 1960, flows of reactive nitrogen in terrestrial ecosystems have doubled, while flows of phosphorous have trebled[2];
- 60% of the increase in atmospheric concentration of carbon dioxide has occurred since 1959;
- Genetic diversity, particularly for cultivated species, has declined globally;
- Over the past few hundred years, the species extinction rate has increased by as much as 1,000, relative to the background rates typical of the planet's history.

The provisioning services of ecosystems are often those that are more immediately profitable and demanded by society, creating a more immediate incentive to exploit them (MA, 2005). This becomes problematic when less immediately apparent ecosystem services like regulatory and support services are negatively impacted in human interactions with the natural environment. In economics, such a scenario is referred to as a negative externality: i.e. where a market activity produces economically valuable output and this process has a negative environmental impact for the rest of society (Mankiw, 2001). Viewed from this perspective, human intervention in the natural environment since the industrial revolution, and more pronouncedly since the mid-20th Century, has resulted in unprecedented increases in the level of negative ecosystem externalities.

This trade-off between ecosystem services, particularly provisioning services with regulating, cultural and support services, is at the heart of the issue of unsustainable use of ecosystems.

[2] Nitrates and phosphates are both used as fertiliser for arable and pasture agricultural land. Nitrates cause various health effects in humans and animals and can change the composition of species in an ecosystem due to susceptibility of certain organisms to the consequences of nitrogen compounds. Increasing phosphor concentrations in surface waters increase eutrophication, which is an increase in the growth of phosphate-dependent plants such as algae. These plants deplete oxygen and prevent sunlight from entering the water making aquatic environments unsuitable for diverse species

One of the underpinnings of natural resource economics is the neoclassical assumption of near perfect substitutability The idea is that natural resources and human-made capital are substitutable for one another; as the resource is depleted so the physical capital stock can be accumulated so as to substitute the resource in the production process in such a way that there is always enough output to hold consumption constant (Perman et al, 2003). For example, despite large-scale ecosystem degradation, the millennium assessment also found that changes made to the world's ecosystems in recent decades have provided substantial benefits for human well-being and national development (MA, 2005).

However, as argued by Villamanga et al. (2013), 'the flow of an ecosystem service is not sustainable when demand cannot be met by current capacity or when meeting demand causes undesirable declines in other services or in the future provision of the same service'. Neoclassical economic theory refers to such unsustainability as an inefficient or sub-optimal allocation of resources (Perman 2003). Within this ideology, renewable resources like forestry and fisheries are only managed optimally if extraction of the resource in one period cannot be increased without it being decreased in another (essentially an optimal trajectory over time of resource extraction). Hence bioeconomic models used in fisheries and forestry economic research attempt to calculate a rate of resource extraction that allows the resource to renew itself sufficiently enough for it to be extracted at the maximum level into perpetuity.

However, within the ecological sciences and in particular, the field of ecological economics, the matter is considered more complex due to the existence of nonlinearities in the relationships between human activities and the flow of ecosystem services (e.g. Burkett et al. 2005; Costanza 1996; Barbier et al. 2008; Limburg et al. 2002). Costanza (1996) argues that 'complex systems are characterized by strong (usually nonlinear) interactions among the parts; complex feedback loops that make it difficult to distinguish cause from effect; significant time and space lags; discontinuities, thresholds and limits; all resulting in the inability to simply add-up or aggregate small-scale behaviour to arrive at large-scale results’. This means that the complexity of ecological (and economic) systems make the impact of human activities on ecosystems and the changes to services that will arise, very difficult to predict.

Given the inescapable reality that trade-offs between different ecosystem services must be made, it is not surprising that the millennium assessment found that the net gains in human well-being and economic development, that have been brought about by significant alteration of ecosystems, have been achieved at growing costs in the form of degradation of many ecosystem services. However, of more urgency is the finding that these man-made changes to ecosystems are increasing the likelihood of nonlinear and potentially irreversible changes to ecosystem functioning, for example, increased risk of abrupt alterations in water quality, the creation of ‘dead zones’ in coastal waters, the collapse of fisheries and shifts in regional climate (MA, 2005). Furthermore, there is also the risk that if adjusting ecosystems to procure provisioning services like food and fuel continues at the current rate, a critical point will be reached whereby the capacity of ecosystems, to provide the regulating and supporting services needed to maintain provisioning services at current levels, will be undermined. One example of this is the need for soil biodiversity for crop nutrient-use efficiency, resistance and resilience against stress and disturbance (Swinton et al 2007). The relevance from a fisheries and coastal management perspective is that management decisions must seek to take account of non-linearities and unpredictability in ecosystem responses to human activities and in economic agents’ responses to changes in environmental policy. This issue is addressed throughout the empirical chapters and is a dominant theme of the thesis.

1.2. Modes of Marine and Coastal Ecosystem Based Management

1.2.1. Fisheries and Ecosystem Management

The previous section focused on the context in which ecosystem based management policy is set, which is primarily characterised by the fact that the Earth's ecosystem services and natural capital are in a state of decline. Costanza et al. (1998) estimated the value of these services to society to be worth in the region of US$33trillion per year (with most of that emanating out of non-market sources). Of those services, 63% were estimated to arise out of marine and coastal ecosystems. It should be noted that this figure has become highly controversial amongst natural resource and environmental economists and more recent valuations tender less controversial figures (e.g. de Groot et al. 2012). Nevertheless this statistic echoes the findings of many studies that highlight the significant benefits society accrues from marine and coastal ecosystems and the loss of services that is taking place due to their degradation (e.g. Barbier et al. 2009; MA, 2005; de Groot et al. 2002; Duarte 2000; Worm et al. 2006; Garcia 2003; Sutinen and Soboil 2003, and many others). The importance of these services for populations located along the Irish and European coastline prompts my interest and research in this area. This chapter thus addresses the changing ideological principles of marine and coastal management, as a new paradigm of ecosystem based management increasingly influences policy decisions. Of note is the fact that the objectives of management measures differ according to ecosystem-type and the services that society and policy-makers prioritise for any one ecosystem. In particular, emphasis is placed upon the fundamental differences between fisheries and ecosystem management, and the newly evolving fusion of these two concepts, ecosystem based fisheries management.

According to Garcia et al. (2003), ecosystem management is a derivative of wildlife management, originating in range and forestry management. Therefore its development evolved in an arena where it was possible to directly monitor and influence human activities, and to manipulate habitat and population in space and age structure. As a result, ecosystem management is area based and requires boundaries to be clearly and formally defined. Its aim is to maintain ecosystems in the sustainable condition necessary to achieve desired social benefits. It requires scientific information as an element in a decision-making process that is fundamentally one of public and private choice Garcia et al. (2003). It is further defined by some as 'a management philosophy which focuses on desired states rather than system outputs and which recognises the need to protect or restore critical ecological components, functions, and structures in order to sustain resources in perpetuity' (Cortner et al. 1994)

In contrast to the origin and development of ecosystem management, fisheries management applied to a domain where the possibility of direct intervention (i.e. control of marine ecosystems) was extremely limited, so that management strategies had to focus on controlling human activities as opposed to the ecosystem itself. The best managers could do to interact with the ecosystem was to observe proxies for the state of an otherwise opaque system and fugitive resource, and use this information to further control fishing effort (Garcia at al. 2003). Garcia et al. (1995) state that 'fisheries management aims at optimising the use of fishery resources as a source of human livelihood, food and recreation, dynamically regulating fishing activity, meeting resource related objectives or constraints, mainly indirectly'. Clearly, this differs markedly from the ecosystem concept of management which focuses on 'desired states rather than system outputs'. While it is true that desired states are a function of desired outputs and vice versa, the measurability and tangible accessible controllability of terrain based biomes allowed for a more direct intervention with ecosystem states, while fisheries, historically, has tried to achieve desired states by controlling inputs and outputs. Table 1.2 summarises the features and schematic comparison of fisheries and ecosystem management laid out by Garcia et al. (2003).

Table 1.2: Schematic comparison between fisheries and ecosystem management

Criteria		**Fisheries management**	**Ecosystem management**
Paradigm		Sector-based. Vertically integrated. Focusing on target resource and people.	Area-based. Holistic. Loosely cross-sectoral. Focusing on habitats and ecosystem integrity.
Governance	**Objectives**	Not always coherent or transparent. "Optimal" system output. Social peace.	A desired state of the ecosystem (health, integrity).
	Scientific input	Formalized (particularly in regional commissions). Variable impact.	Less formalized. Less operational. Often insufficient. Stronger role of advocacy science.
	Decision-making	Most often top-down. Strongly influenced by industry lobbying. Growing role of environmental NGOs.	Highly variable. Often more participative. Strongly influenced by environmental lobbies. Stronger use of tribunals.
	Role of the media	Historically limited. Growing as fisheries crisis spreads	Stronger Use of the Media.
	Regional and global institutions	Central role of the Food and Agriculture Organization of the UN and regional fishery bodies.	Central role of the Food and Agriculture Organization of the UN and regional fishery bodies.
Geographical basis		A process of overlapping and cascading subdivision of the oceans for allocation of resources and responsibilities.	A progressive consideration of larger-scale ecosystems for more comprehensive management, e.g. from specific areas to entire coastal zones and Large Marine Ecosystems (LME).
Stakeholder and political base		Narrow. Essentially fishery stakeholders. Progressively opening to other interests.	Much broader. Society-wide. Often with support from recreational and small-scale fisheries.
Global instruments		1982 Law of the Sea Convention, UN Fish Stock Agreement and FAO Code of Conduct.	Ramsar Convention, UN Conference on Environment and Development and 1992 Agenda 21, Convention on Biological Diversity and Jakarta Mandate.
Measures		Regulation of human activity inputs (gear, effort, capacity) or output (removals, quotas) and trade.	Protection of specified areas and habitats, including limitation or exclusion of extractive human activities. Total or partial ban of some human activities.

Source: (Garcia et al. 2003)

1.2.2. Ecosystem based fisheries management and the precautionary approach

Ecosystem based fisheries management (EBFM) has been defined as ‘an approach that takes major ecosystem components and services - both structural and functional - into account in managing fisheries... It values habitat, embraces a multispecies perspective, and is committed to understanding ecosystem processes... Its goal is to rebuild and sustain populations, species, biological communities and marine ecosystems at high levels of productivity and biological diversity so as not to jeopardize a wide range of goods and services from marine ecosystems while providing food, revenues and recreation for humans’ (US National Research Council, 1998).

Because fisheries and ecosystem management originated for the purpose of managing different ecosystem-types (one being aquatic and less observable than the other), combining them into a single form of EBFM is challenging. Despite the emerging urgency for the management of fisheries to be ecosystem based, the marine environment remains an ‘opaque system’ not easily incorporated into ecosystem management (which requires ‘scientific information as an element in a decision-making process’). Some argue that the complexity of marine ecosystems challenges the ability of science to accurately quantify the impact of man’s activities on ecosystem services; Lauck (1998) posits that ‘the data requirements needed to validate any such model are vastly beyond our current capacity... full understanding and predictability of anything as complex (and, we should add, unobservable) as a marine ecosystem will forever remain a chimera’.

As a result of the uncertainty involved in managing marine ecosystems and evidence of significant declines in fish stocks from over fishing (Ludwig et al., 1993; Pauly et al., 1998; Morato et al., 2006; Pauly et al., 2001; Pauly et al., 2003; Myers and Worm, 2003; Worm et al., 2006) many fisheries scientists stress the need for a precautionary approach (Garcia 1994;

Lauck et al. 1998; Charles 2002; Ludwig 2002; Weeks and Parker 2002). The FAO expert consultation on the Precautionary Approach to Fisheries Management (FAO, 1996) establish some of the key elements of the precautionary approach:

a) It involves the application of prudent foresight, taking account of the uncertainties in fisheries systems and the need to take action with incomplete knowledge;
b) It considers the needs of future generations and the avoidance of changes that are not potentially reversible;
c) It requires prior identification of undesirable outcomes and of measures that will avoid them or correct them promptly;
d) It requires that any necessary corrective measures are initiated without delay, and that they should achieve their purpose promptly, on a timescale not exceeding two or three decades;
e) It requires that where the likely impact of resource use is uncertain, priority should be given to conserving the productive capacity of the resource;
f) It requires that harvesting and processing capacity should be commensurate with estimated sustainable levels of resource, and that increases in capacity should be further contained when resource productivity is highly uncertain;
g) All fishing activities must have prior management authorization and be subject to periodic review;
h) It recommends an established legal and institutional framework for fishery management, within which management plans that implement the above points are instituted for each fishery and appropriate placement of the burden of proof by adhering to the requirements above.

According to Sanchirico et al. (2008), many argue that in the short term a precautionary approach should involve the formation of large-scale protected areas to deal with the inherent risks in complex ecosystems; a form of insurance, where the event to insure against is a stock collapse. And indeed, some equate the formation of marine protected areas, and the precautionary approach *with* EBFM (Essington, 2001; Gerrodette et al., 2002). Despite such calls, there is little guidance on how to operationalise the concepts of EBFM and the precautionary approach in fisheries management (Sanchirico et al., 2008).

1.2.3. An alternate view of the state of fisheries and the need for trade-offs

Ray Hilborn is one of a number of fisheries scientists that offer an alternative view on the status of fish stocks and the sustainability of fisheries globally. During the 1990s and 2000s, suggestions of extreme overexploitation of global fish stocks and irreversible degradation of marine ecosystems received major media attention (Ludwig et al., 1993; Pauly et al., 1998; Morato et al., 2006; Pauly et al., 2001; Pauly et al., 2003; Myers and Worm, 2003; Worm et al., 2006). In a presentation at the 2010 New Zealand Seafood Industry Conference, Hilborn claimed that 'this string of papers has had a very significant impact on [the] publics', journalists' and even most scientists' perceptions about the state of fisheries' (Hilborn, 2010). He argued that many of the studies in question used catch data assembled by the FAO which was a poor proxy for fish stock abundance, since increases in catch restrictions would appear in analyses to be stock declines. Hilborn believed that calls for extreme conservation measures in fisheries, from the public, journalists and scientists were based on exaggerated estimates of stock over-exploitation.

Collaborating with many of the authors whose findings he had initially challenged, Hilborn used scientifically designed surveys and fisheries stock assessments from around the developed world (as opposed to catch data) to estimate global stock abundance (Worm et al., 2009). The study joined 'previously diverging perspectives to provide an integrated assessment of the status, trends and solutions in marine fisheries' (Hilborn, 2010). The findings indicated that for 5 of 10 well-studied ecosystems, the average exploitation rate had recently declined and was at or below the rate predicted to achieve maximum sustainable yield for 7 systems. While the results of the study suggested that global fisheries were not as overfished as previously argued, they also suggested 63% of assessed fish stocks worldwide required rebuilding, and even lower exploitation rates were needed to reverse the collapse of vulnerable species.

The study recommended combining diverse management measures such as catch restrictions, gear modifications and closed areas to reduce exploitation sufficiently. This would involve reducing exploitation rates to levels below those producing maximum sustainable yield (MSY), resulting in fewer depleted species while still allowing fisheries to operate near maximum economic yield but with less environmental impact. Where fishing activities were excessively destructive on sensitive marine habitats, area closures were recommended along with the need for technological development to reduce the by-catch of sensitive species. While the concluding solutions and recommendations were thus similar to the studies Hilborn had criticised, the paper offered a more tempered view of states and trends in fisheries and the urgency/extremity of management measures required to address problems. In particular, the study addressed the reality that trade-offs exist in the management of marine ecosystems.

Hilborn (2010) states, 'fishing is always going to impact the environment and we simply have, as a society, to accept it; we can't take food from the ocean without reducing biodiversity'. Hilborn is addressing the reality of trade-offs in managing marine ecosystem services. In reality, finding the level of fishery exploitation most desirable for society hinges on determining the level of marine biodiversity and habitat degradation society is willing to sacrifice to procure the food provisioning services of marine ecosystems. Thus, the interpretation of EBFM in the academic community spans two opposite spectrums; in one is the idea that marine ecosystems are heavily under threat from fishing activities and therefore management responses must be swift and extreme (Pauly et al., 2003; Myers and Worm, 2003; Worm et al., 2006). In the other is the idea that in many cases fisheries are on a sustainable course and while further measures to reduce environmental impacts are required, this trade-off in ecosystem services is part and parcel of ecosystem management (Hilborn, 2010).

1.2.4. Integrated and behavioural fisheries management approaches

While precautionary measures have the potential to reduce the risk of irreversible damage to the marine ecosystem and are sometimes essential for conservation of sensitive species and habitats (Garcia, 1994; Lauck et al., 1998; Ludwig, 2002), top-down management decisions that reduce the revenues of the fishing community are likely to be detrimental to stakeholder-management relations and to jeopardize the willingness of stakeholders to cooperate with new legislative measures (Hilborn; 1985; Jones et al., 2014). Some argue that for EBFM to achieve societal objectives it must incorporate an integrated management (IM) approach(e.g. Gilman, 2001; Done, 1998). The term integrated management 'implies the use of a collaborative/participative approach involving the main stakeholders in a flexible, responsible and transparent planning process, respectful of existing rights and duties' (Garcia, 2003). Furthermore it is based on ecosystem orientated objectives and the precautionary approach

(Garcia, 2003). Its importance is noted in a recent report to the European Commission Directorate General for Fisheries and Maritime Affairs which states: ‘the buy-in of the fishing sector to new management initiatives is expected to be strongest where the sector has been actively involved in developing the evidence base and where scientific data has been open to scrutiny’ (Anon, 2010).

In addition to IM, Grafton et al. (2006) argue that by itself the ecosystem approach is not sufficient to address inappropriate incentives bearing on fisher motivation. Clark (2006) observes that most fishery failures are fully predictable on the basis of simple economic principles and Hilborn (2011) claims that most fisheries problems arise from a failure to understand and manage fishermen, and that study of the dynamic behaviour of fishermen should be a major part of fisheries research. Hilborn (1985) suggests that the collapse of many fisheries can best be explained as the result of misunderstanding fisher behaviour, rather than a lack of knowledge of fishery resources. Like Hilborn, others (Hanna and Smith 1993; Wilen et al. 2002) stress the need to understand the nature of fishers’ operations and responses to regulation or other stimuli in relation to their preferences in order to develop efficient management schemes.

In large part, fisheries policy to date has been based on conventions developed in the field of fisheries economics and capital theory from the 1950s onwards (e.g. Gordon, 1954; Clark and Munro, 1975; Munro 1979). Such work has predominantly been concerned with attempting to determine the “optimal” harvest rate of a fishery using a bioeconomic modelling approach. The problem with this form of management is that it assumes policy makers and fishery managers can successfully control fishery inputs and outputs to achieve desired targets. However, repeated failures to constrain fishery inputs and outputs and account for the response of fishermen to regulation have led to overfishing and unforeseen negative environmental impacts of behavioural responses to changes in protective legislation (Salas and Gaertner, 2004; Grafton et al., 2006; Clark, 2006; Hilborn, 2011).

If fishery management decisions do not incorporate fishers' behavioural responses to regulatory changes, then the unpredicted ecosystem impacts that arise from these changes in human activity may undermine EBFM initiatives. Gaertner et al. (1999) suggest that long-term models, commonly used in fisheries assessment, rarely capture the rapid, short-term changes generated by the decisions that fishers make about where, when and what to fish for. Therefore the short term responses of fishermen to regulatory changes and the ecosystem impacts that arise from these responses are often unknown when new regulations are implemented, since managers generally make simplistic assumptions about fishers' nature and attitudes when defining management policies (Salas and Gaertner, 2004).

As a result, behaviourally oriented fisheries research and the use of discrete choice econometrics to investigate fishermen's choices have gained in popularity (e.g. Mistiaen and Strand, 2000; Smith, 2005; Ward and Sutinen, 1994; Larson et al., 1999; Valcic 2008; Smith et al., 2008; Eggert and Tvertas, 2004, and many others). In this vein of research, economists investigate the influence of economically relevant factors and the impact of policy changes on fishermen's choice(s) across a finite number of discrete options which fishermen have available to them. This avenue of analysis is also pursued in this thesis where in chapter 5 I investigate the spatial fishing decisions of vessels in the Irish demersal otter trawl fishery using the random utility model and Vessel monitoring data for the entire fleet.

1.2.5. Coastal Management

As concepts of ecosystem based management developed from the 1960s onwards (e.g. King, 1966; Helliwell, 1969; Hueting, 1970; Odum and Odum, 1972), so too did the idea that such concepts required a specialised application for coastal zones. These ideas evolved into the concept of integrated coastal zone management; (ICZM) began to emerge in the scientific and environmental literature of the 1970s (Billé, 2008) and achieved international political acceptance in the Rio Earth Summit of 1992. Its development reflected a growing awareness of the need to take an integrated and ecosystem based approach the management of coastal areas:

‘ICZM aims for the coordinated application of the different policies affecting the coastal zone related to activities such as nature protection, aquaculture, fisheries, agriculture, industry, off shore wind energy, shipping, tourism, development of infrastructure and mitigation and adaptation to climate change. It will contribute to sustainable development of coastal zones by the application of an approach that respects the limits of natural resources and ecosystems, the so-called 'ecosystem based approach'.

In one respect, the merger of coastal and ecosystem management is more straight-forward than is the case for fisheries; terrain-based coastal ecosystems and the immediate coastline are easier to manage in the spatial, environmentally controlled historical format of land based ecosystem management. However, coastal ecosystems still exhibit high levels of spatial and temporal heterogeneity in their habitat and biological characteristics, and these latter characteristics tend to be very sensitive to human activities (Koch et al., 2009; Charton and Ruzafa, 1999). High population density, diverse resource use and incongruent standalone management policies make sustainable management of coastal ecosystems an equally complex task. In addition, the lack of sector-wide organisation and coherency in law, management, regulation and control in the coastal zone further complicates matters.

1.2.6. Summary of Ecosystem and Fisheries Management Concepts.

Overall, there is consensus amongst policy makers and scientists that an ecosystem based approach to management of marine and coastal environments is required. However, because of an entirely different historical basis and trajectory of development (see Fig 1.1), marrying fisheries and ecosystem management into a single format is a complex process, requiring experimental approaches in both policy application and scientific research. Already in the debate among fisheries scientists, there are divergent ideas about the degree of overexploitation of global fish stocks and what is an acceptable trade-off between food provisioning and other marine ecosystem services. While EBFM and a precautionary approach are proposed by many, there is little guidance on how to operationalise these concepts. Integrated and adaptive fisheries management, based on the behavioural dynamics of fishers, is argued by some as key to the success of future fisheries policy. To this end, research into behavioural responses to changes in management regulations has become a burgeoning field of fisheries economics (e.g. Mistiaen and Strand, 2000; Smith, 2005; Ward and Sutinen, 1994, and many others).

The situation for coastal ecosystem management is equally complex, albeit for different reasons. Integrated coastal zone management has emerged as the primary policy mechanism by which ecosystem based approached to coastal management are evolving. However, ICZM is currently a concept only and like EBFM, there is little guidance on how it can be applied or whether its principles form a coherent policy instrument that can assist in overarching coastal management policy (Billé, 2008).

1.3. Overview of Research Objectives

Incorporation of ecosystem management into policy and development is a recent affair and due to the complexity of ecosystem and human-behavioural dynamics there is as of yet no real consensus on what form it should take. The challenge is even more pronounced for the management of marine and coastal ecosystems which are less observable and exhibit greater spatial and temporal habitat and species heterogeneity than terrain-based ecosystems. This creates great demand for increased scientific understanding of these systems, but also for research into the dynamics of human interaction with marine and coastal ecosystems as well as management methodologies that can contribute to an ecosystem based approach. Thus the overarching objective of this thesis is to add to the growing literature on the incorporation of novel ecosystem based approaches to marine and coastal management, and through these approaches, to identify pitfalls and critical barriers to socially optimal management.

To that end, the research has three main goals, which are addressed in three separate empirical papers. Specifically, the research aims to analyse:

1. The species-targeting behaviour of fishermen in a multi species fishery;
    - Do single-species quota changes effect fishermen's species-targeting decisions in a multi-species fishery?
    - What are the wider ecosystem/multispecies impacts, and how can incorporating behavioural dynamics help to prevent negative/unforeseen outcomes?

2. The strategic and local principles of ICZM;

- Given disparate scientific understanding of coastal ecosystems, do scientists and local stakeholders exhibit different attitudes towards coastal ecosystem services and their sensitivity to human activities?
- To what extent do differences in scientific understanding undermine optimal coastal zone management?

3. Fishermen's spatial choices and displacement
   - How do changes to multiple-species quotas in a mixed fishery impact fishermen's fishing location choice?
   - What repercussions does fishing displacement have for ecosystem based goals, such as by-catch and discard reduction and habitat conservation?

Novel methods are used to achieve this and in two cases, portfolio theory, a methodology routed in financial economic research, is adopted to explicitly incorporate the trade-off between risks and benefits in decision-making processes. The second research objective is addressed using primary data collected from marine and coastal scientists and local stakeholders, specifically for this thesis. For the third research objective, this thesis demonstrates the first application of econometrically and spatially orientated discrete choice methods to satellite-based vessel-location data. The single theme that binds all three research pieces is their focus on the dichotomy between ecosystem management objectives and stakeholder attitudes and behaviours and the challenge this poses for sustainable marine and coastal management. The specific objectives of this thesis, as they are addressed in the individual papers, are outlined in more detail in the following paragraphs.

1.4. Structure of thesis

The main objective of this thesis is to examine the potential use of novel ecosystem based approaches for application to a number of different marine and coastal management scenarios. With respect to fisheries, this is done by applying either optimisation or econometric techniques to Irish fisheries data collected by the Central Statistics Office of Ireland, the Irish Navy and The Irish Sea-Fisheries Protection Authority. For research focusing on coastal ecosystem management and ICZM, primary survey data collected from marine scientists and local stakeholders is used. The main body of the thesis consists of three separate empirical papers outlined above and begins with a chapter focused on marine and coastal management policy (Chapter 2).

The first paper (Chapter 3) uses portfolio theory and the expected utility hypothesis to predict the impact of single-species quota changes on a fishing fleet's species-targeting behaviour in a multispecies fishery. The paper begins with an introduction to the research area which is followed by a literature review of earlier applications of portfolio theory to fisheries research. In the methodology section, portfolio theory and the expected utility hypothesis are outlined and an explanation of the calculation of predictive variables in the fleet's objective function is provided. Following this, there is a description of Irish mixed species fisheries and the data used in the analysis. The penultimate section of the chapter presents the results of the species-portfolio optimisations before any quota constraints are included and compares the risk-revenue profile of these portfolios to the profile of the actually observed harvest portfolios in the data-set; this gives an indication of how accurate an approximation of the fleet's objective function the specified model provides. Following this, results of the species-portfolio optimisations given the quota simulations are presented and critiqued to assess the impact of the quota changes on the fleet's multispecies targeting behaviour. The final section of the paper provides a brief discussion and conclusions.

The second paper (Chapter 4) addresses incoherence in the principles of integrated coastal zone management (ICZM) through an application of biome portfolio analysis (BPA). The paper begins with a review of ICZM and the criteria that management tools such as BPA must satisfy. Following this, the methodology section provides a more in-depth description of BPA and describes how it can be employed to assist in coastal management decision making and also to measure an 'attitude gap' between those identifying with either the strategic or local principles of ICZM. After this, the paper describes the coastal study site and the list of biomes (geographic habitats), services and threats that were selected for the analysis. There is also a description of the survey structure, the stakeholder group and the nature of the stakeholder workshops which were used to collect the survey data. As this piece of research is based in a spatial analysis, the methodology section also includes a description of the GIS data used to represent the habitat types of the area. This is followed by the results and analysis section which firstly presents the results of the stakeholder workshops, then the results of the scientific consultations and finally a comparison of both. The final section of the paper offers a discussion of the results with respect to their connotations for the ICZM principles, and the concluding remarks of the study.

The third paper (Chapter 5) deals with the economics of spatial choice in Irish demersal otter trawl fisheries and the issue of displacement when multispecies quota changes arise. It begins with an introduction to recent changes in European fisheries policy and a background of the Irish demersal otter trawl fleet. The literature section then summarises behaviourally orientated fisheries research, in particular, discrete choice models in fisheries economics research. Following this, the data used in the analysis, particularly the Vessel monitoring systems (VMS) data, is described in detail. Next, the methodology section outlines various discrete choice random utility models which are applicable to the research question and explains why the conditional logit was the eventual model selected to analyse fishing location choice in this fishery. After this there is a further section on methods that explains how model variables that are used as regressands in the conditional logit analysis are calculated. Furthermore, this section describes the methodology by which the VMS data was used with logbook catch data to create the distinctly discrete fishing location alternatives used in the analysis. The results and analysis section presents the resulting coefficients of the conditional logit model, and then compares the ability of this model, based on the estimated coefficients, to accurately predict site-visitation for the fisheries under analysis. This is done by comparing the actual site visitation percentages in the data set to the model's predicted percentages before any additional quota constraints have been included. After this, the impact of the quota constraints on the model's predicted site-visitation percentages is presented and the results are discussed. The final section provides a discussion and conclusion of the paper.

Chapter 6 is the concluding chapter of the thesis and offers a discussion of the thesis's core findings and concluding remarks.

1.5. Thesis Outputs

The research carried out in this thesis has led to a number of outputs. In particular, there have been two academic publications in peer reviews journals. The research has also been presented at several international conferences, seminars, courses and workshops over the life of the PhD. The more important research outputs are listed below.

1.5.1. Papers

- ‘The impact of precautionary quota constraints on the composition of multispecies harvest portfolios’, submitted to the *Journal of Ecological Economics* (awaiting feedback).

- ‘Shortcomings in the European principles of Integrated Coastal Zone Management; assessing the implications for locally orientated coastal management using Biome Portfolio Analysis’ *The Journal of Marine Policy*, (2014).

- ‘The impact of quota changes on the spatial decisions of fishermen in Irish demersal otter trawl fisheries’ (working paper for SEMRU series).

- ‘Addressing failures of the Common Fisheries Policy: How principles of devolved community-based governance may improve the sustainability of Irish fisheries’, *The Irish Review of Community Economic Development Law & Policy*, (2012).

1.5.2. Presentations

- ‘Discrete fishing site choice using VMS data’, *2nd International Conference on Fishery Dependent Information*, Rome, Italy, March 2014.

- ‘Incorporation of financial risk management tools into the management of ecosystem services’,
    1. 87th *Agricultural Economics Society Conference*, Warwick, England, 2013;
    2. *Rural Economy Research Centre*, Teagasc, Athenry, Ireland, February 2013.

- ‘Applying portfolio theory to the management of fish stocks’,
    1. *Centre for Fisheries and Aquaculture Management and Economics*, PhD workshop, Esbjerg, Denmark, January, 2011.
    2. 2nd Annual Beaufort Marine Socio-Economic Symposium, Galway, Ireland, November 2010.

2. Policy Background

This chapter covers EU policy as it relates to freshwater, coastal and marine environmental management. Such EU Directives and policies began to arise in the early 1970s and are constantly being created, adapted and improved upon even up until the present time. These policies are important to understand because apart from shaping the way in which the culture of marine and coastal natural resource management in Europe has evolved, there is an ongoing effort to increase the capacity of such Directives and policies to operate more efficiently. EU environmental policy is thus highly developmental and no analysis of coastal and marine management issues would be complete without taking stock of which stage such policies are in, in their ongoing development. This policy review is also relevant to the following chapters because it contextualises the empirical studies of the thesis; each of the three methodologies employed in this thesis were selected and adapted according to the relevant policy structures. For example, and as will be discussed in the following sections of this chapter, recent policy has focused on multispecies, spatial and integrated forms of management. All three empirical studied demonstrated in this thesis are designed to accommodate such policy goals and work with such objectives in mind.

The chapter begins with an overview of the challenges that environmental policy has faced in its application to coastal areas and marine and freshwater systems. Section 2.2 treats of the earliest environmental policy in this area and how these Directives have evolved into the Water Framework Directive. Section 2.3 discusses the Marine Strategy Framework Directive while section 2.4 discusses the somewhat related Common Fisheries Policy. Following this the Habitats and Birds Directives are described in section 2.5, as well as how these two fit in with the general frameworks of the Water Framework Directive and the Common Fisheries Policy. Section 2.6 provides details on European and Irish socioeconomic statistics and then discusses coastal habitats and EU coastal management policy and Maritime Spatial Planning. Section 2.7 discusses the Integrated Maritime Policy. The chapter concludes with a brief summary.

2.1. Overview

Coastal, marine and freshwater habitats are quintessentially heterogeneous across space due to variation in geomorphic and biogeographic features as well as human population density (Costanza, et al., 1997). In addition to this, the marine environment, coastal areas and aquatic systems provide a highly diverse set of ecosystem services to society. On one level, they provide the types of non-marketable services described in Chapter 1. Human activities like industrial manufacturing, construction, tourism, waste disposal and the simple act of populating and subjugating a space of land undermine the sustainability of such services and the spaces from which they arise. Much of the environmental policy discussed in the following sections is designed around the protection and sustainability of such non-marketable ecosystems and services. On another level however, policy makers cannot ignore the fact that EU coastal areas are also some of its most heavily populated, and in addition to desiring the conservation of non-marketable ecosystem services, such populations also require the very amenities and economic activities that undermine them. The fact that coastal regions, economies and marine industries can sustain dense populations is an ecosystem service in and of itself. The challenge this poses for society and for policy makers then, is to develop a coherent policy framework that allows the myriad of ecosystem services, marketable or not, to be optimally traded off against each other and to be managed efficiently. However because coastal and marine areas exhibit such heterogeneity across space and because the variety of demands for ecosystem services are diverse, EU environmental policy in this area has tended to be disjointed and incoherent. As a result, individualistic management initiatives have originated with only one or a small number of sectors in mind, from miscellaneous, cross-border disconnected state agencies (EC, 2001).

At the policy level, this disjointedness translated into numerous environmental policies designed for specific environmental problems, but which nevertheless showed a large degree of repetition in terms of environmental goals and operational concepts (EC, 2001). Over the years the crossover between Directives allowed for further integration of EU coastal and marine management policy so that various environmental Directives and policies were amalgamated into a more coherent set, intended to work synergistically to achieve various environmental, social and economic targets. The following sections discuss the relevant EU Directives and policies in this area, their development and current status and how they relate to each other.

2.2. EU water policy

In 1973 the first of several five-year Environmental Action Programmes (EAP) introduced the objectives and principles of the environmental policies of the European Commission (EC, 2006). Over time, several regulatory instruments designed to help prevent water pollution were introduced. For example, the surfaces water Directive (75/440/EEC), the bathing water Directive (76/160/EEC), the fish water Directive (78/659/EEC), the shellfish water Directive (79/923/EEC) and the drinking water Directive (80/778/EEC). Early Directives treated aquatic ecosystems as individually protected commodities and targets were set individually, rather than across space. In the 1990s, continued problems with pollution in EU waters, in particular eutrophication of sea and fish waters, led to the adoption of two new legal instruments which set strict rules on the treatment of waste water and the use of nitrates in agriculture; the Waste Water Treatment Directives (91/271/EEC and 98/15/EEC) and the Nitrate Directive prevention and control (96/61/EC) (EC, 2006). In 2000, in an effort to harmonise the various water-related Directives that had arisen over the previous three decades, the Water Framework Directive (WFD) was adopted and since then has been the main operating version of EU water policy (2000/60/EC). The Directive applies to all aquatic systems, surface waters (rivers and lakes), groundwater and coastal waters, bundling many of the individual Directives developed in previous years to promote a more coherent set of policies that allow aquatic resources to be managed across national and geographical boundaries in an integrated fashion (EC, 2006). The focus of the WFD is on establishing a number of key measures of resource status and maintaining and improving the quality of the aquatic environment as it is measured via these indicators. Indicators include variables like water quality, ground water quality and surface water quality. A water body is said to be of good status when both its ecological status and its chemical status are at least "good". Ecological status is an expression of the quality of the structure and functioning of aquatic ecosystems associated with surface waters whereas the chemical status level must not exceed the environmental quality standards established by the Directive and other relevant Community legislation setting environmental quality standards at a Community level (2000/60/EC). As the Directive states, the ultimate aim is to achieve the elimination of priority hazardous substances and contribute to achieving concentrations in the marine environment near background values for naturally occurring substances. Furthermore the Directive acknowledges the connectedness of inland freshwaters with transitional and coastal waters and states that effective and coherent water policy must take account of the vulnerability of aquatic ecosystems located near the coast and estuaries or in gulfs or relatively closed seas, as their equilibrium is strongly influenced by the quality of inland waters flowing into them. While reviewing the WFD in its entirety is beyond the scope of this chapter, the individualistic Directives of early EU water policy which treated water bodies as separate entities has been replaced by a unifying water Directive which recognises the interconnectedness of waterways both inland and along the coast. It is also important however for the objectives and operationalising concepts of the WFD to integrate with policy relating to other policy initiatives, which are discussed below.

2.3. Marine Strategy Framework Directive

The Marine Strategy Framework Directive (MSFD) was adopted by the European Commission in June 2008 and similarly to the WFD, aims to achieve good environmental status (GES) of the EU's marine waters by 2020 (EC, 2008). GES differs somewhat to the standards of good status set out by the WFD however. It has been defined by the commission as existing when 'the different uses made of the marine resources are conducted at a sustainable level, ensuring their continuity for future generations' (EC, 2008). It also requires that:

- 'Ecosystems, including their hydro-morphological (i.e. the structure and evolution of the water resources), physical and chemical conditions, are fully functioning and resilient to human-induced environmental change;
- The decline of biodiversity caused by human activities is prevented and biodiversity is protected;

- Human activities introducing substances and energy into the marine environment do not cause pollution effects. Noise from human activities is compatible with the marine environment and its ecosystems' (EC, 2008).

In 2010 the European Commission produced a detailed set of criteria and indicators to guide Member States on the means through which to implement the MSFD (EC, 2010). The report focused heavily on the prioritisation of biological biodiversity through measures of the quality and occurrence of habitats and the distribution and abundance of species. Other indicators included in the guidelines were measures of non-indigenous species within marine ecosystems, level of fishing pressure, eutrophication levels, sea floor habitat integrity, measures of contaminants in marine environments and measures of marine litter. The core message of the Directive was that management of the marine environment should follow an ecosystem approach based on environmental protection and sustainability. Beyond following the guidelines and achieving the indicator targets for GES 2020, Member States were responsible for the development of their own marine strategy. In Ireland, this responsibility has been assigned to the Department of Environment, Community and Local Government (MI, 2012). Member States are expected to undertake an analysis of the features or characteristics of, and pressures and impacts on, their marine waters, identifying the predominant pressures and impacts on those waters, and an economic and social analysis of their use and of the cost of degradation of the marine environment. Member States should then establish and implement programmes of measures which are designed to achieve or maintain good environmental status in the waters concerned, while accommodating existing Community and international requirements and the needs of the marine region or sub-region concerned (EC, 2008).

In addition to the establishment and maintenance of environmental targets and associated indicators of good environmental status, the MSFD also requires that following the initial environmental and socioeconomic assessment carried out by each Member State, any intended management measures should give due consideration to sustainable development and, in particular, to the social and economic impacts of those measures (EC, 2008). Member States are also required to carry out impact assessments and cost benefit analyses to determine whether new measures are cost effective and technically feasible.

While not all of the guidelines on the MSFD are focused on fisheries, the issue of habitat degradation is particularly relevant to Chapter 5 of this thesis, which looks at the fishing location decision of fishermen in the Irish demersal otter trawl fleet. This form of fishing is associated with substantial habitat damage along the sea bed, as the trawling method causes the fishing gear to make contact with the ground. However, different types of sea bed are affected in different ways. For example nephrops fishing takes place along sandy bottoms, and therefore has little to no habitat impact, while fishing for species with a more vegetative of coral based habitat is far more destructive. Since the empirical analysis of chapter 5 is concerned with predicting changes in fishing site choice following quota changes, it is possible that this information could indicate to fishery managers where displaced and reallocated fishing effort will be directed, and with that information, to develop a better understanding of what the habitat impact of quota changes will be.

2.4. Common Fisheries Policy

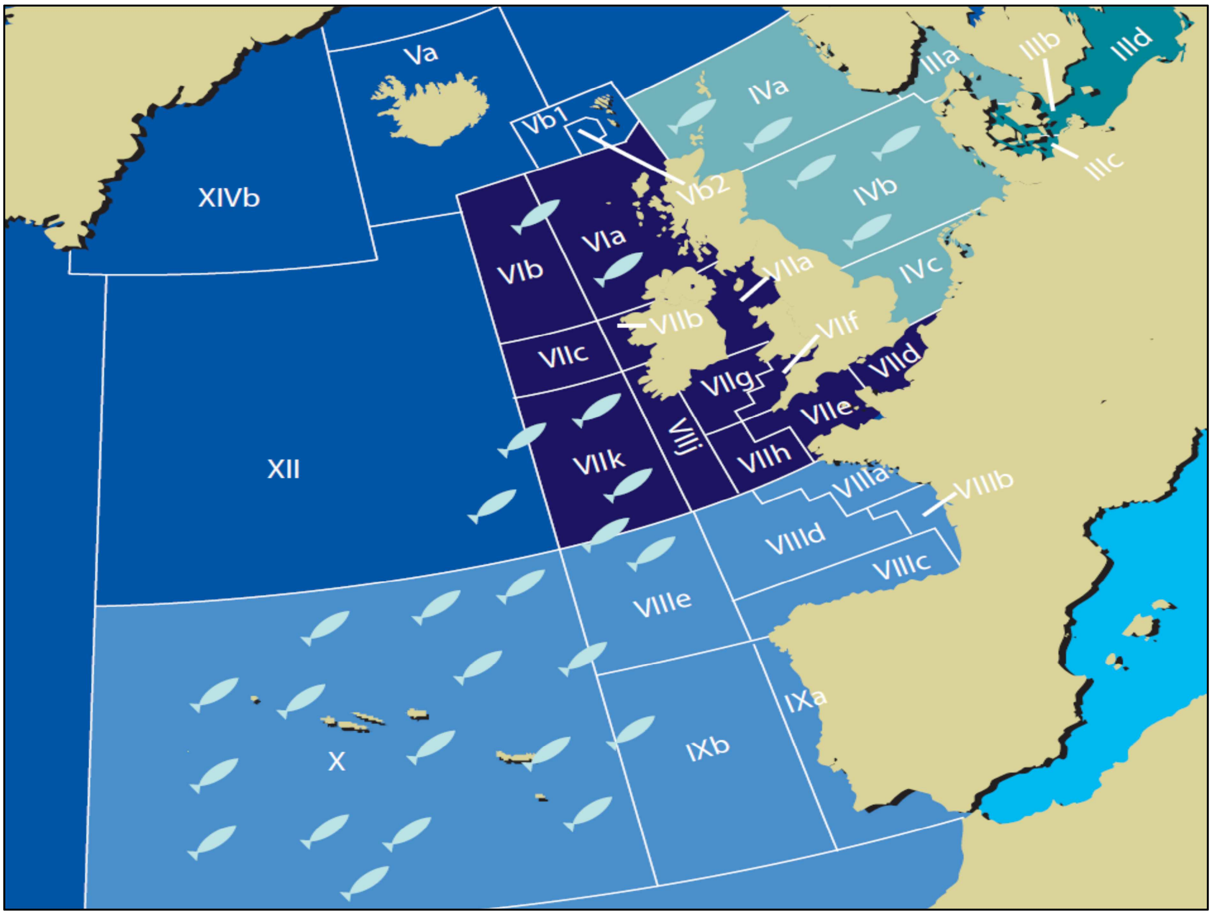
Given the fact that one of the major indicators of GES 2020 under the MSFD is fishing pressure levels in EU marine waters, it is clear that implementation of the MSFD has major connotations for the EU fishing sector. In addition to the level of fishing pressure, other fishery related indicators of GES include the reproductive capacity of fish stocks as well as their population age and size distribution. Since the main policy Vessel used to manage fisheries and improve these very indicators of a fishery's status within the EU is the Common Fisheries Policy (CFP), the MSFD will be required to operate alongside CFP legislation. Indeed, it is likely that only through a successful application of the recent reforms of the CFP that the GES 2020 targets of the MSFD may be realised.

Irish fishing waters are currently governed as part of the CFP according to Council Regulation (EEC) no 170/83 (EC, 2012a). The CFP is a collaborative effort by all EU Member States to ensure the sustainable governance of EU fisheries (EC, 2012a). The CFP tries to ensure sustainable fishing practice by setting allowable levels of catch (Total Allowable Catch (TAC)), limiting the number of days at sea (fishing effort), restricting the use of certain fishing gear (Technical Conservation Measures (TCM)) and reducing overcapacity in the EU fishing fleet (through fleet decommissioning) (EC, 2012a). Total Allowable Catch (TAC) levels are set for each EU fishing zone (Figure 2.1). The procedure for carrying this out is provided for by Council Regulation (EEC) no 170/83 of 25 January 1983 establishing a [European] Community system for the conservation and management of fishery resources. This document outlines the procedure for conserving fishing resources in Fisheries of the EU. According to this document, when a limit is to be placed on catch, this limit must be shared out amongst each Member State. This is carried out at an EU level. Each Member State then has the responsibility to apportion that share amongst its fishers. The share allocated to each Member State is done in a manner that ensures stability of shares relative to historical levels of catch. Specifically, Article 4 states that 'the volume of the catches available to the [European] Community referred to in Article 3 shall be distributed between the Member States in a manner which assures each Member State relative stability of fishing activities for each of the stocks considered'. Thus, the CFP must delineate an ecological necessity of restricting effort and does so by Member State. Article 4 outlines the process by which this is carried out, which recognises each Member State's tradition of fishing in a given region and apportions the Total Allowable Catch based on this[3].

Figure 2.1 illustrates the fishing zones within the EU. It should be noted that Council Regulation (EEC) No. 3760/92 establishing a Community system for fisheries and aquaculture states that the CFP extends to all fisheries of the EU, including those immediately adjacent to a nation's shoreline. Within the TAC set for each fishing zone a share is assigned to each nation's fishing fleet. This is known as a quota. All fishing vessels within a nation are required to have a licence to fish according to this quota. However, restrictions exist for what vessels may fish in each area when fishing this quota. Under both Council Regulation (EEC) No.170/83 and Council Regulation (EC) No. 2371/02, it has been established that the area within the 12 nautical mile limit of coasts (about 22km), is reserved for local fishermen and small fishing boats from other European countries that have traditionally fished in these areas. From 12 to 200 nautical miles, access is open to all EU boats, with International waters beyond that. However, the EU fishing areas outlined in figure 1 may cross many of these boundaries and thus TACs are applicable to all fishing waters right up to the shoreline. As such, the current TAC scheme limits the overall catch allowed per region, whilst the national quota scheme limits fishing effort by nation (EC, 2012a).

[3] It should be noted that although the CFP delineates fishing effort amongst Member States, it does not affect the freedom of movement of fishers to catch a portion of a given Member State's quota. To illustrate, Irish fishing vessels are licensed according to the Fisheries (Amendment) Act 2003, with the determinants for licencing as a member of the Irish fleet being predicated on whether a vessel "*is wholly owned by a national of a Member State or a body corporate established under and subject to the law of a Member State and having its principal place of business in a Member State"* (Fisheries (Amendment) Act, 2003 no. 23 pt2, s4"). This regulation is in accordance with EU freedom of movement legislation, as provided for by Article 45 of the Treaty on the Functioning of the European Union. Under this legislation, nationals of any EU Member State may apply for licensing as part of the Irish fleet. To understand this interaction, it is helpful to think of the quota allocation as a given country's natural resource and labourers are free to move between Member States to harvest that resource.

Figure 2.1: Fishing Areas in EU waters



Source: EC, 2012a

The motivation for an EU-level system stems from the migratory nature of many fish species, whereby fish cross many political jurisdictions during their lifecycle. For example many pelagic species like Mackerel and Herring migrate from southern regions of the Atlantic such as the coast of Spain in winter months, towards cooler northern regions such as the Norwegian and Icelandic coastlines as the temperature of the Atlantic rises (Uriarte and Lucio, 2001). When this happens, governance at the national or sub-national fishing community level[4] may be inadequate as sustainable practice in one jurisdiction may be undermined by unsustainable practice in another (Ebbin, 2004). In such circumstances, the local fishing community-level or the national-level does not match the international

[4] When referring to community-level in the context of fisheries governance, I refer to the sub-national, localised group of resource users. This community in a specific localised area shares the common use of a particular coastal fishery.

geographical scale of the migratory species in question (Berkes, 2006) and an international, centralised system of governance such as the CFP is required.

Both chapters 3 and 5 present an empirical study with a fisheries focus. Each methodology incorporate the needs for a multi-species approach to fisheries management and takes account of fishermen's behavioural responses by use of a mathematically quantitative objective function. Chapter 3, which demonstrates a portfolio approach to fisheries management specifically treats of risk in fishermen's objective function, while chapter 5 incorporates more of a spatially orientated approach.

2.5. Habitats and Birds Directives

Two further pieces of environmental legislation, specifically the Habitats Directive (EC, 1992) and the Birds Directive (EC1979) (HBD), by virtue of their focus on the conservation of natural habitat across EU coastal, marine and freshwater systems, will also have a bearing on other Directives pertaining to this area. In particular, the aim of these two Directives are supportive of the aims of the MSFD in that their environmental measures (favourable conservation status (FCS) and status of population (SP)) have a similar emphasis on biodiversity and sustainability indicators as the GES measures of the MSFD. All three Directives also have proactive as well as reactive elements in that they are not only concerned with protection, maintenance and management of specific elements of biodiversity but also the restoration and recovery of habitats and species, where possible (EC, 2012b). With this said, there are also some significant differences between the HBD and the MSFD, in particular the fact that the MSFD is more geared towards an overall ecosystem based management approach (EC, 2012b); The MSFD has a much broader material scope in that it aims to, inter alia, achieve and maintain GES, which includes all marine biodiversity whilst HBD focus on the conservation of particular habitats and species. In particular, the HBD have two main undertakings:

- To protect habitats as well as species and their habitats (e.g. breeding, feeding, resting, staging sites) for 193 bird species listed in Annex I of the Birds Directive and for regularly occurring migratory birds, and for species/habitat types listed in Annexes I and II of the Habitats Directive by designating protected areas.
- To establish a system of species protection for all wild birds in the EU and for species listed in annex IV (strict protection) and annex V (subject of exploitation) of the Habitats Directive (Arts 5-9 BD, Arts 12-16 HD). This protection regime applies across their entire natural range within the EU i.e. both within and outside protected areas and may include measures to ensure exploitation and taking in the wild is compatible with maintaining them in a FCS.

The designated sites which the HBD refer to are determined through the formation of a system of protective areas known as the Natura 2000 network, an EU wide marine, coastal and inland network of sites (EC, 2014a). The network is formed through the establishment of special areas of conservation (SACs) which have been designated under the 1979 Birds Directive (EC, 1979). In 2007 the European Commission proposed guidelines for the establishment of the Natura 2000 network in the marine environment (EC, 2007). These guidelines outline plans for a system of marine protected areas (MPAs) and elaborate upon the means by which these MPAs might assist in achieving the targets of MSFD for 2020 as well as how such a network would fit into the broader context of a new EU maritime policy and CFP. While not legally binding, Commission services updates to the 2007 guidelines (which allow for regular adaptive guidelines to be made) state that it is the responsibility of Member States to designate protected areas in the marine area and to establish conservation measures and that in order to maintain or restore the conservation status of relevant habitats or species, Member States should assess if there is a need for fisheries management measures (EC, 2014b). This is important because it shows the level of crossover between the various Directives in play; in this case the CFP and the HBD.

The spatial component of the empirical analysis employed in Chapter 5 relates strongly to the issue of marine protected areas and special areas of conservation. While chapter 5 focuses on the impact of quota simulations, as opposed to the impact of forming a single MPA, the reallocation of fishing effort that arises from quota changes could impact key conservation sites for habitat, birds and other wildlife. The results of the study suggested significant displacement and reallocation of fishing effort in the Irish demersal otter trawl fishery following quota changes, and thus the impact of quota changes is indeed a highly relevant issue for the birds and habitat directives and indeed any set of directives focused on the formation of special areas of protection for wildlife and habitat in marine areas.

The MSFD is also relevant in this context as the Commission document also encourages Member States to ensure a good coordination between fishery and environmental authorities at Member State level or stakeholder level, which in turn, will alternate depending on the jurisdiction in which the Natura 2020 site is located. Within 12 nautical miles of the Member States' coast, the Member State can take non-discriminatory measures to minimise the effects of fishing on the conservation of the marine ecosystem. The exception is if these measures are liable to affect the vessels of other Member States, in which case these proposed measures can only be adopted after the Commission, other Member States and the Regional Advisory Councils (RAC) concerned have been consulted on a draft of the measures. If the Network 2020 site is located offshore (>12 nautical miles from the shoreline), then the proposed measures fall under the scope of the CFP and Member States would have to address a formal request of adoption of such measures to the Directorate General of Fisheries and Maritime Affairs (DG MARE) of the Commission (EC, 2014b).

The above shows the degree to which different Directives operate within different policy spheres, yet show substantial crossover and must be operationalised coherently in order for EU and MS environmental management to be effective. Clearly, orchestration of various Directives in any kind of harmony is a dynamic and complex affair. In general, the HBD relates to all spheres of the natural environment at land and at sea, but is specific to key sites and species only. The MSFD, relates only to the marine environment, but to all species, habitats and human activities in a type of ecosystem based approach. Where a Natura 2020 sites fall within the spatial range of the MSFD, the goals of HBD can contribute to the goals of the MSFD and vice versa. However the way in which this occurs and the Directives through which it occurs will alternate depending on distance from the shore. Further out at sea fisheries management measures begin to fall under the exclusive competence of the CFP. In this way, even if it relates to the HBD or MSFD, any conservation effort requiring fisheries measures cannot be achieved without working through the channels of the CFP. Further inshore, (within 12 nautical miles), measures employed by Member States to manage fisheries need not be determined by CFP legislation, yet reference to stakeholders and other environmental regulation such as the HBD and MSFD is paramount. If we consider Natura 2020 sites that are inshore or inland, i.e. fall within ‘coastal areas’, then we are now entering the sphere of EU coastal management Directives or even water Directives as they relate to fresh water river and lake basins. The WFD has already been discussed, but in many respects the meeting point for the various environmental Directives is the coastline. Coastal areas account for great species and habitat diversity, as well as ecosystem service diversity and economic activity and are therefore of central importance for EU environmental management and sustainable economic development.

2.6. Coastal Areas and Management Policy

This section on coastal areas focuses on policy alone but detailed discussion on EU and Irish coastal statistics, which is also very relevant to the policy agenda, features in Appendix A.

2.6.1. Coastal areas and environmental management policy

Coastal areas are characterised by habitats, species and population densities that are highly heterogeneous across space and specific to the coastal zone. The diverse range of habitats and species can be highly sensitive to human activities such as urbanisation and development which can have a lasting environmental impact (Gurran et al., 2007). The coastline of the Republic of Ireland stretches for 5,631 km (Hynes and Farrelly, 2011). The interaction of wind and wave with the highly exposed Irish coastline has created a diverse range of habitats for plants and animals. Tentatively, Irish coastal ecosystems are formed across five major habitat types: estuaries, sand-dunes, salt-marsh, sea-cliffs and shingle beaches (Lucey and Doris, 2001) but within these categories diverse habitat heterogeneity exists. Information about ecosystems and their derivative services are described in detail, as is relevant, in each of the empirical papers.

In addition, the mix of ecosystem services provided by coastal habitats can be dissimilarly valued by different groups of the population, which can lead to conflicts of interest and disagreements about how coastal areas should be managed (Billé, 2008). Like EU water and environmental policy, a historically sectoral approach to coastal management led to disjointed and contradictory management measures that risked undermining each other, inefficient use of resources and missed opportunities for more sustainable coastal development (EC, 2014c). However as was the trend for other Directives discussed in this chapter, this period was followed by one with a more integrated approach to coastal management.

By 2002 the European Commission outlined a general framework and provided council recommendations for a European version of ICZM (EC, 2002). The recommendations were made up of guiding principles along three main tenets (McKenna et al., 2008):

- Two 'procedural' principles: support and involvement of relevant administrative bodies and use of a combination of instruments that are focused on the attributes of the methods and procedures that might be used to best advance ICZM
- Three 'strategic' principles: broad overall perspective, long-term perspective, and working with natural processes. These principles mainly focus attention on long-term goals, and fit easily into the sustainability ethos that dominates contemporary environmentalism.
- Three essentially 'local' principles: local specificity, adaptive management during a gradual process, and involving all the parties concerned. These can be regarded as a balancing set to the second group, because they focus interest on specific areas and problems, encourage tailoring of management to local conditions and encourage the participation of the public in formulating management policy.

The idea behind ICZM is that as well as prioritising sustainability or 'strategic' goals, local and economic goals should also be part of the policy design process. As of March 2013 this set of principles was followed up by a draft proposal for a Directive establishing a framework for Maritime Spatial Planning (MSP) and integrated coastal management (EC, 2013). MSP, like ICZM, is concerned with managing human activities but with a greater focus on sea orientated development such as renewable energy installations, maritime shipping, fishing activities, and ecosystem conservation and tourism. Naturally, such at-sea activities also have a terrestrial coastal presence; renewable energy installations will be linked to land-based power sites, maritime shipping requires ports and the same is true of fishing and marine ecosystem tourism. The reality then is that any such developments must be coherent with an ICZM framework and vice versa. It is therefore sensible that the two operate through the one Directive. Like ICZM, MSP will involve stakeholder input and will be geared towards sustainable development of human activities. Furthermore, the development of MSP and ICZM in any Member State will affect and be affected by the MSFD, WFD the CFP, the HBD and the Renewable Energy Directive (RED).

In April of this year (2014) the European Parliament endorsed a Directive for Maritime Spatial Planning aimed at assisting Member States to develop plans to better coordinate the various activities that take place at sea. It also aims at ensuring that these activities are as efficient and sustainable as possible. The new Directive will help avoid potential conflicts between such diverse uses as fishing, aquaculture farms, cables, pipelines, shipping lanes and oil, gas and wind installations etc.

Chapter 4 focuses on ICZM and the use of surveys, stakeholder participation and scientific information in the design of local coastal management policy. In particular, the empirical methodology in chapter 4 considers the various principles of ICZM and evaluates whether they are coherent and effective in achieving the objective of ICZM, by highlighting in particular the fact services tend to be dissimilarly valued by various parties. The results of the study which included the participation of marine scientists and local stakeholders indicated that the local and strategic principles of ICZM do indeed lack coherence and that future policy reforms would need to prioritize specific principles of ICZM to make it truly operational and effective.

2.7. Integrated Maritime Policy

Clearly, the many Directives discussed in this chapter will be simultaneously employed in future management of the EU's environmental resources. For such management to be successful, it is important that these various Directives are fully integrated with each other. To that end, the Integrated Maritime Policy (IMP) seeks to increase coordination between all marine and coastal related Directives. The Commission is very clear on the fact that it is intended for such coordination and not for replacement of policies on specific maritime sectors (EC,2014d). It is to be especially relevant when management issues do not fall under a single sector based policy or when different sectors and actors are required to resolve an issue (EC, 2014d). Specifically the IMP covers the following 'cross-cutting' policies:

- Blue growth
    - Intended to develop sectors such as aquaculture, tourism, marine biotechnology, ocean energy and seabed mining. It also focuses on the development of marine knowledge, Maritime Spatial Planning and integrated maritime surveillance (intended for border control, fisheries control etc.).
    - Included action plans to ensure cooperation between countries with a stake in different sea regions. Of relevance to Ireland is the Atlantic Ocean action plan. This plan 'aims to revitalise the marine and maritime economy in the Atlantic Ocean area' such that Member States 'share information, costs, results and best practices, as well as generate ideas for further areas of cooperation of maritime activities. This includes both traditional activities, such as fisheries, aquaculture, tourism and shipping, as well as emerging ones such as offshore renewables and marine biotech' (EC, 2014d).

- Marine data and knowledge
  - Marine Knowledge 2020 is intended to act as a central collator of data from many sources with a view to assisting industry, public authorities and researchers.
- Maritime Spatial Planning
- Integrated Maritime Surveillance
  - This policy initiative is intended to provide authorities with a mechanism through which to exchange maritime information and data, reducing the cost of surveillance and improving its effectiveness. For example border control, safety and security, fisheries control, customs, environment and defence currently collect data separately, sometimes double collecting.
- Sea Basin Strategies
  - Will promote growth and development strategies that exploit the strengths and address the weaknesses of each large sea region in the EU

Many of the policies of the IMP are focused on development and synergies outside the scope of normal economic activities and environmental policies, but that which if successful, have the potential to aid these two causes.

2.8. Summary

This chapter has discussed the various policies and Directives that have arisen in EU coastal, marine and aquatic management in the last forty years. The discussion of these various pieces of legislation has shown that due to the complexity of the resource under management, the policy framework has itself become multi-faceted and integrated across habitat, sectoral and legal spheres. Early environmental policies like the Surfaces Water Directive and Bathing Water Directive gave way to a more comprehensive Directive in the form of the WFD. Given the interrelated nature of freshwater aquatic systems, reaching eventually to coastal estuaries, saltmarshes and bays, even this more comprehensive Directive could not stand alone if aquatic habitats and ecosystems were to be managed affectively. This Directive is thus intended to operate alongside the MSFD which provides policy guideline on management of the entire marine environment through the attainment of GES. The MSFD itself must then operate alongside the CFP such that GES can be attained. And operating in tandem with these polices is the HBD and the Natura 2020 network.

The link between freshwater aquatic systems, coastal habitats and the sea at large is catered for in a policy sense via a new policy framework which builds upon previous ICZM legislation and incorporates MSP to account for at-sea projects and development as well as those pertaining to areas of coastal proximity. These two sets of policy, run concurrently, are intended to allow stakeholders, coastal managers and other relevant parties to cooperate in designing coastal and marine management initiatives that promote environmental sustainability but also allow for local economic development. In addition to the now extensive (and growing) legislation that exists for marine and coastal management, the IMP is intended to act as a link between the various pieces of legislation in this area and a stopgap for maritime issues that arise, which do not fall under the jurisdiction of any of the aforementioned legislation. EU environmental policy relating to marine and coastal areas is still very much in its development, but the rate of change is rapid and transforming the face of European environmental management. Management methodologies that attempt to incorporate spatial and integrated methodologies and which can help to balance between the environmental and economic trade-offs of economic development and natural conservation will be important for the success of this transformation. It is with this latter consideration in mind that the empirical studies demonstrated in this thesis have been selected, adapted and applied.

3. The impact of precautionary quota constraints on the composition of multispecies harvest portfolios

3.1. Introduction

In June 2013 the European Parliament and Council of Ministers agreed upon a new and reformed European Common Fisheries policy (CFP) to be implemented across all EU marine waters in January 2014. One outcome of the agreements is that quotas and the use of species' maximum sustainable yields (MSY) will remain the primary means by which Member States (MS) attempt to achieve sustainable fisheries. Political problems with this form of fisheries management and with maintaining the scientifically recommended MSY throughout the political process have been documented within the EU (Daw and Grey, 2004). Despite these highlighted problems, the reforms indicate that the degree to which scientific recommendations of MSY are adhered to in practice will be far more binding than has been the case historically, such that by 2020, all stocks are to be managed at MSY. It is now clear that major changes to fishing quotas in European waters will occur in the next 6 years.

Further changes to the CFP include a banning of all discards and the adoption of multi-annual and multi-species planning. This means that the quantity of any fish stock that can be sustainably harvested will be determined on the basis of interaction with, and impacts upon, other species and marine habitats. If sustainable fisheries are to be attained, the impact of fishing for a single commercial species on other commercial species will be of great importance. It is foreseeable that in waters where the by-catch of biologically sensitive species is high, quotas for any target species in question will be set lower than their potential MSY level (had they been considered in isolation).

According to the European Commission, EU legislators will only define the general framework, the basic principles and standards and the overall targets of the CFP while

Member States will themselves develop recommendations on the actual implementing measures (EC, 2013). National policy makers will thus be charged with the responsibility of deciding upon and implementing the medium term management initiatives that will achieve the overall targets of the CFP. In this new policy environment, when setting species' total allowable catches (TACs), fishery managers must pay particular attention to the multispecies impact of harvesting an individual species, not least, the impact on other commercial species within the fishery and in neighbouring fisheries.

Models assisting the management process that follows the reforms will need to assess the environmental and ecosystem impacts of commercial fishing activity. In addition, behavioural economic models have a role to play since they offer a framework for attempting to describe the response of fishermen to any policy changes. According to Fulton et al (2011), human behaviour, and in particular fisher behaviour, is almost never explicitly considered by fisheries scientists in the assessment and management process. They posit that the uncertainty generated by unexpected resource user behaviour is as critical as ecosystem and environmental uncertainty because it has unplanned consequences and leads to unintended management outcomes. Indeed, technical measures can lead to results which actually work directly against specific sustainability targets for which they are designed (Briand et al., 2004).

While behavioural models may be underutilised by fisheries scientists, empirical analyses on the socio-economic impacts of fisheries regulations are plentiful (e.g. Jentoft, 2000; Nielsen, 2003; Hatcher and Pascoe, 2006; Wislon et al., 2006). Given the recent EU policy developments prioritising the by-catch issue and multispecies management, empirical analyses that have the potential for multispecies level analyses are desirable. This article presents a behavioural modelling approach based on financial portfolio theory and the expected utility hypothesis in an attempt to model the change in the harvest behaviour of a fishing fleet affected by precautionary quota constraints. The intent of the research is to demonstrate how the portfolio methodology could be employed by fishery managers to predict the likely behavioural responses of a fishing fleet to changing quota restrictions. While this process is useful in its own right, it also demonstrates the need for improved fishery data collection processes to implement such models successfully. The portfolio approach is based on portfolio theory as developed by Markowitz (1952). Markowitz's portfolio analysis is a mathematical tool to determine how to select the optimum proportion of assets in a portfolio for investment. The approach lends itself well to multispecies fishery analysis because given certain assumptions about the objective function of a fishing fleet it is possible to estimate changes in multispecies targeting behaviour given changes in single species harvest constraints. Thus a "multi-species-wide" impact of precautionary measures can be assessed. While portfolio theory has been extensively used for research into financial, agricultural and energy markets, its application to fisheries management and policy is rare. Some of the few papers that have done so are reviewed in the following section.

In what follows I first discuss previous literature that applies portfolio theory to fishery economic issues. Section 3 then presents the theory underlying the portfolio approach and how I apply it to the concept of mixed fisheries management. Section 4 provides a description of the multispecies Irish fishery investigated in the analysis and a brief description of the data used. The estimation results of alternative management scenarios are then presented in section 5. The paper concludes with a discussion of its major findings and their implications for fisheries management.

3.2. Previous applications of portfolio theory within fisheries economics

Typically, empirical multi-species analysis follows one of two formats; a bio-economic model which determines the optimal harvest rate of more than one species using estimated predator-prey or competitor parameters, or structural ecosystem models that can be used to determine optimal TACs across multiple species. More recently however, portfolio theory has been applied to fisheries management topics due to its capacity to embody a multi-species perspective and directly incorporate risk. Hanna (1998) advocates portfolio theory as a means of balancing fisher objectives and societal objectives while others extend this idea to 'explicitly recognize fishery resources as risk-bearing capital assets that can provide society with benefits indefinitely' (Edwards et al., 2004; 2005). These studies focus on realigning the goals of individual fishers with societal goals by adopting property rights, incentive schemes and fishing restrictions such that ecosystem service payoffs (as opposed to commodity payoffs) can be delivered to society. Others see the portfolio approach as a means of protecting fishing communities from the risk of fluctuations in the abundance, availability, or price of individual species, where fishers choose among a diverse portfolio of harvestable resources rather than being forced by regulation to specialize in one or an extremely limited number of species (Hillborn et al., 2001).

Elsewhere, Yang et al. (2008) use portfolio theory to assess the behaviour of New Zealand fishers' who face multiple targeting options to predict the optimal targeting strategies under a Quota Management System (QMS). Species considered by Yang, et al.(2008) were selected based on two criteria; the commercial value of the species and the availability of data. These two criteria were also highly relevant in the analysis of the Irish mixed fisheries and will be discussed further in section 3.3.

Sanchirico, Smith and Lipton (2006) also adapted financial portfolio theory as a method for ecosystem based fishery management (EBFM) that accounts for species interdependencies, uncertainty, and sustainability constraints. Illustrating the method with routinely collected species catch data available from Chesapeake Bay in the United States, the authors demonstrate the gains from taking into account species variances and covariances in setting species total allowable catches. They find over the period from 1962–2003 that managers could have increased the revenues from fishing and reduced the variance by employing ecosystem frontiers in setting catch levels. Sanchirico, Smith and Lipton (2006) also point out that compared to structural models of the ecosystem, deriving ecosystem frontiers provides a complementary view that is simple to implement and flexible enough to accommodate different ecological, economic, and social objectives by including additional constraints or objective functions. However, they also point out that a limitation of ecosystem frontiers is that the policy prescriptions are only as good as the estimates of the means and covariances that characterize the multivariate stochastic process.

Elsewhere, Perusso et al. (2005) highlight the fact that fisheries regulations tend to be species specific but that species can be part of a multi-species fishery. Therefore since harvest rates are correlated, net revenues attributed to each species are also likely to be correlated. The authors contend that this correlation means that portfolio theory is well suited for multi-species fisheries that exhibit joint productive characteristics. The authors therefore used a portfolio approach to model the behaviour of fishermen faced with multiple targeting options in a random harvest fishery. The approach draws from the expected utility hypothesis and financial portfolio theory to predict optimal targeting strategies. The methodology was applied to the pelagic long line fleet operating in the U.S. Atlantic Ocean, Caribbean and Gulf of Mexico. Results from the model provide evidence that area closures aimed at reducing juvenile swordfish mortality will be more effective in certain regions. Efficient risk-return frontiers were also generated for use in predicting targeting behaviour in lieu of a closure.

The frontiers suggested that trips that target swordfish exhibit a smaller degree of variability than trips that do not.

More recently, Theophille (2012) uses a mean-variance portfolio optimization approach to determine whether there is potential for fishers in Dominica to reduce the variability of net trip revenues. Their results suggested that fishers could attain their ex ante targets and that given the potential for trip-level harvest portfolios with a more efficient mean-variance profile, the variability of net trip revenues could be reduced.

I employ portfolio theory to combine a multispecies and precautionary approach under a single empirical framework and follow Perusso et al. (2005) by incorporating the expected utility hypothesis into the analysis. Through this approach, I attempt to predict the impact of hypothetical quota-based precautionary measures on the utility of fishermen in the Hake-Monkfish-Megrim and the Cod-Haddock-Whiting fisheries in Irish waters. To infer realistic hypothetical precautionary measures I refer to Cawley et al. (2006) where the authors review the status of various Irish fish species and the potential measures that need to be adopted to protect specific stocks from decline. I assess the behavioural response of the fleet to precautionary measures by observing the subsequent changes in the contribution of target species to the overall fisheries harvest portfolio. This contribution is often referred to as the, portfolio “weight”, of a particular species.

3.3. Methodology

Portfolio theory assumes that economic agents are profit maximizing and risk averse, balancing a range of expected payoffs (and the risk/variability associated with attainment of each payoff) to maximize their expected utility. The portfolio problem is thus formulated using the expected utility function of Von Neuman-Morgenstern (1944) and can be written:

$$\max_{x_1, \ldots, x_N} E[U(W_1)] = E\{ U \left[\left(1 + \sum_{i=1}^{N} x_i R_i \right) W_0 \right] \}$$

where E(.) is the expected value of (.), U is utility level, W_0 and W_1 are initial and updated wealth respectively, R_i is the return on the ith asset and x_i is the percentage contribution of the ith asset to the total harvest portfolio. In this study, I assume that the fishery manager looks at the fleet as a single entity, forming expectations about the revenue it can generate from harvesting each of a set of species and the risk (variability) associated with each revenue stream. It then uses these expectations to select the portfolio of target species that maximises its expected utility. Like others (Mistaien and Strand, 2000; Perusso et al. 2005) I assume the fleet's initial wealth is zero so that the possibility of existing wealth influencing ex ante targeting decisions does not arise. Furthermore, due to the absence of cost data, I focus on the impact of fishery revenues on fleet utility as opposed to the more ideal case of the impact of fishery returns on fleet utility. This means that the main determinants of the fleets targeting decisions arise out of annual revenues. An improvement to this approach may have been to use cost data from equivalent fleets in other countries, or to collect some cost data cost points to pin point costs somewhat, and this may have improved the analysis. However as is explained in more detail later in the chapter (p. 65) the negative impact on the analysis of not including costs has been negated by analysing fisheries where costs are generally uniform across the fleet. Issues around considering the fleet as a single entity and the omission of environmental stochasticity in calculating variances is also eluded to later in the chapter.

Like Perusso et al. (2005), I use a Taylor Series expansion to approximate the utility function, but because the study only has annual and aggregated level data available, I do so for annual revenues $E[W^*]$ for the entire fleet as a unit:

$$U(W^*) = U(E[W^*]) + U'(E[W^*])(W^* - E[W^*]) + \frac{1}{2}U''(E[W^*])(W^* - E[W^*])^2 + R_3,$$

where:

$$R_3 = \sum_{n=3}^{\infty} \frac{1}{n!} U^{(n)}(E[W^*])(W^* - E[W^*])^n,$$

U^n is the n-th derivative of U and E[.] is the expected value of [.]. A convergent Taylor series leads to total fleet expected utility,

$$E[U(W^*)] = U(E[W^*]) + \frac{1}{2}U''(E[W^*])\sigma^2(W^*) + E[R_3].$$

where $\sigma^2(W^*)$ is the variance of annual fleet revenue . And:

$$E[R_3] = \sum_{n=3}^{\infty} \frac{1}{n!} U^{(n)}(E[W^*]) m^n(W^*).$$

where $m^n(W^*)$ is the n-th central moment of W^*:

As per Perusso et al. (2005), this latter equation reflects the fact that expected utility is explained by mean, variance and other high moments of the probability distribution of fleet revenue. Ideally, the availability of cost data would allow the researcher to model expectations about returns/profit. This would allow the model to more accurately reflect the feasibility of alternative combinations of aggregate catch within the fishery. In the absence of such cost data, one can still apply the methodology to fisheries in cases such as ours where the fisheries being investigated are assumed to have equivalent fixed costs (such that there are no barriers to exit one fishery and enter another) and similar variable costs structures. This is discussed in more detail in section 3.5. It would also be superior to have more frequently observed data on catch quantities and prices since variance in annual total revenue over time is unlikely to be randomly or independently distributed. A further issue with the data is that it contains observations of only the Irish fleet's revenues and thus the variability of revenue experienced by other fleets is omitted, so that the actual variance of annual revenues may differ to the figure produced by the model[5]. These shortcomings mean that for a set of harvest targets I cannot be sure that the distribution of expected catch around those targets will be accurately represented by the variance of revenues from earlier years. With that said, the purpose of applying a portfolio model in this case is to demonstrate how the framework could be employed by managers of a multispecies fishery and the types of data that would be needed to do this successfully

As earlier stated, the expected utility of the fleet is a function of the mean and variance of revenue:

$$\mathbf{E}(\mathbf{U_i}) = \mathbf{E}(\mathbf{R_i}) - \boldsymbol{\varphi}(\boldsymbol{\sigma}_\mathbf{i}^\mathbf{2}) \qquad (1)$$

where $\mathrm{E}(.)$ is the expected value of $(.)$, U is the fleet's utility, $\mathrm{R_i}$ is revenue per tonne of species i harvested, φ is the fleets risk aversion parameter and is the same toward all species,

[5] Irish share of EU quota in the areas under study will vary according to species, but generally the Irish share of EU quota in these areas, and thus the degree for which Irish catches can explain overall variance within these fishing grounds, is relatively small. However the catches from a fair sample of the overall population. See EC (2014b).

and σ_i^2 is the variance the fisher expects from the revenue generated from harvesting species i, defined as the variance in average revenue per tonne of species i harvested over the historical period.

Correspondingly, the expected utility of the fleet, written now as a function of the first two moments of the *harvest portfolio* $\emptyset$, is

$$\mathbf{E}(\mathbf{U}_{\emptyset}) = \mathbf{E}(\mathbf{R}_{\emptyset}) - \boldsymbol{\varphi}\left(\boldsymbol{\sigma}_{\emptyset}^{2}\right) \qquad (2)$$

where:

$$\mathbf{R}_{\emptyset} = \sum_{\mathbf{i=1}}^{\mathbf{n}} \mathbf{w_i R_i} \qquad (3)$$

$$\mathbf{R_i} = \mathbf{P_i Y_i} \qquad (4)$$

and $\sum w_i \leq \sum i$ i.e. $0 \leq w_i \leq 1$; $w_i R_i \geq 0$; $Y_i \geq 0$; $P_i > 0$. P_i is the unit price of species i, Y_i is the tonnage of species i harvested and w_i is the weighting on species i in the harvest portfolio. Note that each weight on each species can equal 1 simultaneously since I am using expected revenues rather than expected returns. To elaborate, the focus here is not on returns, which are fractional (meaning all weights should sum to 1) but instead it is on revenues (which are not fractional); the sum of the weights can sum to whatever amount of species exist in the harvest portfolio. The expected revenue for any fleet harvest portfolio $\emptyset$ then, is simply the sum of the expected revenues by the weight allocated to each species in the fleet harvest portfolio. Note that the weight allocated to each species is the percentage share of total catch. For risk however, we must also consider the covariance between the revenues generated from harvesting each species. Such covariance arises out of ecosystem linkages such as a predator-prey or competitor relationships (Garrod and Harding 1981; Daan, Rijnsdorp and Overbeeke 1985; Daan 1989; Köster and Schnack 1994, Trenkel et al. 2004), common sensitivity (be it positive or negative) to environmental fluctuations and fishing types, and indeed, any macro type variable that affects multiple species within the ecosystem, or in this case, the harvest portfolio, $\emptyset$. As earlier alluded to, my annual measures of price and quantity are quite crude compared to the level of detailed data required to get at the true distribution of expected revenues fishermen perceive in achieving their target harvests. However they allow the methodology to be carried out and demonstrate its potential should more quality data be available.

Price sensitivities to market conditions also cause covariance between revenues. Calculations of revenue covariances capture this, but I follow Sanchirico et al. (2006) in assuming that fish prices are unresponsive to ecosystem-wide catch levels due to substitute protein sources and world seafood markets. This means that the degree of price substitution or complementariness between species in the portfolio is irrelevant compared to other market factors, and therefore prices can be classified as exogenous. I also employ the technique of exponential smoothing, as demonstrated by Sanchirico et al. (2006), meaning the less recent an observation of P_i and Y_i, the less influence it has in the calculation of $E(R_\emptyset)$ and $\sigma^2_\emptyset$. The degree to which the influence of past observations on expected values diminishes is determined by a factor referred to as the rate of decay (λ). This technique allows one to mimic the possibility that fishers place more emphasis on recently occurring events when forming expectations about future outcomes (Guttormsen 1999; Bowerman and O'Connell 1993), but we can also relax this assumption by increasing the value of λ until such a point as it reaches 1, whereby all observations, regardless of the time period in which they occur, are equally weighted. Each term in the variance covariance matrix of the fleet's harvest portfolio is then calculated as:

$$\boldsymbol{\sigma_{ij}} = \frac{\sum_{k=0}^{t-1} \boldsymbol{\lambda}^{\mathbf{t-k}}\left((\mathbf{R_{j,t-k}}) - \mathbf{E}(\mathbf{R_i})\right)^{\mathbf{2}}}{\sum_{k=0}^{t-1} \boldsymbol{\lambda}^{\mathbf{t-k}}} \tag{5}$$

where:

$$\boldsymbol{R_i} = \frac{\sum_{k=0}^{t-1} \boldsymbol{\lambda}^{\mathbf{t-k}}(\mathbf{R_{i,t-k}})}{\sum_{k=0}^{t-1} \boldsymbol{\lambda}^{\mathbf{t-k}}} \tag{6}$$

Note that the definition of R_i is the same as before ($P_i Y_i$), only now the observation from each time period is weighted in relevance according to the decay factor λ. The total variance $\sigma_\emptyset^2$ of the fleet harvest portfolio is then defined:

$$\boldsymbol{\sigma_\emptyset^2} = \sum_{\mathbf{i=1}}^{\mathbf{n}} \sum_{\mathbf{j=1}}^{\mathbf{n}} \boldsymbol{\sigma_{ij}} \mathbf{w_i} \mathbf{w_j} = \boldsymbol{\rho_{ij}} \boldsymbol{\sigma_i} \boldsymbol{\sigma_j} \mathbf{w_i} \mathbf{w_j} \tag{7}$$

where σ_i is the standard deviation of R_i, σ_{ij} is the covariance in revenue between species i and j, except when $i = j$ (meaning we have σ_{ii}), at which point it refers to the *variance* of R_i, that is, σ_i^2. ρ_{ij} is the correlation coefficient between the revenues for i and j and when $i = j$, it must be equal to 1.

With the definitions of the different variables in place, the quadratic programming problem is then:

$$\textbf{Minimize } \boldsymbol{\sigma_\emptyset^2} = \sum_{\mathbf{i=1}}^{\mathbf{n}} \sum_{\mathbf{j=1}}^{\mathbf{n}} \boldsymbol{\sigma_{ij}} \mathbf{w_i} \mathbf{w_j} \tag{8}$$

subject to:

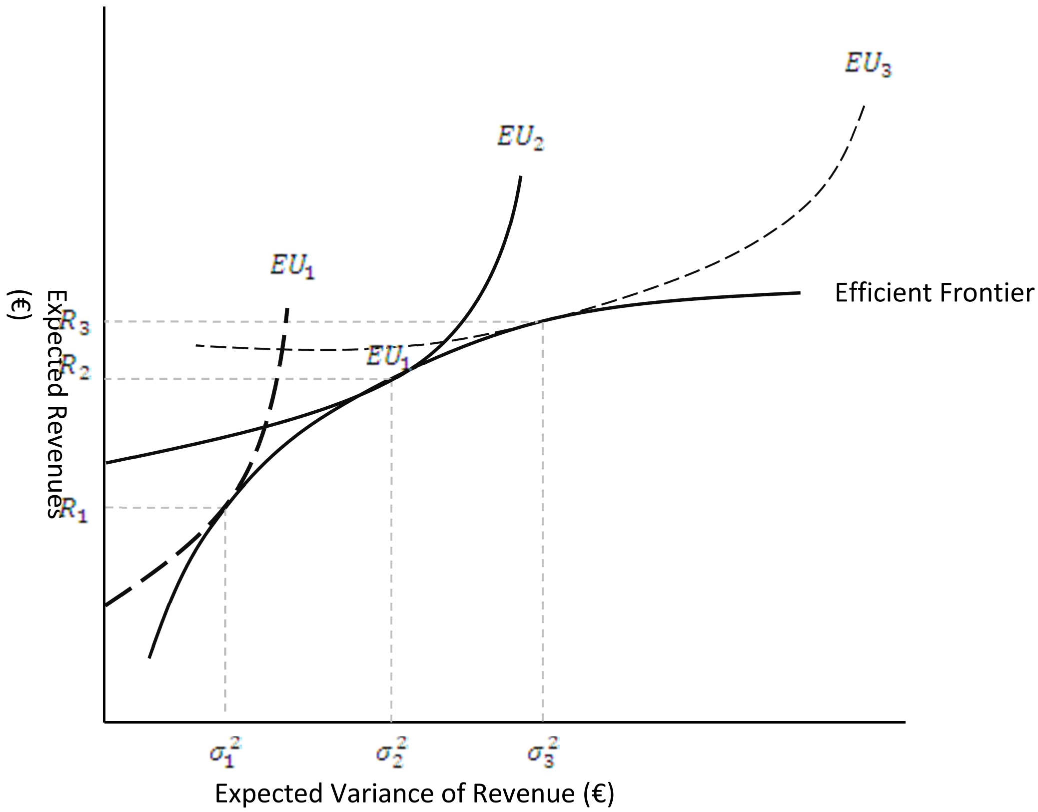
$$\boldsymbol{R_\emptyset} = \sum_{i=1}^{n} \boldsymbol{w_i R_i} \geq \boldsymbol{T_\emptyset},\ \boldsymbol{w_i Y_i} \leq \boldsymbol{PL_i} \tag{9}$$

where $T_\emptyset$ is some target level of revenue for fleet harvest portfolio $\emptyset$, and PL_i is a precautionary catch limit set by management for species i (specifically it is a species weight constraint within the harvest portfolio). By carrying out the optimization procedure for increasing values of $T_\emptyset$ (starting from zero) the minimum level of $\sigma_\emptyset^2$ for each value of $R_\emptyset$ is calculated. Plotting the different values of $\sigma_{\emptyset}^2$ for every value of $R_{\emptyset}$ produces what is termed, the efficient frontier of the entire fleet[6].

In this case, the efficient frontier represents the minimum expected variance the fleet can achieve on the basis of historical covariances in order to attain its target expected revenue. Or perhaps more accurately how fishery managers will expect the fleet to behave given historical outcomes. The final determinant of the fleet's target level of revenue, given the expected variance associated with it, will be the aggregated attitude of all fishers within the fishery towards risk, represented through the risk aversion parameter φ, and this in turn will determine the weight allocated to each species within the fleet's portfolio; it is by adjusting the species weights (either through the fleets own decision making or through management determined quotas) that the fleet target portfolio travels along the efficient frontier. Once a target level of revenue is set, the harvest portfolio associated with it is delineated by directing fishing effort into achieving the weights that will determine such a portfolio's expected revenue and variance. There is a depiction of the relationship between the efficient frontier, aversion to risk, and the expected utility curve in Fig. 3.1. Fleet 1 has a high aversion to risk, and therefore selects a mix of species which result in low revenue, but a correspondingly low expected variance (expected utility curve 1). Fleet 3 is at the opposite end of the scale and is less risk averse and selects the mix of species which achieve higher expected revenue but expose the fleet to a higher level of variance (expected utility curve 3).

[6] The portfolio analysis described above was carried out in the software package GAMS. The algorithm used in GAMS/CONOPT is based on the GRG algorithm first suggested by Abadie and Carpentier (1969). Details on the algorithm can be found in Drud (1985 and 1992). The procedure first uses initial values to compute a feasible solution. Then the constraints and initial parameter values for the predictor variables are combined in order to calculate a gradient for the goodness of fit measure that can allow updating of predictor variable parameters from their initial values. If the change in the parameters along the calculated gradient equals or is below the minimum threshold for goodness of fit change, the algorithm is said to have converged. Otherwise parameters that can be profitably updated are changed in the calculated search direction using a pseudo-Newton updating process. The procedure continues until either: the minimum threshold for goodness of fit change is achieved, conditional upon the specific parametric constraints; or, the maximum allowable number of algorithm iterations is achieved. If this threshold minimum is not achieved in the maximum allowable algorithm iterations, the algorithm is said to not have converged.

Figure 3.1: Hypothetical efficient frontier and expected utility curve of three different individuals, each with a differing aversion toward risk



3.4. Data and the Irish Mixed Species Fisheries

The seas around Ireland contain some of Europe's most important fishing grounds. Irish-Atlantic coastal waters, the West of Scotland coast and Rockall, the Celtic Sea and the Irish Sea possess a rich abundance of commercially fished species and diverse marine habitats which support them. According to statistics from the Irish Sea Fisheries Protection Authority, the total value of fish landings in the Irish fisheries sector in 2008 amounted to €214 million (SFPA 2010). Comprising 16% of total EU waters (Irish Naval Service 2007), Irish territorial waters are currently governed as part of the European Union's Common Fisheries Policy (CFP). The reform of the CFP in 1983 established the concepts of Exclusive

Economic Zones (EEZs)[7] within EU waters, relative stability[8] and conservatory management measures based on TACs[9]. The quantities of fish caught in EU waters today are therefore regulated by determining the annual TAC of each commercially fished species through scientific advice and a political process established under the CFP. Member states are then allocated a share/quota of this TAC on a fixed percentage basis, determined largely by their historical fishing patterns and relative dependency on the fishing industry.

The Irish fish catching sector is largely comprised of deep water, demersal, pelagic and shellfish fisheries (see Table 3.1 for a breakdown of Irish fishing segments and relevant target species).

Table 3.1: Irish species pertaining to each segment of the Irish Fishery

Segment	Targeted Species
Pelagic	Pelagic species: Mackerel, Herring, Horse Mackerel, Blue Whiting, Sprat, Sardines
Polyvalent	Whitefish Species: Monkfish, Megrim, Haddock, Whiting, Cod, etc. Dublin Bay Prawns/Nephrops, Pelagic Species (limited quantity). Inshore Non-Quota Shellfish Stocks
Beam-trawl	Flatfish species: Sole, Plaice, Megrim, Monkfish
Specific	Bivalve Molluscs: e.g. Mussels, Scallop, Razor clams

7 Exclusive Economics Zones can be defined as the territorial waters of a nation, extending to 12 nautical miles from the baseline. First established under the 1983 review of the CFP, this method of allocation was initially adopted to promote political stability, allowing each member state's fishing effort to remain constant, relative to that of others. It also gives preference to the fishing dependant countries of Northern Europe under the Hague Resolution (Boude, et. al., 2001).

8 TACs are shared between EU countries in the form of national quotas. For each stock a different allocation percentage per EU country is applied for the sharing out of the quotas. This fixed percentage is known as the relative stability and ensures current and future catch shares are based on historical levels

9 TACs are placed on each fishing zone, within EU waters. These limits are determined by ecological surveys and analyses, with final catch levels set annually by a meeting of the European Commission of Fisheries Ministers. First established under the 1983 review of the CFP, this method of allocation was initially adopted to promote political stability, allowing each member state's fishing effort to remain constant, relative to that of others. It also gives preference to the fishing dependant countries of Northern Europe under the Hague Resolution (Boude, et. al., 2001).

My analysis focuses, firstly, on the Hake, Monkfish and Megrim fishery and secondly on the Cod, Haddock and Whiting fishery. This is because these fisheries are multi-species in nature and therefore the type of fisheries where cross-species effects of single species quota constraints occur, making them suitable for an application of the portfolio approach. They are also fisheries in which substantial species interrelatedness is documented in the scientific literature (Hislop 1996; Garrod and Harding 1981; Daan et al.1985; Daan 1989; Köster and Schnack 1994; Bromley et al. 1995; Trenkel et al. 2004). Given the extent of species interrelatedness that exists for the fisheries under study, a portfolio theory approach, which estimates a variance covariance matrix across species catch quantities seems a viable approach to incorporating species interdependency.

Ireland's quota for the Hake Monkfish and Megrim fishery comprises 9% of the EU TAC. The fishery generated €18.9m in dockside revenue in 2004 and accounted for 29% of demersal landings. Both Hake and Monkfish can be targeted using either longline, trawl or gillnet methods and are therefore core target species of the polyvalent and beam trawl segments of the Irish fishing fleet. Megrim is largely caught using trawling methods. While the beam trawl segment comprises only 1% of the vessels in the Irish fleet and 2% of the capacity, the polyvalent segment represents 85% of the fleet and 48% of capacity (Cawley et al.et al. 2006). In recent years, the Irish quota for Hake and Monkfish has increased by 12% and 30% respectively yet Cawley et al. (2006) point out that recent ICES advice suggested Monkfish was 'over-exploited in relation to its highest yield'.

The Cod, Haddock and Whiting fishery have also experienced declining stocks in recent years. Indeed there has been a dramatic decline of Cod in all the main fisheries around Ireland and in the North Sea (Cawley et al. 2006). Ireland's quota of Cod, Haddock and Whiting amounts to 17% of the TAC and the first point of sale value was €12.1 million in 2004. Landings of Cod, Haddock and Whiting accounted for 18% of the total value of demersal species landed in 2004, contrasting starkly with a 26% contribution in 1995. According to Cawley et al. (2006), this had led to 'significant displacement of traditional fleets from these areas and today many of the larger vessels from the Greencastle[10] fleet travel regularly to the Celtic Sea to fish. Likewise the traditional Irish Sea whitefish fleet[11] has all but disappeared. It is clear too that as more vessels turn their attention to the Hake, Monkfish and Megrim fishery in the Celtic Sea and to the Dublin Bay prawn fisheries both in the Irish Sea and off the south-west coast, these already heavily fished stocks are very vulnerable to further over-exploitation' (Cawley et al. 2006). More recently, the Irish stock book (2011) finds that Cod and Whiting are overly exploited and severely depleted in the Irish Sea. In the Celtic sea surveys revealed a downward trend in the biomass and abundance of Cod, Whiting and Hake. Recent dedicated anglerfish/Monkfish surveys indicate a decline in abundance since 2007.

The historical price and quantity data used in the analysis is collected by the Sea Fisheries Protection Authority (SFPA) and reported annually by the Irish Central Statistics Office (CSO). The SFPA collects and analyses data on fish landings and fishing activity by all Irish vessels and foreign vessels landing into Ireland. This data includes information on the quantity, value, and location of fish caught, together with effort data and details of fishing methods used. Fish and shellfish are landed at the five major fishery harbour centres (Killybegs, Castletownbere, Howth, Rossaveal, and Dunmore East), at 40 secondary ports (each with landings exceeding €1m) and a further 80 piers and landing places across Ireland (Cawley et al.2006). The revenue $\mathrm{R_{it}}$ generated by each species in each year is calculated using the total quantity $\mathrm{Y_{it}}$ of each species i recorded/landed at all of the main ports around

[10] Greencastle is a significant whitefish fleet in the North of Ireland (Donegal). As such, the large scale vessels within this fishery have the capacity to relocate effort to the South of Ireland in the Celtic sea, adding extra fishing pressure to stocks in these areas.
[11] This fleet constituted family owned artisanal fishing vessels usually less than 15 metres in length

Ireland in that year, and the average dockside price P_{it} of each species for all of the ports during the year.

The sample period used in the analysis is 1977 until 2004. While data for years earlier than this is available from the CSO for the Cod, Haddock and Whiting fishery, it is not available for the Hake, Monkfish and Megrim fishery. The principle variables reported by the CSO for Irish fisheries are species class, aggregate landings by port/consumption category/month/average live weight per tonne and value by main species. While individual vessel level data would be more useful for an in-depth economic analysis of each fishery, the portfolio approach lends itself well to the analysis of aggregate price and quantity data, such as that collected by the CSO in this case.

3.5. Results

Using the portfolio theory approach I consider three different fishery management scenarios. In the first scenario I look at the status quo situation in each fishery. This is specified as the catch composition of the most recently observed harvest portfolio (2004). I then compare this to the optimal portfolio the fleet could have attained based on historical revenues and covariances. This indicates the accuracy of the model's predictions about the fleets' targeting choices as a whole and the extent of any risk-revenue balancing behaviour the fleet potentially engages in.

In the second scenario, I replicate a hypothetical precautionary quota constraint for a single species and observe how the fleet's targeting behaviour toward alternative species in the *same* fishery changes. This hypothetical scenario is simply a running of the model with a constraint placed on the potential contribution of a specific species to the overall harvest portfolio. Sticking with this hypothetical case (the second scenario), I replicate the precautionary measure a second time but allow the fleet to switch its targeting effort to species in the neighbouring fishery also. I have specifically selected fisheries which are relevant to each other both through ecosystem linkages such as a predator-prey or competitor relationships (Garrod and Harding 1981; Daan, Rijnsdorp and Overbeeke 1985; Daan 1989; Köster and Schnack 1994, Trenkel et al. 2004) and in the sense that the multiple species which make up the two fisheries are genuine harvest alternatives to each other. This is because fishers within each fleet can alternate targeting behaviour to the other fishery without having to incur any substantial fixed costs since both fisheries fall into the demersal and seine trawlers category.

If alternating between the fisheries in question required vessels to undergo costly gear and equipment changes, fixed costs would be far more important in the analysis since fixed costs act as a barrier to entering a new fishery. Allowing for species harvesting alternatives in the modelling process that are not realistic in practice (due to fixed costs barriers to entry) could lead to erroneous results if fixed data was not included in the model. This study would benefit from having variable cost data, however this data is not available at this time and by selecting fisheries that had only marginally different variable costs, the implication of omitting costs from the analysis was minimised. The similarity in variable costs between the two fisheries is highlighted by the fact that data on the cost structures of these two fisheries are aggregated in Bord Iascaigh Mharra (Irish Sea Fisheries Board) annual economic fishery surveys.

In the third scenario, I hypothesize a second precautionary quota constraint being placed on a different species in the neighbouring fishery. The intention is to mimic a situation where the initial precautionary initiative forces displaced fishing effort into the alternative fishery, increasing the fishing pressure on its stocks, causing management to respond by implementing a second quota constraint in the affected fishery. I then discuss the results and the implications of the various outcomes, both for the fisheries in question, and the fisheries portfolio methodology itself.

Table 3.2 below presents the descriptive statistics for the species in each of the two fisheries for different values of the decay factor λ. The expected revenue values have the property of non-monotonicity as the value of λ changes. This arises because the historical price and quantity of each species varies across time, and different values of λ weight different time periods differently. Where the expected revenue value is highest when λ=0.741 it is likely that the species was under-exploited in the earlier portion of the sample period, became increasingly exploited in the middle period, and then due to overfishing suffered decline. The result shows the benefit of using a decay factor to describe fishers' expectations since it reflects a more accurate depiction of "current" opportunities in the fishery. Table 3.3 presents the correlation matrix of all potential species in the fisheries' harvest portfolios. The correlation coefficients range from less than 1 to negative values suggesting that there *is* scope for risk diversification in the fishery.

Table 3.2: Descriptive Statistics for Species Revenues (Euros) for different λ

	Average Revenue			St. Dev		
λ	1	0.741	0.549	1	0.741	0.549
Cod	10,971,720	7,131,533	5,649,362	3,041,685	2,870,629	1,972,226
Haddock	4,945,147	6,390,685	5,800,260	2,262,687	1,884,751	1,575,015
Whiting	8,088,805	6,685,095	5,583,926	1,947,514	2,268,947	1,812,277
Hake	5,988,474	5,376,045	4,164,673	3,782,108	2,618,039	1,630,326
Monkfish	6,602,780	10,021,560	9,739,959	4,280,490	1,358,567	1,069,687
Megrim	8,079,675	10,031,690	8,794,949	4,893,231	2,992,306	2,434,117

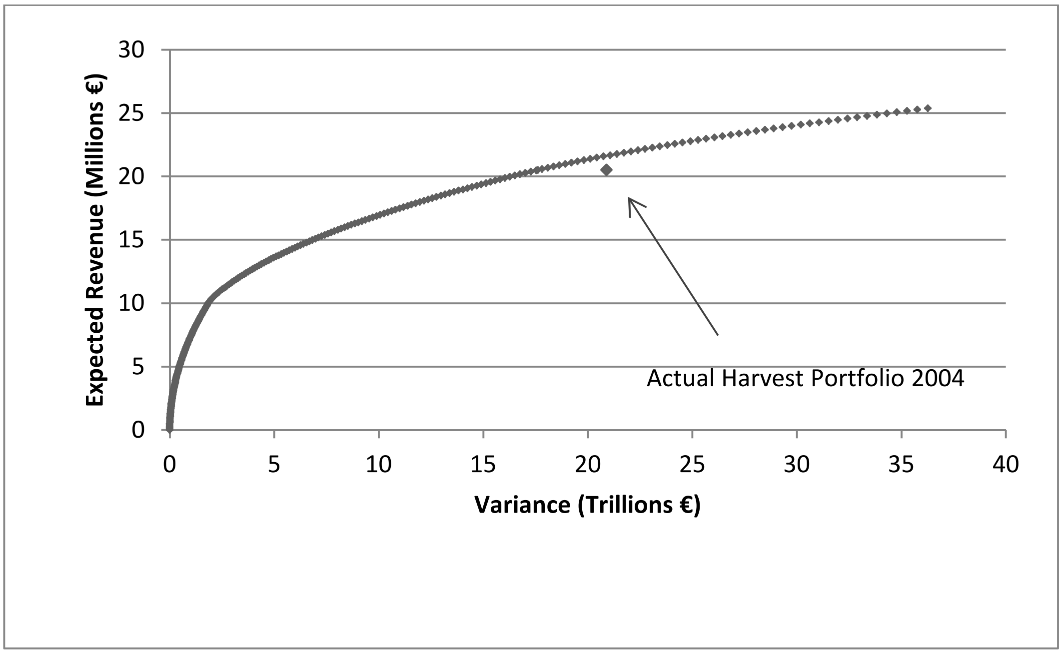
Table 3.3: Variance-covariance matrix of specie revenues (Euros) for λ=1

	Cod	Haddock	Whiting	Hake	Monkfish	Megrim
Cod	1					
Haddock	-0.091	1				
Whiting	0.509	0.023	1			
Hake	0.197	0.495	-0.01	1		
Monkfish	-0.207	0.721	-0.081	0.764	1	
Megrim	0.051	0.747	-0.023	0.829	0.876	1

Scenario 1: The Status Quo Situation

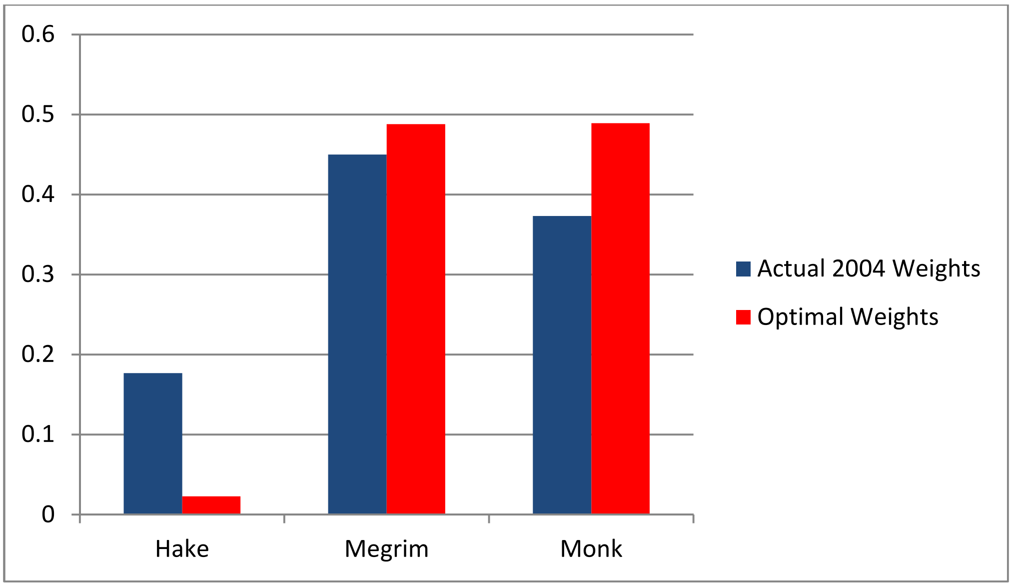
Discussion of the results focuses on a precautionary measure of 50%, but Table 3.4 (see p. 77) also shows the resulting species weights when the precautionary measures graduate from 20%, to 50%, to 80%. A precautionary measure is simply a constraint on the contribution a specific species can make to the harvest portfolio. The Hake, Monkfish and Megrim fishery generated €20.53m in real dockside revenues in 2004. To attain this €20.53m in revenue, Irish fishers within the fishery selected a harvest portfolio with a standard deviation of €4.57m (based on the estimated variance/covariance matrix). By determining species weights optimally, I estimate that the fleet could have achieved that same level of revenue by selecting a harvest portfolio with a standard deviation of €4.196m.

Figure 3.2: Efficient Frontier of Harvest Options for the Hake, Monkfish and Megrim Fishery



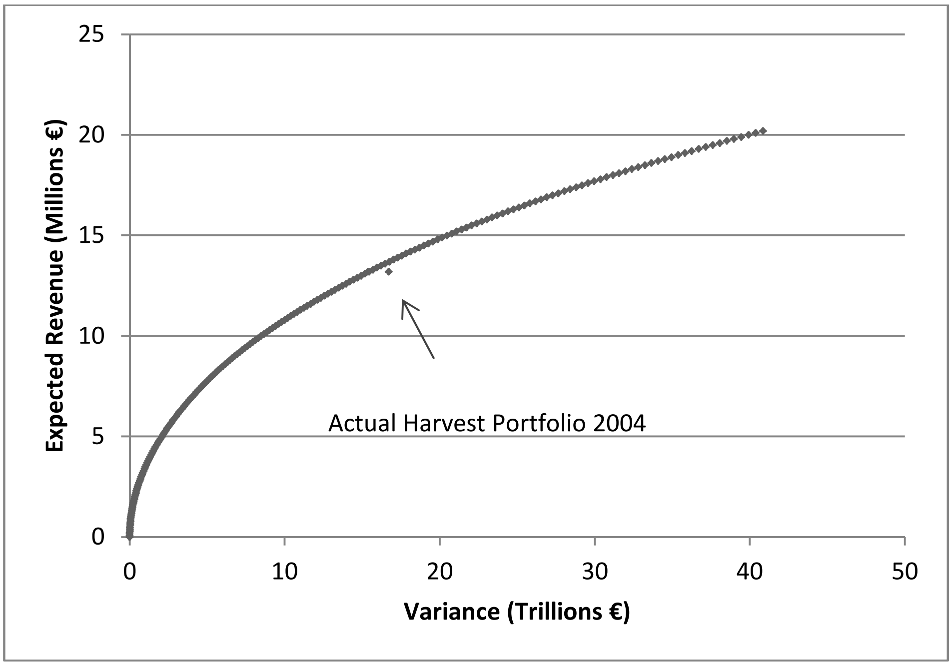
The efficient frontier for the Hake, Monkfish and Megrim fishery is shown in Fig. 3.2. The frontier displays the set of possible minimum variance portfolios for fleet target revenue of between zero and €25.43m (the maximum possible on the basis of historical averages). The point of interest in this scenario is the optimal portfolio for the fleet target revenue of €20.53m. To reiterate, €20.53m is the revenue that the fleet actually generated, and I am concerned with determining whether it was possible to do so with less exposure to variance. While the optimal portfolio at this level of target revenue lies on a point along the frontier, the actual harvest portfolio of 2004 is located below this line. This may highlight the potential for increased efficiency with respect to species selection, specifically, an 8.2% decrease in portfolio variance for the same expected revenue, but since I do not have information on cost or technical interactions that determine the profitability/feasibility of achieving particular combinations of aggregate catch, it is not possible to determine this. It may be that procuring such a portfolio would be less risk-return efficient if profitability was the variable under consideration instead of revenue. The ex post mix of species in the actual 2004 harvest portfolio vs. the optimal weights are shown in Fig. 3.3.

Figure 3.3: Species Weights for Hake, Monkfish, Megrim Harvest Portfolio with Target Revenue of €20.53m



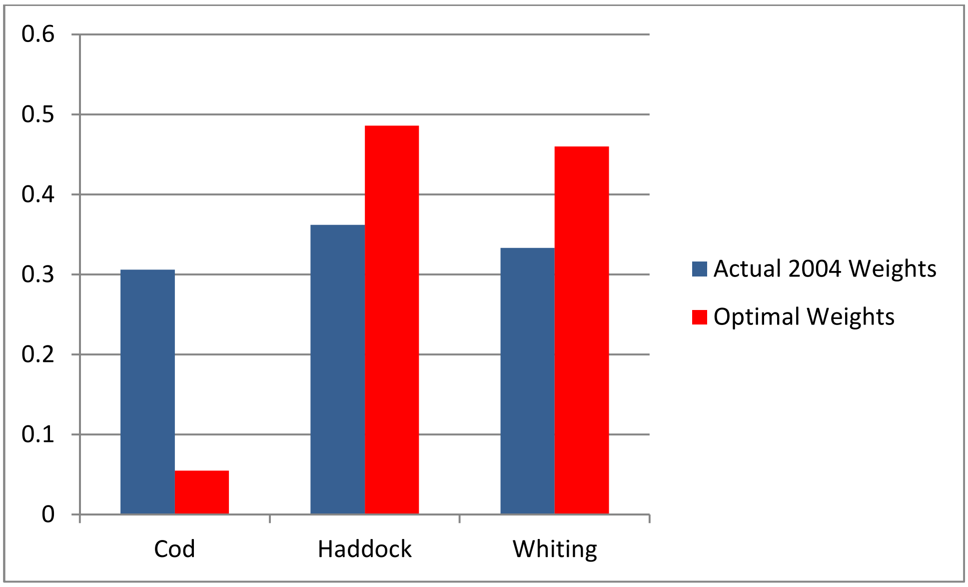
The Cod, Haddock and Whiting fishery generated € 13.16m in real dockside revenues in 2004. To attain this € 13.16m in revenue, I estimate that the fleet selected a harvest portfolio with a standard deviation of €4.088m. The species weights selected through the portfolio optimization for a harvest portfolio of equal total revenue (€4.088m) resulted in a standard deviation of €3.92m, which suggests that at this level of target revenue there is scope for a 4.11% decrease in fleet portfolio variance. Actual harvest portfolio relative to the efficient frontier is shown in Fig. 3.4. Again, including variable costs in the analysis could very well undo the appearance of any possible efficiency gains. The ex post mix of species in the harvest portfolio vs. the optimal portfolio are shown in Fig. 3.5.

Figure 3.4: Efficient Frontier of Harvest Options for the Cod Haddock Whiting Fishery. The actual harvest portfolio of 2004 is the point lying below the efficient frontier



These results suggest that if scope for risk-revenue trade-off efficiency gains exist, they are not large, and may even be less if variable costs are considered. As such, it suggests that fishers already balance targeting strategies between revenue and risk well. From Fig.3.5 we can see some species, such as Cod, are more important in the real world than in the optimization. The historical significance of Cod in Ireland, and the development of an entire fishing culture around it, can easily explain why it features so prominently in the actual fleet harvest portfolio, despite the fact that it has a lower efficient revenue to risk profile (see Table 3.2). It is outside the scope of this paper to factor qualitative observations such as this into the framework, but it is feasible that any characteristics of a particular species that affects the fishing-community utility function in a non-monetary way could be incorporated into such an analysis as this.

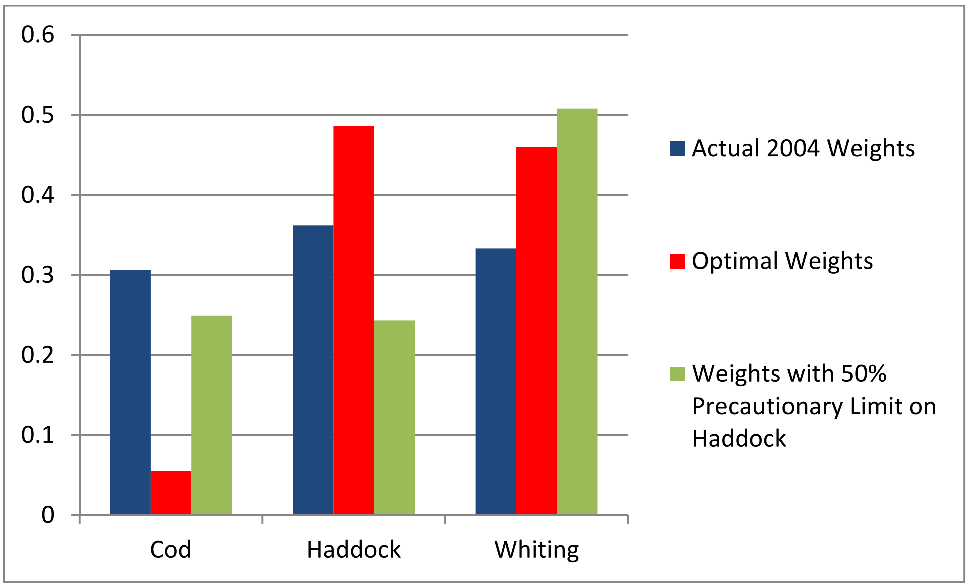
Figure 3.5: Species Weights for Cod, Haddock, Whiting Harvest Portfolio with Target Revenue of €13.16m



Scenario 2: Precautionary Measure in the Cod, Haddock, Whiting Fishery

Cod, Haddock and Whiting stocks around Ireland have all declined in recent years; 78%, 39% and 57% respectively between 1995 and 2004. According to Cawley et al.(2006), Cod is severely depleted in all Irish waters, Whiting in the Irish Sea and off the north-west coast, and Haddock, while not overly depleted, is considered over-exploited. In this second scenario, the hypothetical precautionary measure for a single species that is envisaged is a 50% reduction in the contribution that the Haddock stock can make to the total harvest portfolio. I applied the precautionary quota constraint to Haddock because Cod is already severely depleted and there is little scope left for further quotas restrictions. Once this constraint has been included in the optimization, fisher's must choose a different set of species to achieve the same amount of revenue, maximizing their utility by selecting the portfolio with the lowest associated risk. The species weights of the resulting harvest portfolio are shown in Fig. 3.6.

Figure 3.6: Species Weights for the Cod, Haddock and Whiting Harvest Portfolio with Target Revenue of €13.16m under Actual Status Quo, Optimal Status Quo and Precautionary Scenarios

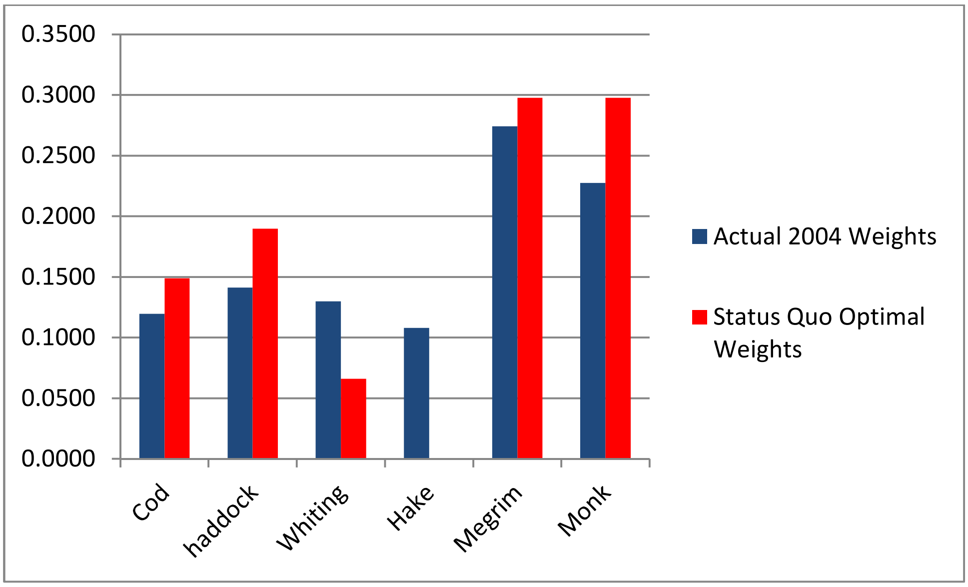


Given that it is an imposed constraint, the share of the fleet portfolio dedicated to Haddock is halved and is now 24.3%. While the expected revenue of the harvest portfolio has remained the same, the risk associated with it has increased, albeit by the least amount possible, therefore utility has fallen. Cod, which had previously accounted for 5.5% of the optimal portfolio without the constraint on Haddock, now accounts for 25%. Whiting has increased from 46% to 50.8%. The results show that Cod would be the main species to which effort from Haddock would be redirected if the fisher were free to do so (unconstrained by Cod fishing restrictions). This is not simply because Cod is a more attractive option (results from scenario 1 show that it is less efficient). A closer examination of Whiting reveals the fact that a 50.8% share of a portfolio with total expected revenue of € 13.16m is €6.69m (the expected revenue for Whiting is calculated using a decay factor of .741). In other words, this is the maximum expected revenue of Whiting based on historical averages.

If the system (biological or regulatory) allowed for any higher revenues to be generated from Whiting, then it would form an even greater fraction of the harvest portfolio when the precautionary constraint was placed upon Haddock. The fraction of Cod increased because it was not optimal to have a higher percentage of it at the outset, so it had not reached its limit. It therefore had more capacity within the optimization as an “alternative opportunity”. The percentage of Cod, Haddock and Whiting in the actual 2004 harvest portfolio was 30.6%, 36.2% and 33.3% respectively. So in reality, this capacity in the stocks of Cod does not exist. The result shows that a constraint on the permitted catch of Haddock in the fishery causes increased effort to be directed toward other species in the fishery. However, catches of Cod and Whiting are already at their upper bounds, or beyond them. Thus the multi-species impact of a precautionary constraint on any one of these species is therefore very unlikely to remain within the fishery.

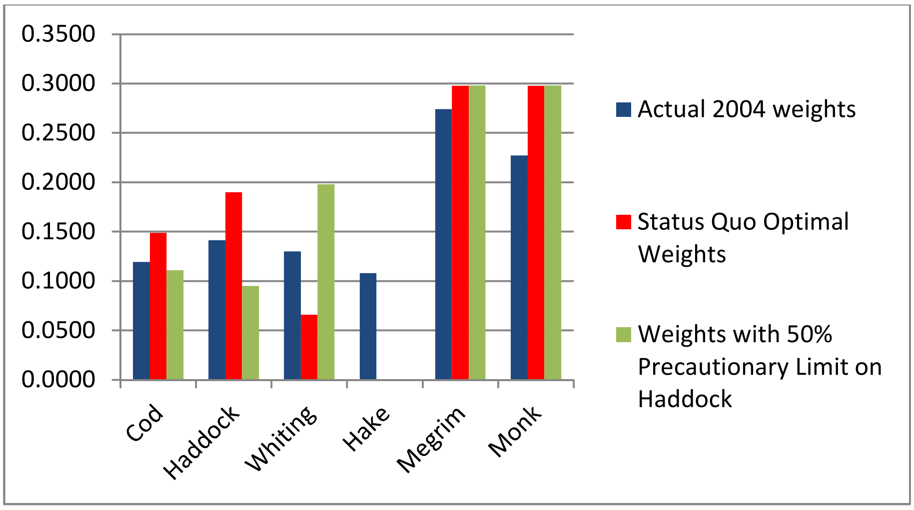
Continuing with scenario 2, I now consider the impact that the precautionary measure has on the Hake, Monkfish Megrim fishery. The actual 2004 Harvest portfolio for the two fisheries generated €33.68m in revenue. To attain this, the fleet selected a harvest portfolio with a standard deviation of €8.35m. The optimal harvest portfolio with revenue €33.68m would have had a standard deviation of €7.83m. Species weights for both portfolios can be seen in Fig. 3.7.

Figure 3.7: Species Weights for both Fisheries and a Harvest Portfolio with Target Revenue €33.685m



Hake is not selected at all in the optimization as it is characterized by relatively low expected revenue (high market value but low historical quantities) and a relatively high variance and therefore only enters the optimal portfolio at target revenues above €33.68m where the fleet will take on more risk for each unit of revenue. It also has a high positive covariance with Monkfish, which is far more efficient in terms of its revenue risk trade-off (see Tables 2 and 3). Upon inclusion of the precautionary constraint on Haddock, the standard deviation of the optimal harvest portfolio with a total revenue of €33.68m rises to €8.013m (up from €7.83m). The species weights are shown in Fig. 3.8. The weighting of Megrim and Monkfish in the portfolio does not increase simply because it cannot; upper bounds on weighting of these species in the portfolio had already been reached before the additional constraint; approximately 30% each of the €33.68m total revenue. However, in the actual 2004 harvest portfolio, Megrim and Monkfish constituted just 27.4% and 22.7% respectively. Given the attractive risk revenue profiles of these stocks, it is therefore *very* likely that a precautionary measure on Haddock would have a knock on affect in the Hake, Monkfish and Megrim fishery.

Figure 3.8: Species Weights for both Fisheries and a Harvest Portfolio with Target Revenue €33.685m under Actual Status Quo, Optimal Status Quo and Precautionary Scenarios

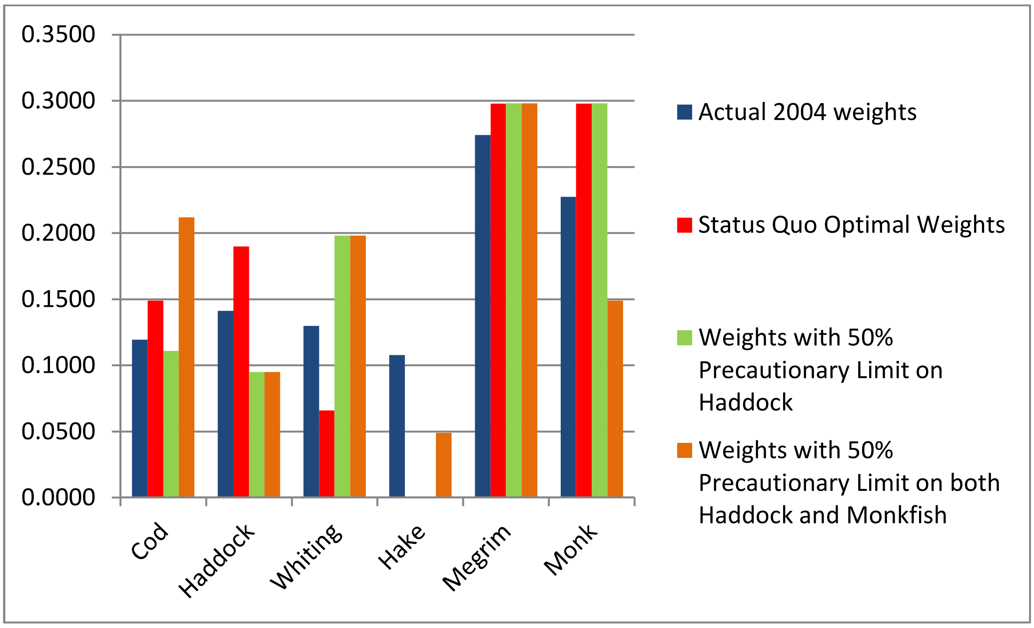


Scenario 3: Precautionary Measures in the Hake Monkfish Megrim Fishery

ICES claim that Monkfish is not over-exploited in relation to its precautionary limit. However, because of the severe decline of whitefish stocks such as Cod and Whiting in recent years, many of the traditional fleets have been affected. The result is an influx of new vessels into alternative fisheries. Where stocks in an alternative fishery are already exploited beyond optimal levels, its capacity to absorb increased exploitation rates is limited.

In this scenario, I implement a hypothetical precautionary quota restriction on Monkfish. The results of a 50% reduction in the contribution Monkfish can make to the harvest portfolio are shown in Fig. 3.9. The new optimal fleet harvest portfolio has a standard deviation of €8.471m.

Figure 3.9: Species Weights for both Fisheries and a Harvest Portfolio with Target Revenue €33.685m under Actual Status Quo, Optimal Status Quo and Precautionary Scenarios for Multiple Species



When precautionary constraints are placed on both Haddock and Monkfish, fisher utility is further reduced by forcing fishers to select a harvest portfolio with a higher level of risk in order to maintain status quo revenues. As the results suggest and as is outlined in the Cawley report, declining whitefish stocks force fishing effort into the Hake, Megrim and Monkfish fishery. However given a scenario of overfishing and precautionary quota constraints on Monkfish, expected revenues from this fishery diminish and the weighting of Cod in the harvest portfolio increases again. This suggests that any upside for Cod stocks that may arise from effort being redirected into the alternative fishery is beneficial only in the short term.

Table 3.4 sets out the impacts on species selection for varying precautionary measures. Increasing the precautionary limits on Haddock and Monkfish to 80% shows the outcome more pronouncedly, with the species weighting on Cod reaching its highest amongst all optimizations. Interestingly, it is infeasible to delineate a harvest portfolio with target revenue of €33.6m with precautionary limits this extreme. Despite a lower variance than harvest portfolios in other precautionary scenarios, the lower revenue associated with this portfolio results in the lowest possible utility for fishers amongst all scenarios. This captures the reality that fisheries today face. In the medium term, the impact of declining stocks can be offset by bearing more risk or investing in superior fishing technology. Eventually however, stocks are driven so low, that the only possible outcome is a fall in revenue. This is a classic result of overcapacity, and the impetus for measures such as property rights and fleet decommissioning.

Table 3.4: Species Weights under Various Precautionary Scenarios

Harvest Portfolio		Cod	Haddock	Whiting	Hake	Megrim	Monkfish	Portfolio Revenue (Euros)	St. Dev
2004 Observed		.1195	.1413	.1299	.1079	.2741	.2273	33,684,782.61	8,348,096
Precautionary Reduction in Haddock	Precautionary Reduction in Monkfish								
0%	0%	.1489	.1898	.0659	0	.2977	.2977	33,684,782.61	7,833,068
20%	0%	.13	.152	.123	0	.298	.298	33,684,782.61	7,898,937
	20%	.212	.038	.198	.016	.298	.238	33,684,782.61	7,986,150
	50%	.212	.038	.198	.105	.298	.149	33,684,782.61	8,188,782
	80%	.219	.039	.206	.165	.308	.062	33,684,782.61	8,487,715
50%	0%	.111	.095	.198	0	.298	.298	33,684,782.61	8,013,212
	20%	.171	.095	.198	0	.298	.238	33,684,782.61	8,130,674
	50%	.212	.095	.198	.049	.298	.149	33,684,782.61	8,397,052
	80%	.212	.095	.198	.138	.298	.06	33,684,782.61	8,777,342
80%	0%	.168	.038	.198	0	.298	.298	33,684,782.61	8,166,559
	20%	.212	.038	.198	.016	.298	.238	33,684,782.61	8,3284,38

3.6. Discussion and Conclusions

This Chapter used the portfolio theory framework to develop a model which might assist fishery managers and policy makers to better predict the likely changes in the composition of fishers' harvest portfolios when precautionary measures on a single species are implemented. Fleet expectations about species revenues and covariances were modelled for the Irish Cod, Haddock and Whiting and Hake, Monkfish, and Megrim fisheries using the historical averages of species prices and landed quantities. An exponential weighting factor, which captured fisher's inclination to weight recent events more highly in forming expectations about future events, was also used. Actual vs. optimal status quo (the result of the portfolio optimisations before constraints are added) species selection and the scope for potential efficiency gains were shown, both in terms of species weights and relativity to the efficient frontier. Hypothetical precautionary scenarios/constraints were also set out, and the potential impact of these measures on fisher's targeting choices and species weights in the harvest portfolio selection were assessed.

The results of the comparison between the actual and optimal status quo scenarios suggest that the Cod, Haddock, Whiting and Hake, Monk, Megrim fisheries may already engage in risk-return balancing behaviour, since the difference between portfolios was small and given data on cost and technical interactions, may be even less substantial. The quota change simulations indicated that large quota changes on a single species will lead to a greater retention in the live-weight catch of alternative species in a multispecies fishery. Furthermore, it was possible to estimate quantitative values for this, the idea being that fishery managers could use such predictions to inform them about changes to future harvest rates for various species in a multispecies fishery.

There were certain issues with the data and the methodology which limit the reliability of the results and which should be considered when interpreting results for policy recommendations. Primary data constraints were a lack of cost and technical interaction information. This information is very relevant to fishermen's final harvesting decisions and its omission compromises the assuredness with which the results can be interpreted. The consequences of such omission was minimised however by considering fisheries where variable costs are similar across species and fixed costs of entry do not exist between fisheries. Another issue is that variance is measured as changes in annual total revenue over time. This variation may be dependent on many factors, and therefore not strictly identically and independently distributed over time. It is also possible that the actual variance in species harvests is different to that estimated in this study because the data used represent only a fraction of retained catch in the fisheries under study, the rest of the retained catch being accounted for by non-Irish fleet.

There is scope for the portfolio framework to assist in setting multiple TACs across species. The ability to predict the direction into which fishing fleet will refocus targeting effort once quotas have shifted can help to inform policy makers about quotas required for other relevant species. Also, where negative multi-species/multi-fishery impacts of a protective measure are predicted, a structural rationalisation such as fleet decommissioning may be appropriate in the fisheries affected rather than simply relying on more precautionary quotas alone.

Overall, while development of the methodology and improvement in data would be needed for practical use, and future work in this area should take stock of this fact, the results suggest that there is scope for the portfolio theory framework to add value and assist fisheries managers in multi-species-based fisheries management. Further work implementing the methodology using simulated fleet cost functions as estimated by Rockmann et al. (2009) could lead to improved estimator accuracy. Multi-disciplinary collaboration with fisheries scientists and managers in design and application of the portfolio approach may also be fruitful. Finally, comparison of species weights resulting from alternative scenarios with the output of structural ecosystem models might shed further light on fleet behavioural dynamics following quota changes.

4. Shortcomings in the European principles of ICZM; assessing the implications for locally orientated coastal management using Biome Portfolio Analysis

4.1. Introduction

Despite the initial incorporation of Integrated Coastal Zone Management (ICZM) as one of the primary mechanisms of environmental policy geared towards sustainable development of European coastal areas, further evaluations have led to an awareness of the need for an updated ICZM initiative. In particular, the development of a Marine Strategy Framework Directive (MSFD) and an overarching Maritime Policy (EC, 2012; NWWRAC, 2007) is likely to assist in the European adoption of ICZM, since they may well provide the medium through which ICZM is shaped, implemented, and brought into legislation. The MSFD (EC, 2008) in particular recommends environmental and ecological indicators as a means of assessing current environmental status and to track effectiveness of the Directive's measures. The ability to update directive measures according to their performance across the marine regions is also outlined under the principles relating to adaptive management and spatial considerations. As such, the MFSD and the nature of the EU Commission's maritime policy very much reflect developments in the existing literature on what form ICZM should take and how it should be implemented (Breton et al, 2006; Deboudt et al, 2008; Diedrich et al, 2010; Forst, 2009; Meiner, 2010). In sum, what is emerging is the requirement of an integrated, spatially based form of coastal management which inherently addresses the issue of risk and uncertainty and is adaptive over time to allow for improvements which were not foreseeable in earlier versions of ICZM.

In this paper Biodiversity Portfolio Analysis (BPA) (Hills et al, 2009) is put forward as a management format which attempts to incorporate all of these requirements into its approach.

At first glance, it may seem unusual for a methodology stemming from the management of financial assets to have an application in the field of biodiversity conservation, but in recent years, researchers in that field have highlighted the suitability of the concept, due to its explicit trade-off between expected payoffs and exposure to potential risks/losses (Figge, 2008). Markowitz (1952) developed a quantitative definition of the relationship between the riskiness of an investment, and the expected return. Asset managers compose portfolios of assets such that both objectives (minimising risk but achieving a desired level of expected return) can be optimized. In a similar way, society must balance between two alternative decisions. One of these is to ensure healthy environmental status allowing society to consume the wide array of services that flow from healthy ecosystems. The other is allowing human activities which are economically necessary, to proceed. Limiting one to promote the other is at the heart of all environmental decision making. As Figge (2008) points out, ‘the expected benefit which society derives from species, genes or ecosystems is uncertain, but this risk can be partially diversified away by combining various species, genes or ecosystems in a biodiversity portfolio’. In this way, rather than making isolated environmental management decisions, society’s decisions between the two alternatives would focus on aggregate values of services and risks, aiding the decision making process and allowing for optimality in the trade-off between the two societal goals. It should be noted that the approach does not go all the way to valuation of ecosystems, but rather looks at survey participants perception of ecosystem ‘service values’ and considers their trade off with participants’ perceptions of risks.

Aside from the need to update ICZM and tailor it to recent policy recommendations, more critical problems exist for the ICZM paradigm as a whole. ICZM is steeped in the notion that stakeholder participation in environmental decision making can improve the effectiveness of environmental management initiatives. When the failure of environmental management initiatives is due to a lack of coordination between various stakeholders and centralised coastal managers, there is truth to this latter notion. However, there may be instances where a lack of coordination is not the source of the problem. Environmental problems, and the failure of management responses to solve them, may arise not out of poor coordination, but out of entirely contradictory environmental/societal and economic/stakeholder goals. Simply put, there may exist, economic incentives for stakeholders to act against the environmental goals of society. In such an instance, environmental problems may not be solvable by group discussion and consensus, but by prioritization of, in all likelihood, centralised coastal management decisions. While management integration may be desirable, deciding between top-down (managerially centralised) and bottom-up (locally based) management decisions is not only a methodological issue, but a political, legislative and philosophical one. To explore this issue further, this paper analyses the differences in the attitudes of marine scientists and local stakeholders towards an environmentally sensitive area of the Irish/European coastline.

BPA is employed as the format through which to explore some of these problems in this study. In this way the validity of BPA as a tool for ICZM is tested (given the updated status of European maritime policy) but it also serves as a medium for highlighting the extent of the implications of having contradictory ICZM principles in EU policy recommendations. Section 2 discusses the emergence of ICZM and some of the literature which identifies specific problems with the concept. Section 3 outlines the BPA methodology. Section 4 presents the background to the study area and section 5 presents the results. Section 6 includes a discussion and the conclusion of the paper.

4.2. Strategic versus local Principles in ICZM

The concept of ICZM emerged in the scientific community of the 1970s, developed through the 1980s and entered the international political scene during the Rio Earth Summit of 1992 (Billé, 2008). The European Union Recommendation of 2002 outlined 8 core principles which a European adoption of ICZM should include (EC, 2002). McKenna et al. (2008) divide these principles into three distinct groups, listed here as they appear in the paper:

1. Two 'procedural' principles: support and involvement of relevant administrative bodies and use of a combination of instruments that are focused on the attributes of the methods and procedures that might be used to best advance ICZM

2. Three 'strategic' principles: broad overall perspective, long-term perspective, and working with natural processes. These principles mainly focus attention on long-term goals, and fit easily into the sustainability ethos that dominates contemporary environmentalism.

3. Three essentially 'local' principles: local specificity, adaptive management during a gradual process, and involving all the parties concerned. These can be regarded as a balancing set to the second group, because they focus interest on specific areas and problems, encourage tailoring of management to local conditions and encourage the participation of the public in formulating management policy.

McKenna et al. (2008) claim that because the principles are presented as a menu of free-standing options, with no prioritization either within or between groups, irreconcilable differences in strategy arise. Billé (2008) also argues that the idea that all conflicts can be resolved with a consensus agreement is a simplistic belief which arises out of three flawed assumptions; firstly, that environmental management is a problem of coordination, secondly, that consultation is the solution to this lack of coordination and thirdly, that consultation is inseparable from consensus. Billé (2008) also raises a further criticism of ICZM which he refers to as the positivist illusion. Many calls for improved management of coastal areas stress the need to develop the scientific understanding of marine and coastal ecosystem processes (Botsford et al, 1997; POC, 2003; Pikitch et al, 2004). However, many natural processes are (and will remain) far beyond the reach of scientific understanding. For example Johannes (1998) demonstrates theoretically that the inception of a rational management of Indonesian coral reefs alone would require at least 400 person-years to collect the necessary data, a process which would have to be repeated annually.

Realistically, management of coastal areas involves making decisions under imperfect knowledge and uncertainty. Collating explanatory data about human and ecosystem processes until definite outcomes can be predicted (while something to be strived for) cannot realistically be the precursor to every management decision. The therefore subjective reality of management decisions, as opposed to the positivist illusion, can make management decisions affecting the economic, cultural and social goals of the local community controversial in nature. Examples of controversial environmental legislation are abundant; constraints on commercial fisheries such as catch quotas and marine protected areas have significant impacts on the livelihoods of fishing communities, input constraints on agricultural production, designed to attain set levels of environmental standards, reduce agricultural output and elsewhere, Hynes and Hanley (2006) document the conflict between typical water use values and hydro-electric schemes on “wild” rivers. In any of these examples, scientific diagnosis about the environmentally damaging effect of the practice in question, and predictions about the subsequent benefits of said constraints, is subject to scientific uncertainty (Kinzig and Starrett, 2003).

The reality then is that while scientific understanding about environmental processes may not be in a position to perfectly inform society and its policy makers on the optimal use of environmental resources, decisions still have to be made. The objective of any approach to environmental decision making then, must be to provide environmental manager's with the best information possible, and a feasible way of making decisions that can optimise resource use (Fabbri, 1998). Since deciding between management alternatives will unavoidably involve qualitative, as well as quantitative distinctions, the decision making process requires a modelling framework which assists in this. Such assertions support the basis of using BPA, given its integrated, qualitative and spatial framework and the next section examines how such a technique might be used in practice. Furthermore, the contrast between scientific and stakeholder perceptions can be quantified by comparing their feedback in the BPA analysis to one anothers'.

4.3. Methodology

The Biodiversity Portfolio Analysis (BPA) as developed by Hills et al. (2009), is a spatially orientated framework which marries the input of the local and scientific communities, stakeholders and local agencies to form a broad overview of the contribution that various geographical biome types in the local area make to society. It is intended to assist coastal managers in deciding between alternative policy decisions by allowing for a qualitative assessment of their impacts on the cultural, social, economic and environmental services that the various biome types of an area provide. One of the attractive features of BPA is that it incorporates threats/risks to the biomes under study into the analysis, and uses this information when balancing between alternative management strategies. It is a derived from the financial portfolio theory of Markowitz (1952) and therefore deals explicitly with optimal trade-offs between risk and return across diverse assets.

BPA requires the identification of geographical areas or "biomes" from which ecosystem services, and hence the value of their societal "returns", are derived. Associated with each ecosystem service/return, and thus each biome, is a risk to the return in terms of the scale of the extent and seriousness of various threats. In this case then, the "assets" in question are environmentally sensitive biomes which derive their anthropocentric value from the multiple market and non-market services that their biogeographic features provide to society.

According to Hills et al. (2009), once a basic understanding of biomes, risks and returns for a study area is built up, various scenarios can be developed based on possible management interventions; these scenarios can be assessed for their effect on the risk and return of the biodiversity portfolio. Four key sets of data are required for the framework to be operational; biome type, spatial area of biome, services arising out of each biome and threats to each biome's functions. The degree of return for the study area's biodiversity portfolio can be defined as:

$$\sum_{i=1}^{n} esv_i * ba_i \qquad (1)$$

and the degree of risk as:

$$\sum_{i=1}^{n} bt_i * ba_i \qquad (2)$$

Where esv_i is the ecosystem service value of biome *i*, ba_i is total area of biome *i*, and bt_i is the scale of threat on biome *i*. The method requires values be placed upon the overall return from each biome, the mean return of all biomes in the study area, the risk/threat to each biome and the mean of the biome risks in the study area. Risk and return values are evaluated on Likert scales allowing for direct comparison of the trade-offs between the two. Local stakeholders and/or scientists with knowledge of the study area provide the rankings on these scales for the alternative biomes. Typically, in a non-market valuation study, researchers attempt to estimate the monetary value of an environmental asset. BPA is not generally concerned with monetary values, but requires a scaling system which treats market and non-market services and risks with equal weight. This is a crucial strength of the BPA approach and what sets it apart from typical cost benefit analysis.

The attributed scale values allow for a comparison of the risks and returns derived from different biomes, the relative positioning of the biomes when plotted on a scatter gram plot (in terms of risk and return) and identification of the relationship between the risk and return of different biomes. The latter exercise, where correlations in the risk factors across all biomes are identified, is one of the potentially most useful tools of BPA. Correlations are identified by determining the extent to which biomes have a common response to threats. Biomes with a similar response to threats have positive correlation, biomes with an alternative response to threats have negative correlation and those showing neither alternative nor similar response are not significant. There are three terms for describing the extent of any relationship between biomes:

Independent: where correlation between any pair of biomes is not significant, then the threat factors for these biomes are not related, i.e. they respond in an independent fashion to threat factors;

Associated: where the correlation between risk factors for a pair of biomes is significant and positive, then the threat factors impact upon the biomes in very similar way;

Resilient: where the correlation between any pair of biomes is significant and negative, then threats that can greatly impact upon the ecosystem services in one biome tend to have little impact upon the other biome.

Hills et al. (2009) also develop the notion of portfolio impact sensitivity (PIS) which they calculate by scoring all of the biome pairs as +1, -1 or 0 according to whether they are associated, resilient, or independent, respectively, and then sum the values to determine the biome portfolio's overall level of association/resilience. For example, a portfolio made up of biomes that are largely associated with each other in terms of responsiveness to threats will have very high PIS, whereas the opposite is true of a portfolio made up of resilient biomes. The lower the PIS of the biome portfolio, then the easier it will be for manager's to make decisions that yield highest possible biome returns while "containing" risks. A higher PIS value however will mean that a single or small number of threats could have a highly negative impact on the ecosystem services arising out of many of the biomes within the management area.

Integral to the methodology is the process of determining each biome in the study area as well as the ecosystem services provided by that biome and the risks/threats it is exposed to. As a starting point, this is achieved through reference to the relevant literature (Hills et al, 2009; Costanza et al, 1998). However, given that BPA is intended as an integrated and stakeholder engaged management format, this information is also compiled through the organisation of a locally based stakeholder workshops. Also during the workshop, values for the biome risks and services are assigned by the participants. The depiction of local attitudes is therefore captured by the results of a stakeholder analysis workshop.

One of the aims of this study was to identify the extent of the potential “attitude gap” between the scientific community and local inhabitants with respect to the question of how ecosystem services rank in terms of their anthropocentric value and to what extent risks, both manmade and environmental, threaten the provision of those services. In order to do this it was also necessary to attain a depiction of a scientific or strategically minded attitude towards the same biomes, services and risks, which featured in the stakeholder analysis. Therefore, in addition to the stakeholder analysis, a second analysis was carried out, using the same survey, but valuing services and risks according to the opinions of marine scientists with in-depth knowledge of the study area. The survey format used in both consultations can be viewed in Appendix B.

Through the consultation with the stakeholder group, all ecosystem services and risks were rated on a scale of 0 to 3 where:

0 : *{if service = negligible ecosystem service provided*

{if risk = threat factor has no impact

and:

3 : *{if service = extensive to complete service provided*

{if risk = threat could destroy biome function

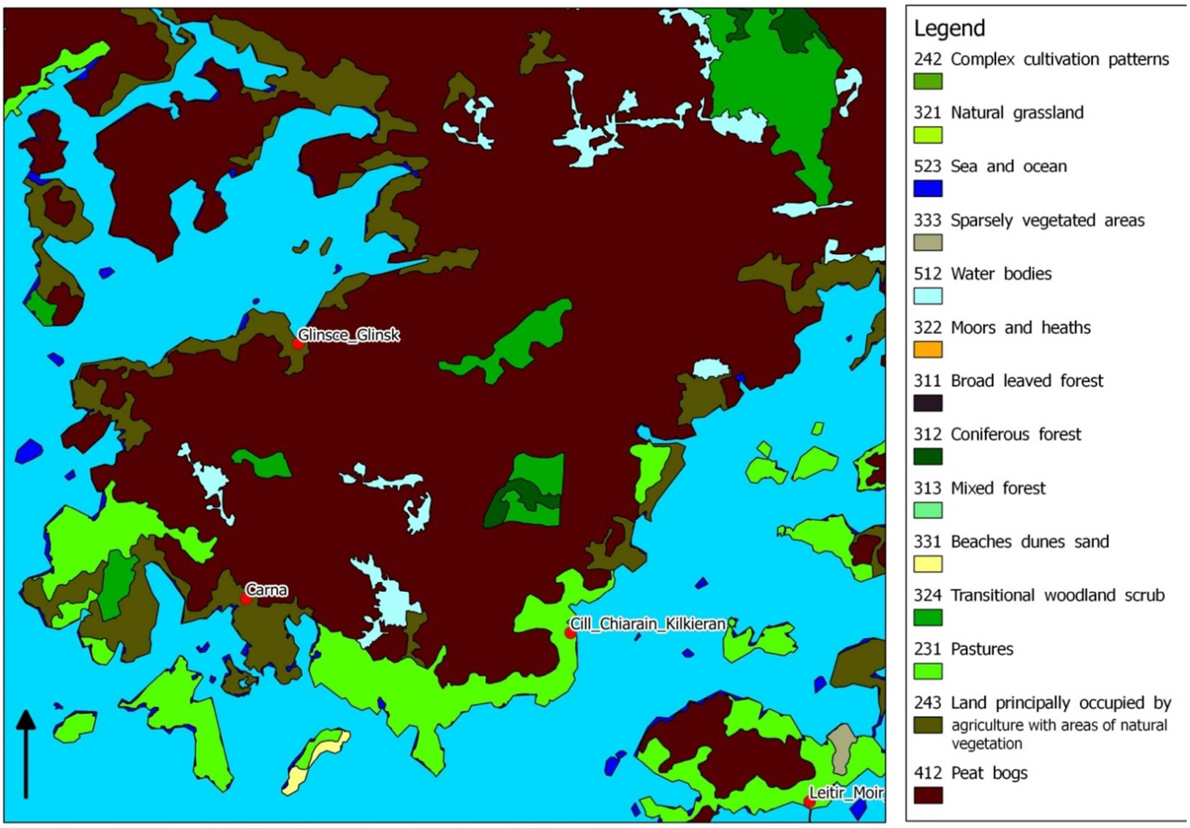
where ‘extensive to complete’ is taken to mean that the service is highly valued and present in the area.

In later consultations with scientific opinion, the very same analysis was carried out, recording the values assigned to ecosystem services and threats to allow for comparison with the results of the stakeholder analysis.

4.4. Coastal Study Site and Selection of Biomes, Services and Threats

The study area in which the BPA was based was the peninsula of Iarras Aithneach, which is located in Connemara on the western coast of Ireland to the South of the Connemara Mountains (the Twelve Bens), West of Lough Corrib and North of Galway bay. Geographically, this area is characterised by hilly, rocky, low yielding agricultural land, blanket bog, heathland and coniferous forest. Being a peninsula, the landscape is bounded by a rocky coastline interspersed with sand beaches. There are also many small freshwater loughs distributed around the landscape. A map of the Iarras Aithneach study area, its location in Ireland and the associated biomes identified in the stakeholder analysis are shown in Fig.4.1.

Figure 4.1: Iarras Aithneach Peninsula in Co. Galway



This area was specifically selected due to the diverse biogeographic features by which the peninsula is characterized. The coastal inlets around Iarras Aithneach are the location of previous, current or potential aquaculture projects which offer good economic and employment opportunities to the local community but are controversial from an environmental perspective (Ellingsen et al, 2009; Liu et al, 2007; Liu et al, 2010). Unique habitat types which Iarras Aithneach possesses warrant strategic conservation measures, but often these conservation measures can clash with local economic goals. For example the cutting of turf on local peat bogs is prohibited, which directly increases the costs of fuel consumption for the average household in Iarras Aithneach. Moreover, the harvesting of peat is a tradition in itself and has a cultural value for the community. Other areas amongst the diverse terrain types are of cultural importance in the area, such as the many small islands off the coast of Carna. Some of these islands were once inhabited by the same ancestors of those living in the community today, and islands such as Oileán Mhic Dara are part of an annual traditional religious pilgrimage. The economic, social, cultural and ecological trade-offs that exist for the management of this area make it ideal for an assessment of potential incoherence in the local and strategic “free-standing” principles of the European Council recommendation on ICZM. Free standing in the sense that each should not have its principles undermined by other elements of ICZM legislation.

There were a total of eight biomes identified in the area for inclusion in the BPA. In some cases, biome types existed to only a negligible degree in the Iarras Aithneach study area, and sufficient data for their inclusion in the study did not exist. Removed biome types were salt marsh and shallow water. Residential/commercial space is not included in the analysis. The main reason for this is that it does not constitute a “biome” in the sense that it is not a major regional or global biotic community dominated by plant life. Despite this, case studies of ICZM for large urban densely populated areas can include residential/commercial space in the analysis when its scale within the biomes under study is significant (Anilkumar et al, 2010). The Iarras Aithneach peninsula however, being 21,262 square kilometres in size and having a population of 1,838 (CSO, 2011), has a population density of only 0.086 per square kilometre, thus the scale of human uptake of biome space is low. Furthermore, the population is widely dispersed around the study site rather than condensed within a particular urban area. Census 2011 data indicates that less than 16.38% and 15.56% of the population inhabit the area’s villages of Carna and Kilkieran respectively. In fact these percentages are much lower in reality because the census 2011 data is not at a fine enough spatial scale to represent the immediate village boundaries alone, and includes some surrounding land areas.

In some cases, biome types were grouped under one category. This was done when various classes of biome type were deemed to have the same resource use and exploitation patterns. Sand inlets, sand dunes, shingles and rock platforms were categorised as, ‘sand beaches’.

Other biome types unique to the area were added to the list such as peat bogs, agricultural land (which also included pastures) and coniferous forest. There are many biome types excluded from the analysis since they do not occur in the geography of the study site. For example, Costanza et al. (1998) arrive at an economic valuation of biomes such as coral reef, tropical forest and desert, amongst others. From the stakeholder analysis it also emerged that coastal islands should constitute a separate biome type, due to their significant cultural and historical value to the local community. The water bodies biome accounted not just for the many fresh water Loughs of the area but also for the diverse network of rivers also. A definition of each biome is included in Appendix C.

All maps of the study area and biomes contained within it were created using ArcGIS and Corine Land Cover data which is a digital map of the European environmental landscape. Corine Land Cover 2006 is the third dataset in a series, the previous datasets corresponding to base years of 1990 and 2000. The ecosystem services included in the study were identified through a combination of stakeholder analysis and referral to relevant literature (Hills et al, 2009; Costanza et al, 1998). They are agriculture, fishing, aquaculture, intertidal gathering, sand/grave/rock/peat extraction, conservation interest, recreation and tourism, cultural/educational, flood protection/coastal defence, nutrient/waste absorption, renewable energy generation and land-take (car-parks/range/causeways).

Risks/threats to the biomes were identified through a combination of stakeholder analysis and referral to relevant literature (Hills et al, 2009; Costanza et al, 1998). These included: climate change, erosion, flooding (including sea level rise), saline intrusion (the influx of sea/salt water into an area that is not normally exposed to high salinity levels, for example, the study sites agricultural land and many freshwater lakes), tourism and recreation impact, new causeways and other infrastructure, agricultural change, pollution (including oil spills), invasive species, marine and terrestrial litter/dumping, over-gathering of shellfish/overfishing, over-regulation and salt damage. Salt damage was added as a risk/threat type during the stakeholder analysis as some participants felt it captured additional forms of damage that can arise from the sea's salinity, in particular, airborne salt damage to property and vegetation. This risk type took on an extra dimension in the scientific consultations, as the risk of salt damage was deemed to apply to the sea and ocean biome itself, given the threat of changes in oceanic salinity which could arise as a result of climate change.

4.5. Results and Analysis

The stakeholder group consisted of 14 individuals; three inshore fishermen, two small farmers, a hotel employee, several homemakers and community representatives such as a journalist, guard (local police enforcement), school teacher and priest. A local project development officer of Udaras na Gaeltachta, (a regional authority responsible for the economic, social and cultural development of the Gaeltacht -Gaelic speaking areas-) was also present. Many of the stakeholders present were members of Forum Iarras Aithneach; a community committee made up of local inhabitants in the area, so they were well known in the community and had a good understanding of social and economic issues affecting its habitants. Dialogue between stakeholders during the survey was encouraged, allowing participants to discuss their views and share their knowledge about any particular issue. The stakeholder group was small, but at the same time, representative of a small rural community and able to represent the types of concerns that predominate such an area. There was also a focus on explaining scientific terms and concepts as they related to ecosystem services and risks in the area.

The list of identified biome types, related ecosystem services, and the values assigned to them from the stakeholder workshop are displayed in Table 4.1(a) and 4.1(b). Each value that the stakeholder group attributed to an ecosystem service is shown in part (a) of Table 4.1, but in part (b) it is scaled to the spatial area of the biome type it arises out of. For example, the ecosystem service "fishing" derived from the biome type "coastal water" received an ecosystem service value of 3 from the workshop participants. Scaling this to the spatial area of the biome, which is 10,490 square kilometres, means a return of 31,470 is generated from fishing in the sea and ocean. For any biome type, the total of all ecosystem service values arising out of that biome are scaled and then summed to determine the total return of the biome. Continuing the example, coastal water also receives a non-zero value for the ecosystem services aquaculture, conservation interest, recreation and tourism, cultural/recreational and nutrient/waste absorption. In total, stakeholders gave all of the services provided by coastal water a value of 17, meaning that once spatially scaled, the coastal water contributes 178,338 credits to the total return of the localities biome portfolio.

Biome return, or ecosystem service value for each biome, prior to spatial scale being considered, can be viewed in the penultimate row of Table 4.1(a). At this stage, coastal water provides the greatest return, followed by sand beaches and coastal islands, with peat bogs and agricultural land also receiving a high valuation. The lowest valued biome returns were coniferous forest, coastal lagoons and water bodies, respectively.

The penultimate row of Tabl4.1(b) shows biome return when scaled according to biome area. The result is a substantial shift in the ranking of returns in the biome portfolio. For larger values, it is useful to view biome returns in terms of their proportionate contribution to the total biome portfolio return. The final row of Table 4.1(b) gives the normalised value of all biome returns. This is simply the return of an individual biome divided by the sum of total biome portfolio return. The coastal water biome remains the biome providing the greatest return, since in addition to its high valuation in the stakeholder analysis, it constitutes a large part of the study area. However the next greatest contribution to portfolio returns come from peat bogs. Agricultural land, coastal islands, water bodies and coniferous forest make the subsequent, descending contributions to biome portfolio return. Coastal lagoons and sand beaches make less than a 1% contribution to portfolio return which appears as zero due to rounding error.

Clearly, the inclusion of biome's spatial areas in the calculation of return alters the rankings of biome returns considerably. The peat bog biome has an area of 13,500 square kms, dominating Iarras Aithneach landscape. This compares to biomes like agricultural land and coastal islands, which have an area of 3,293 square kms and 2044 square kms respectively. As a result, they are overtaken by the peat bog biome in the ranking of return provision. On one level this is a legitimate re-ranking of biome returns; if there is a greater supply of a biome then there is a greater provision of its services. The flip side is the impact this has on the valuation of returns from very small biomes. For example, despite the fact that sand beaches received the second highest return through the stakeholder analysis, its contribution to portfolio return is less than 1% due to its extremely small (just 37 square kms) spatial area[12].

Coastal managers may also want to observe the value of an ecosystem service across *all* biomes, not only its contribution within *one*. For this reason the penultimate *column* of Table 4.1(b) shows the values an ecosystem service is given relative to, and scaled to, the biomes

[12] While it is desirable from a spatially orientated perspective to relate the value of a biome's services to its area, it can also mean that highly valued services from small biomes all but disappear from overall contribution percentages. This is a concerning feature of BPA. Suppose for example that a rare and endangered species exists in the coastal lagoon biome. A non-market value may produce a very high existence value for the species, which is completely reliant on the coastal lagoon biome for survival. However, because the biome contributes so little to total portfolio return, BPA, in this case, could be used to justify decisions which threaten the biome, for example, land-take development. In reality, small biomes should be assigned much higher substitution values than larger biomes, simply because there are less of them. This example highlights why neither non-market valuation or BPA should be relied upon in isolation but should be just 2 elements within a suite of tools used for ICZM decision making. This issue is further discussed in the discussion section of the paper.

from which the service is provided, summed across all biomes. The last column in Table 4.1(b) gives the proportionate contribution of each service to the total portfolio return.

Table 4.1a: Estimated ecosystem service values of biomes based on stakeholder analysis

Service	Biome Type							
	Coastal Waters	Coastal lagoons	Water bodies	Sand Beaches	Coastal Islands	Coniferous forest	Agricultural land	Peat bogs
Agriculture	0	0	3	2	2	1	3	3
Fishing	3	1	3	0	3	0	0	0
Aquaculture	3	0	1	0	0	0	0	0
Intertidal Gathering	3	0	0	3	0	0	0	0
Sand/gravel/rock/peat extraction	0	0	0	0	0	0	3	3
Conservation interest	2	2	0	3	3	2	1	1
Recreation and tourism	3	0	1	3	2	0	1	2
Cultural/educational	3	0	0	2	3	0	2	3
Flood protection /coastal defence	0	0	0	3	2	0	0	0
Nutrient/waste absorption	0	0	0	0	0	0	0	0
Re. Energy generation	0	0	1	0	0	0	0	0
Landtake (carparks/range/causeways)	0	1	0	0	0	0	2	0
Total Service	17	4	9	16	15	3	12	12
Area of each biome (sq km)	10490	31	1306	37	2044	1171	3293	13380

Table 4.1b: Product of biome spatial areas and estimated ecosystem service values based on stakeholder analysis

Service	Biome Type									
	Coastal Waters	Coastal lagoons	Water bodies	Sand Beaches	Coastal Islands	Coniferous forest	Agricultural land	Peat bogs	Total value for each service	Norm. Value for each service
Agriculture	0	0	3918	74	4088	1171	9879	40140	**59270**	**13.94%**
Fishing	31470	31	3918	0	6132	0	0	0	**41551**	**9.78%**
Aquaculture	31470	0	1306	0	0	0	0	0	**32776**	**7.71%**
Intertidal Gathering	31470	0	0	111	0	0	0	0	**31581**	**7.43%**
Sand/gravel/rock/peat extraction	0	0	0	0	0	0	9879	40140	**50019**	**11.77%**
Conservation interest	20980	62	0	111	6132	2342	3293	13380	**46300**	**10.89%**
Recreation and tourism	31470	0	1306	111	4088	0	3293	26760	**67028**	**15.77%**
Cultural/educational	31470	0	0	74	6132	0	6586	40140	**84402**	**19.86%**
Flood protection /coastal defence	0	0	0	111	4088	0	0	0	**4199**	**0.99%**
Nutrient/waste absorption	0	0	0	0	0	0	0	0	**0**	**0.00%**
Re. Energy generation	0	0	1306	0	0	0	0	0	**1306**	**0.31%**
Landtake (carparks/range/causeways)	0	31	0	0	0	0	6586	0	**6617**	**1.56%**
Total Service Value S for each biome	**178330**	**124**	**11754**	**592**	**30660**	**3513**	**39516**	**160560**	**369184**	**100%**
Normalised value for each biome (% scale)	**42%**	**0%**	**3%**	**0%**	**7%**	**1%**	**9%**	**38%**	**100%**	

Table 4.2a: Estimated risk value to ecosystem biomes based on stakeholder analysis

Risk/Threat	Biome Type							
	Coastal Waters	Coastal lagoons	Water bodies	Sand Beaches	Coastal Islands	Coniferous forest	Agricultural land	Peat bogs
Climate Change	0	0	0	0	0	0	0	0
Erosion	0	0	0	1	1	0	0	0
Flooding (inc. Sea level rise)	0	1	1	0	1	0	0	0
Saline intrusion	0	1	0	0	0	0	0	0
Tourism and recreation impact	0	0	0	0	0	0	0	0
New causeways and other infrastructure	0	0	0	0	0	0	0	0
Agricultural change	0	0	0	0	0	1	3	1
Pollution inc. oil spills	1	0	0	0	2	0	0	0
Invasive species	2	0	0	0	0	0	1	1
Marine and terrestrial litter/dumping	1	0	0	0	0	0	0	0
Overgathering of shellfish/overfishing	1	0	0	0	0	0	0	0
Other 1 (Over-regulation)	0	0	0	0	0	0	0	3
Other 2 (Salt damage)	0	0	0	0	0	0	0	0
Total Risk	5	2	1	1	4	1	4	5
Area of each biome (sq km)	10490	31	1306	37	2044	1171	3293	13380

Table 4.2b: Product of biome spatial area and estimated risk value to each biome based on stakeholder analysis

Risk/Threat	Coastal Waters	Coastal lagoons	Water bodies	Sand Beaches	Coastal Islands	Coniferous forest	Agricultural land	Peat bogs	Total value for each risk	Norm. Value for each risk
Climate Change	0	0	0	0	0	0	0	0	**0**	**0.00%**
Erosion	0	0	0	37	2044	0	0	0	**2081**	**1.45%**
Flooding (inc. Sea level rise)	0	31	1306	0	2044	0	0	0	**3381**	**2.36%**
Saline intrusion	0	31	0	0	0	0	0	0	**31**	**0.02%**
Tourism and recreation impact	0	0	0	0	0	0	0	0	**0**	**0.00%**
New causeways and other infrastructure	0	0	0	0	0	0	0	0	**0**	**0.00%**
Agricultural change	0	0	0	0	0	1171	9879	13380	**24430**	**17.05%**
Pollution inc. oil spills	10490	0	0	0	4088	0	0	0	**14578**	**10.17%**
Invasive species	20980	0	0	0	0	0	3293	13380	**37653**	**26.28%**
Marine and terrestrial litter/dumping	10490	0	0	0	0	0	0	0	**10490**	**7.32%**
Overgathering of shellfish/overfishing	10490	0	0	0	0	0	0	0	**10490**	**7.32%**
Other 1 (Over-regulation)	0	0	0	0	0	0	0	40140	**40140**	**28.02%**
Other 2 (Salt damage)	0	0	0	0	0	0	0	0	**0**	**0.00%**
Total Threat Value R for each biome	**52450**	**62**	**1306**	**37**	**8176**	**1171**	**13172**	**66900**	**121,281.00**	**100%**
Normalised value for each biome (% scale)	**37%**	**0%**	**1%**	**0%**	**6%**	**1%**	**9%**	**47%**	**100%**	

The format for valuing biome risks (and by valuing, 'giving a quantity', is meant), shown in Table 4.2(a) and 4.2(b), follows that for valuing ecosystem services. Once a risk is identified; its threat to each individual biome is valued by the workshop participants and then scaled according to biome areas. Total threat to each biome and the proportional contribution of that biome to total biome portfolio risk is shown in the last 2 rows of Table 4.2(b). The total value of each threat type across all biomes and the proportional contribution of that threat to total biome portfolio risk is shown in the last 2 *columns* of Table 4.2(b). An immediate glance at the data collected on biome risks shows that risk was given far lower values than services from equivalent biomes. While the coastal water biome is valued as that with the greatest exposure to risk, once areas are included in the calculation of biome risk, peat bogs becomes the most at risk biome. The least at risk biomes, are coastal lagoons and sand beaches, both before and after spatial scale has been included in the calculation. Interestingly, the greatest threat across all biomes is over-regulation, one of the risks identified during the stakeholder workshop. The identification of this threat and high value it received in the stakeholder analysis, coupled with the low values given to most other environmental risks, represents a subtle indication of the divide that exists between local and strategic mind-sets in coastal development, and the feelings that exist amongst stakeholders about the implications of environmental protection and its impact on local livelihoods.

The relationship between the normalised risk and return for each value is shown in Fig. 4.2. A high return relative to risk ratio in a biome can be considered more desirable since is provides the returns related to the biome with less threat of loss of those returns. The ratio acts as an indicator as to which biomes coastal managers can focus on in order to maximize the biome portfolio return relative to risk. Policies and management decisions that can lower the exposure of biomes to risk or increase services without affecting risk exposure, improves the return relative to risk profile of the portfolio. Coastal waters and peat bogs provide the greatest return and are associated with the greatest risk. If risks affecting coastal water and peat bogs were addressed and reduced, this would lead to a substantial increase in the return relative to risk ratio of the Iarras Aithneach biome portfolio. For example, risks to the coastal water biome which received a positive value in the stakeholder analysis were pollution, invasive species, marine dumping and over-gathering of shellfish and overfishing. Policy steps which successfully reduced these risks would constitute a positive contribution to the risk return profile of the areas biomes. The same can be said for any risk amongst any of the biomes; however, the impact would be most noticeable for large biomes.

Figure 4.2: Normalised risk-return profiles of all portfolio biomes relative to spatial area of each biome

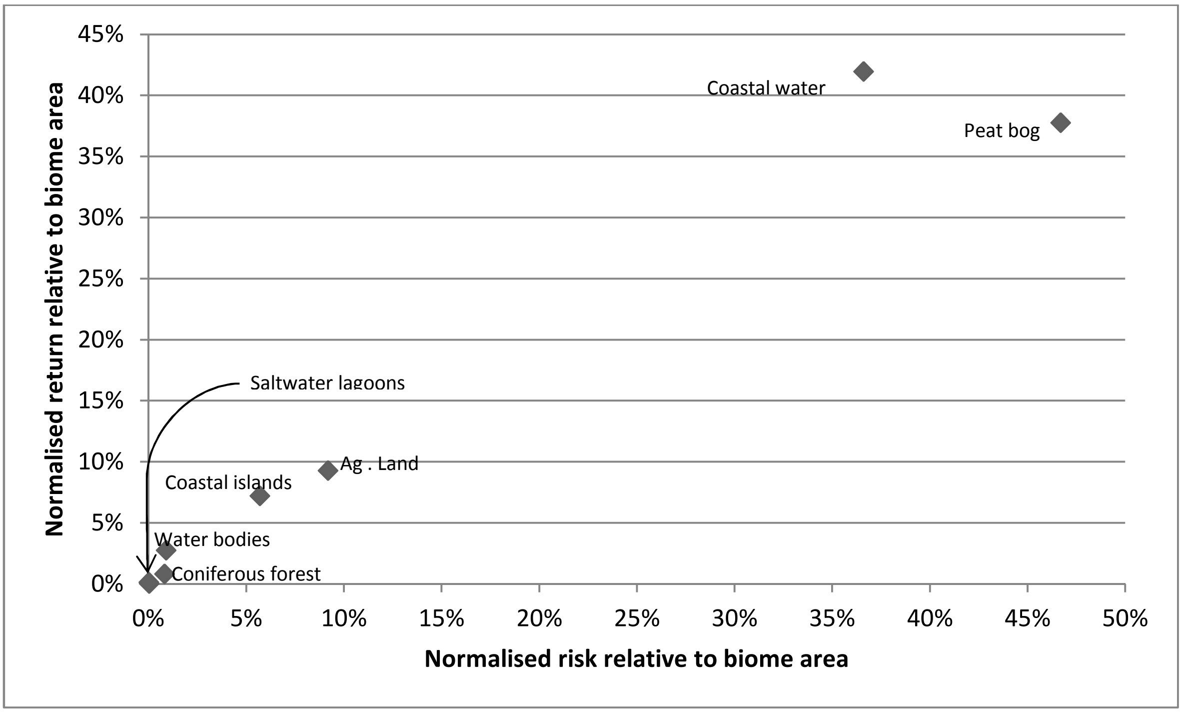
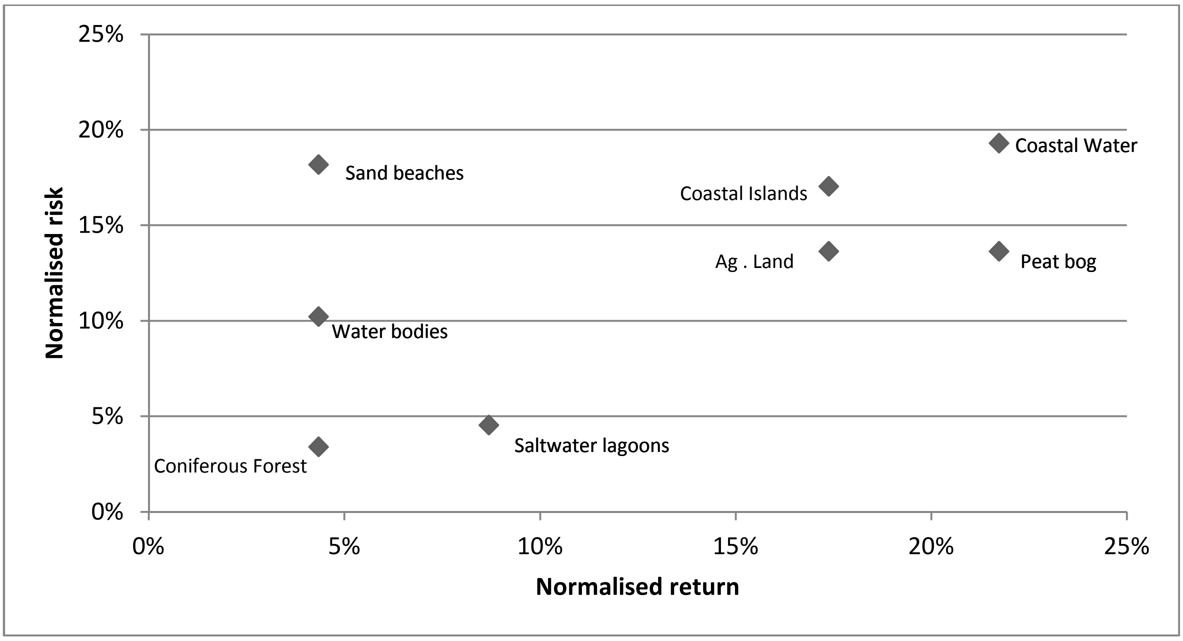


Figure 4.3: Normalised risk-return profiles of all portfolio biomes regardless of spatial area of each biome



The spatial magnitude of some biomes means that they dominate the Iarras Aithneach peninsula's landscape and therefore the results, since calculations are spatially based. For this reason, the risk return profiles of the biomes are also depicted *before* spatial area has been included in the calculations of risk and return. The extent of the transformation brought about by inclusion of biome area in the final calculation of biome risk or return warrants this. Fig. 4.3 shows the risk return relationship for each biome prior to being scaled according to area. The result of depicting biome risk-return relationships in this way is a much more in-depth and diversified portrayal of which ecosystem services local stakeholders attach value to.

One of the potentially most useful tools of BPA is its ability to assign a risk correlation to two biomes, indeed, the risk correlations amongst all biomes in the portfolio. Since any threat can relate to multiple biomes, understanding the common sensitivity of these biomes to risk informs coastal managers about the responsiveness of portfolio risk to various hypothetical scenarios. The risk correlation is categorised using Pearson's r statistic[13] and depending on the value of this calculation, biomes can be associated, resilient or independent. If two biomes tend to score highly for the same types of risks, then they will have a significantly positive pairwise correlation (associated). If many biomes within the portfolio are associated, the portfolio will have high portfolio impact sensitivity (PIS). For such an area, biome portfolio risk can be reduced most efficiently by tackling those risks which are common to the majority of biomes, as this will lead to the greatest reduction of portfolio risk, and therefore the greatest improvement of the biome portfolio's risk return profile.

If portfolio biomes tend to score highly for alternative risks (are resilient), significantly negative pairwise correlation will dominate the portfolio, which will therefore have a negative PIS value. In this case, decisions which further expose an individual biome to risk will not expose other biomes in the portfolio to the same risk. This suggests that overall biome portfolio return can be increased by developing biomes productivity, (for example through agriculture, fishing and resource extraction) to derive more returns to the community (since the associated risks are confined to individual biomes). It is important to note that such development should not overly exacerbate exposure to risk in a single biome either, confined as it may be to a single or small number of biomes.

Table 4.3 shows the pairwise correlations for the threat factors of each biome in the Iarras Aithneach area when the calculation is carried out using the risk and return values recorded during the stakeholder analysis. Only 2 biomes display a statistically significant pairwise correlation with each other (at the .01 threshold), namely, agricultural land and coniferous forest. At a threshold level of .05, the coastal lagoon and waterbodies biomes can also be deemed as associated. This means that the biome portfolio, according to the risk and returns

[13] The Pearson R correlation indicates the magnitude and direction of the association between two variables that are on an interval or ratio scale.

values given by local stakeholders has a low, but not negative, PIS value. This indicates that from a coastal management perspective, the return of the biome portfolio is resilient to development of most of the major biomes, without systemic risks affecting other biomes in the portfolio.

Table 4.3: Pairwise correlation (Pearson's r) of the threat factors for each of the biomes based on stakeholder analysis data

	Coastal Water	Coastal lagoons	Water bodies (inc. fresh lochs and rivers)	Sand Beaches	Coastal Islands	Coniferous forest	Agricultural land	Peat bogs
Coastal Water	1							
Coastal lagoons	-0.262	1						
Water bodies (inc. fresh lochs and rivers)	-0.178	0.677*	1					
Sand Beaches	-0.178	-0.123	-0.083	1				
Coastal Islands	0.094	0.135	0.330	0.330	1			
Coniferous forest	-0.178	-0.123	-0.083	-0.083	-0.147	1		
Agricultural land	0.069	-0.160	-0.108	-0.108	-0.190	0.946**	1	
Peat bogs	0.011	-0.196	-0.133	-0.133	-0.234	0.213	0.276	1

** Highly significant at 0.01 threshold

* Significant at the .05 threshold

The analysis now turns to the scientific consultations and the resulting data and management connotations. As previously mentioned a group of marine scientists were also presented with the same scale risk return tables as the local stakeholder group. These individuals were based at a university operated shell and fin fish research laboratory (aquaculture) in the study area. As such they also had an in-depth knowledge of the marine and coastal biomes in the area through their research work. Table 4.4(a) and 4.4(b) show the values attributed to biome returns during this scientific consultation. For every biome the overall value given to total return exceeded that of the stakeholder analysis. The greatest difference was in the coastal lagoon biome, receiving a value of 18 from scientific consultations and 4 through the stakeholder analysis[14]. The least differently valued biomes were sand beaches (22:16), coastal waters (20:17) and peat bogs (19:12).

[14] While this paper examines differences in the assessment of ecosystem services and values across different stakeholder groups it has previously been demonstrated in the literature that cultural differences both within and between different societies can have a significant influence in terms of attitudes and willingness to pay for environmental goods and services (Hynes et al, 2008).

Table 4.4a: Estimated ecosystem service values of biomes based on scientific consultation

Service	Biome Type							
	Coastal Waters	Coastal lagoons	Water bodies	Sand Beaches	Coastal Islands	Coniferous forest	Agricultural land	Peat bogs
Agriculture	0	0	3	0	3	3	3	2
Fishing	3	2	3	0	3	0	0	0
Aquaculture	3	0	3	1	3	1	0	0
Intertidal Gathering	1	1	0	3	3	0	0	0
Sand/gravel/rock/peat extraction	3	1	0	3	1	0	3	3
Conservation interest	3	3	3	3	3	1	2	3
Recreation and tourism	2	2	2	3	3	3	1	1
Cultural/educational	3	2	3	2	3	2	2	2
Flood protection /coastal defence	1	3	3	3	2	1	2	3
Nutrient/waste absorption	3	1	2	1	1	2	2	2
Re. Energy generation	2	0	2	0	2	1	2	0
Landtake (carparks/range/causeways)	0	3	0	3	1	1	3	3
Total Service	**24**	**18**	**24**	**22**	**28**	**15**	**20**	**19**
Area of each biome (ha)	**10490**	**31**	**1306**	**37**	**2044**	**1171**	**3293**	**13380**

Table 4.4b: Product of biome spatial areas and estimated ecosystem service values based on scientific consultation

Service	Biomes									
	Coastal Waters	Coastal lagoons	Water bodies	Sand Beaches	Coastal Islands	Coniferous forest	Agricultural land	Peat bogs	Total value for each service	Norm. Value for each service
Agriculture	0	0	3918	0	6132	3513	9879	26760	**50202**	**7%**
Fishing	31470	62	3918	0	6132	0	0	0	**41582**	**6%**
Aquaculture	31470	0	3918	37	6132	1171	0	0	**42728**	**6%**
Intertidal Gathering	10490	31	0	111	6132	0	0	0	**16764**	**2%**
Sand/gravel/rock/peat extraction	31470	31	0	111	2044	0	9879	40140	**83675**	**12%**
Conservation interest	31470	93	3918	111	6132	1171	6586	40140	**89621**	**13%**
Recreation and tourism	20980	62	2612	111	6132	3513	3293	13380	**50083**	**7%**
Cultural/educational	31470	62	3918	74	6132	2342	6586	26760	**77344**	**11%**
Flood protection /coastal defence	10490	93	3918	111	4088	1171	6586	40140	**66597**	**10%**
Nutrient/waste absorption	31470	31	2612	37	2044	2342	6586	26760	**71882**	**11%**
Re. Energy generation	20980	0	2612	0	4088	1171	6586	0	**35437**	**5%**
Landtake (carparks/range/causeways)	0	93	0	111	2044	1171	9879	40140	**53438**	**8%**
Total Service Value S for each biome	**251760**	**558**	**31344**	**814**	**57232**	**17565**	**65860**	**254220**	**688486**	**100%**
Normalised value for each biome (% scale)	**37%**	**0%**	**5%**	**0%**	**8%**	**3%**	**10%**	**37%**	**100%**	

The reasoning behind similar valuations of these biomes is not as close to consensus as it appears however. Scientists and local stakeholders may be relating the value of returns from these biomes to different ecosystem services. The coastal water biome has a significant status in Iarras Aithneach; it has provided substantial economic opportunity in the area through fisheries, aquaculture, intertidal gathering and tourism and as a result of these naturally has a strong cultural and historical significance to the community. However, it received zero return values for services like nutrient/waste absorption and renewable energy generation. Contrastingly, these services received positive values during the scientific consultations and services like "conservation interest" received maximum return value (3). This situation is true also for the peat bog biome. While the biome scored highly in the stakeholder analysis for services such as peat extraction, conservation received a low scale value (1). Contrastingly, conservation was given a value of 3 during the scientific consultations. The picture of similarly rated biome returns can therefore be misleading[15].

[15] This is a positive feature of the BPA framework. BPA allows not only for a valuation of the biome returns, but even in cases where values seem to converge, it allows the analyst to observe where attitudes differ about *where* that return is coming from. This example shows the importance of properly reflecting on the results in order to avoid misinterpretation.

Table 4.5a: Estimated risk value to ecosystem biomes based on scientific consultation

Risk/Threat	Biome Type Sea and ocean	Coastal lagoons	Water bodies	Sand Beaches	Coastal Islands	Coniferous forest	Agricultural land	Peat bogs
Climate Change	3	2	3	3	3	2	3	3
Erosion	1	3	3	3	3	3	3	3
Flooding (inc. Sea level rise)	3	3	3	3	3	1	3	2
Saline intrusion	0	1	1	0	2	2	3	3
Tourism and recreation impact	2	2	3	3	3	2	2	2
New causeways and other infrastructure	1	3	1	2	2	2	3	3
Agricultural change	3	3	3	2	3	2	3	3
Pollution inc. oil spills	3	3	3	3	3	3	3	3
Invasive species	1	3	3	2	2	0	0	0
Marine and terrestrial litter/dumping	2	3	2	3	2	1	1	2
Overgathering of shellfish/overfishing	3	2	3	3	3	0	0	0
Other 1 (Over-regulation)	1	1	1	1	1	0	1	1
Other 2 (Salt damage)	2	2	1	1	1	0	2	2
Total Risk	**25**	**31**	**30**	**29**	**31**	**18**	**27**	**27**
Area of each biome (ha)	**10490**	**31**	**1306**	**37**	**2044**	**1171**	**3293**	**13380**

Table 4.5b: Product of biome spatial area and estimated risk value to each biome based on scientific consultation

Risk/Threat	Sea and ocean	Coastal lagoons	Water bodies	Sand Beaches	Coastal Islands	Coniferous forest	Agricultural land	Peat bogs	Total value for each risk	Norm. Value for each risk
Climate Change	31470	62	3918	111	6132	2342	9879	40140	**94054**	**11%**
Erosion	10490	93	3918	111	6132	3513	9879	40140	**74276**	**9%**
Flooding (inc. Sea level rise)	31470	93	3918	111	6132	1171	9879	26760	**79534**	**9%**
Saline intrusion	0	31	1306	0	4088	2342	9879	40140	**57786**	**7%**
Tourism and recreation impact	20980	62	3918	111	6132	2342	6586	26760	**66891**	**8%**
New causeways and other infrastructure	10490	93	1306	74	4088	2342	9879	40140	**68412**	**8%**
Agricultural change	31470	93	3918	74	6132	2342	9879	40140	**94048**	**11%**
Pollution inc. oil spills	31470	93	3918	111	6132	3513	9879	40140	**95256**	**11%**
Invasive species	10490	93	3918	74	4088	0	0	0	**18663**	**2%**
Marine and terrestrial litter/dumping	20980	93	2612	111	4088	1171	3293	26760	**59108**	**7%**
Overgathering of shellfish/overfishing	31470	62	3918	111	6132	0	0	0	**41693**	**5%**
Other 1 (Over-regulation)	10490	31	1306	37	2044	0	3293	13380	**30581**	**4%**
Other 2 (Salt damage)	20980	62	1306	37	2044	0	6586	26760	**57775**	**7%**
Total Threat Value R for each biome	**262250**	**961**	**39180**	**1073**	**63364**	**21078**	**88911**	**361260**	**974833**	**100%**
Normalised risk for each biome (% scale)	**31%**	**0%**	**5%**	**0%**	**8%**	**3%**	**11%**	**43%**	**100%**	

Table 4.5(a) and 4.5(b) show the values attributed to biome risks during the scientific consultations. In almost all cases the values are far higher than in the stakeholder analysis. Fig. 4.4 depicts the risk return relationship for each biome in Iarras Aithneach based on the scientific consultations and also those from the stakeholder analysis to allow for comparison. The difference between the strategic and local survey results appears less pronounced when spatial area is included in the depiction of the risk return profile of the biomes. The contrast is starker when comparing total values alone, before spatial area is included in the calculations.

Figure 4.4: Normalised and risk-return profiles of all portfolio biomes for both local stakeholder analysis and scientific consultations (biome service and risk values * biome area)

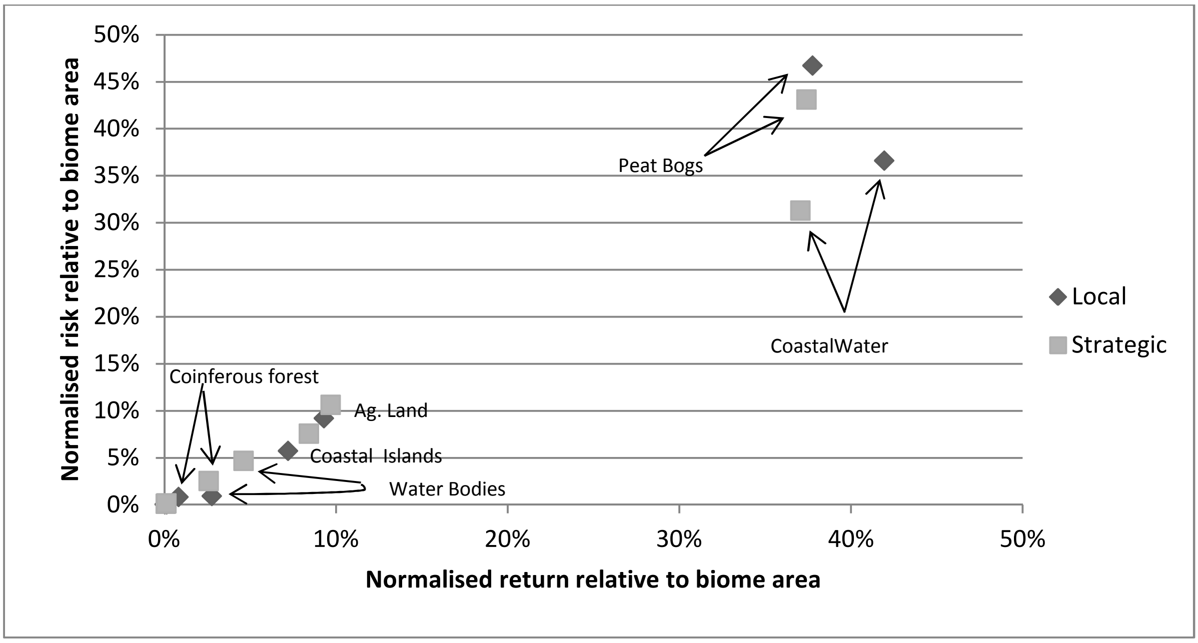
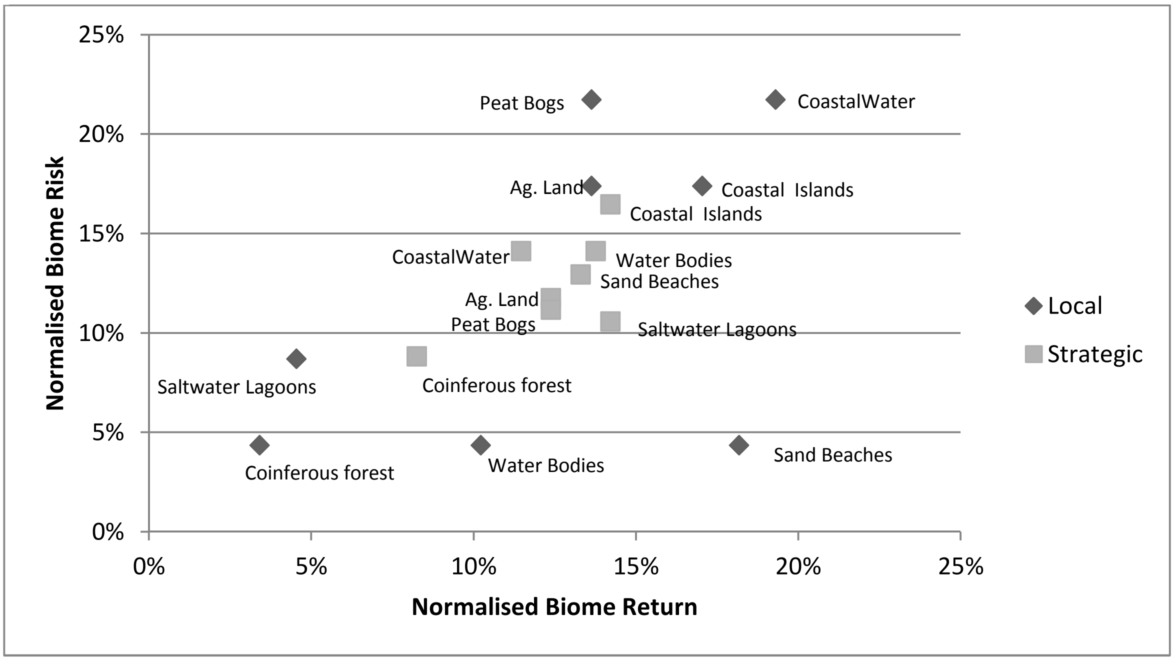


Figure 4.5: Normalised risk-return profiles of all portfolio biomes for both local stakeholder analysis and scientific consultations (spatial area of biome not included in calculation)



Because the true contrast in values given from both viewpoints is concealed when spatial area plays such a large role in risk-return calculations, the contrast in risk return profiles *before* spatial area is examined in Fig. 4.5. This is a much more realistic depiction of the biomes which local stakeholders valued highly for both risks and returns. Biome portfolio return in a local stakeholder context is largely made up of returns from peat bog, coastal water, agricultural land and coastal islands. In contrast, the biome portfolio under the scientific context exhibits more evenly proportioned sources of biome risk and return[16].

[16] This is because the scientific consultations lead to upper *and therefore similar* valuations of each biome's services, despite the fact that the scientific valuations were higher than local valuations in every case. This seems justified since a scientific understanding precedes a greater awareness of biome functions and ecosystem services, and correspondingly higher values. For further comparison of the risk-return results of the survey, see Fig. 4.5.

Table 4.6: Pairwise correlation (Pearson's r) of the threat factors for each of the biomes based on scientific consultation data

	Coastal Water	Coastal lagoons	Water bodies (inc. fresh lochs and rivers)	Sand Beaches	Coastal Islands	Coniferous forest	Agricultural land	Peat bogs
Coastal Water	1.000							
Coastal lagoons	0.354	1.000						
Water bodies (inc. fresh lochs and rivers)	0.619*	0.511*	1.000					
Sand Beaches	0.653*	0.626*	0.788**	1.000				
Coastal Islands	0.563	0.435	0.855**	0.733**	1.000			
Coniferous forest	0.028	0.298	0.272	0.282	0.588*	1.000		
Agricultural land	0.072**	0.148	-0.023	-0.016	0.330	0.789**	1.000	
Peat bogs	0.006	0.157	-0.103	-0.017	0.254	0.841**	0.939**	1.000

** Highly significant at 0.01 threshold

* Significant at the .05 threshold

Table 4.6 shows the pairwise correlations between biomes when using the values attained from the scientific consultation. There are a total of 7 biomes which have a statistically significant positive pairwise correlation at the .01 threshold level of statistical significance. At the .05 threshold level of statistical significance, there is a total of 11. Clearly, the scientific consultations result in a far higher PIS value for the biome portfolio. The implication of this from a coastal management perspective, is that a management decision affecting one biome, which may appear to pose no threat to other biomes in the area (based on the values attained through the stakeholder analysis), could affect the return from other biomes according to more scientifically informed points of view and analyses.

4.6. Discussion and Conclusion

One of the major weaknesses of the European ICZM initiative described by McKenna et al. (2008) was that the strategic principles, which require management to take a wide view of spatial and temporal factors, are incompatible with local principles which focus on the "specific needs of specific people in specific places". This incoherency, arising out of two conflicting objectives, will affect any coastal management initiative so long as the issue is not resolved at the policy level. Indeed, BPA generally requires that a panel of various stakeholders and scientific experts, representing diverse interest groups, reach consensus about the value of each ecosystem service in a biome, as well as the scale of any threats to that biome's function. Yet Billé (2008) suggests that the idea that all conflicts can be resolved with a consensus agreement is a simplistic belief and that "such misconceptions are partly responsible for the inability of numerous participatory processes to adequately take charge of the environmental problems that justified their inception".

This raises a question: If Billé is right, is BPA not then redundant? From this analysis, it appears not. While the Hills et al. (2009) vision of BPA was that of a tool which could arrive at consensus values across a diverse groups, this analysis supports a variation of their theme. Rather than grouping scientists with local stakeholders and struggling to attain some form of consensus, the methodology could be used to evaluate the perspectives of both groups separately, after which, the data can be used to draw distinctions and understand where attitude gaps and similarities lie. This would appear to be a far more arming process for coastal managers, for as the literature clearly indicates, the real challenge for the future of European coastal management will be balancing the various objectives of multiple interests. To do this, coastal managers need to understand what those interest groups are and how their attitudes toward biome use and management compare.

Despite the proposition that defining an "attitude gap" is a more useful function for BPA, there is still scope for it to assist in deciding between alternative management decisions on a qualitative basis when sufficient quantitative data to do so does not exist. This scope is limited however by the incoherent nature of the ICZM principles at present and until such incoherency is resolved, any management format based on participation and consensus is undermined. This weakness is a feature of ICZM policy as opposed to management frameworks like BPA, which are framed by the policy context they are applied to. A further point is worth noting; the entire process of categorizing a management area by biome types, risks and returns plays the role of informing policy makers about the diversity of environmentally important spaces an area possesses (as well as the derivative ecosystem services and environmental and man-made risks to those services).

Similar biome valuations from both the scientific and local survey participants acquired value from contradicting services, for example, conservation and peat extraction. This meant that at times, contradicting views on the value of some biomes, and in turn how they should be managed, showed up as convergence between the strategic and local consultation groups. It is recommended that BPA always be carried out by defining categorical groups such as local, strategic, stakeholder, relevant interest group etc, so that the risk-return values of each group for each biome can be compared and assessed, and the sources of biome value can be identified.

A further caveat of the BPA methodology is the role of the size of a biome in determining its contribution to the biome portfolio. This is acceptable in the case of an ecosystem service which can only be provided in the context of a large spatial area, such as carbon sequestration, but less suitable when considering ecosystem services that are threatened by habitat damage such as biodiversity and existence value. Further reduction in the size of such biomes indicates that they contribute less to society, but of course, this is precisely when the value of biodiversity within such a biome must be acknowledged as increasing in value, since the service in question is now more scarce. Future improvements to the BPA methodology should attempt to account for this fact.

Finally, the PIS of the biome portfolio was highly dependent on which study group the data was based upon; the scientific group providing data which led to a far higher PIS value. While BPA can only highlight where the two domains differ, the incoherence in local and strategic objectives in EU policy is a problem that coastal managers face. Because there has been no prioritisation of either set of principles, there is no guidance or legislation off of which coastal managers can base their decisions.

During the stakeholder analysis the negative attitude of stakeholders towards the perceived risk of over-regulation became evident. It is likely that this attitude contributed to the low values assigned by the stakeholder analysis participants for risks across all biomes. In recent decades, regulation, especially with respect to coastal livelihoods like fishing, aquaculture, and shellfish harvesting have reduced the capacity of local stakeholders to harvest, profit or gain employment from such local economic activity. Although it can be also said that this impact of legislation is the result of a lack of regulation historically, the perception among stakeholders is that it is restrictive. Recently, prohibitive regulation on peat extraction in peat bogs has also been brought into legislation. Over-regulation is a very real concern for local inhabitants of coastal areas of environmental importance and tackling the development of such negative perceptions is an important part of an integrated approach to coastal management. A vital part of tackling such negative perceptions must be the identification of areas where the negative impact of regulation on regional economic development can be mitigated and sustainable enterprise and development is promoted at the policy level.

It is clear that the nature of the irreconcilable differences in EU ICZM objectives will require some controversial decisions to be made regarding prioritisation of principles. With respect to BPA, one possibility for consideration is that coastal manager's would base decisions on data from local participatory stakeholder groups *and* data from scientific consultations. Because locally based data will be likely to have lower risk values and therefore a lower PIS value, any management decisions based on achieving local objectives would first have to be analysed through the scientific consultation data. Where no predetermined "red lights" with respect to the scientifically based PIS values were set off, local development orientated decisions could be proceeded with. Further case studies demonstrating this type of analysis are needed.

There is a wider debate taking place on this topic in relation to the justification of basing environmental management decisions on the value assessments of consumer preferences when many individuals do not understand the various environmental and ecosystem processes which provide the services society consumes. If non-market values are to be used within cost-benefit analysis (CBA) to inform public policy choice and the management of environmental assets, then the main tenet of welfare economics on which CBA is based – namely, the primacy of consumer preferences – creates problems for many when these preferences are based on very incomplete understanding of how ecosystems work, of the importance of ecosystem services to well-being, and of the importance of different aspects of biodiversity. As Atkinson and Mourato (2008) point out, 'to the extent that groups or individuals are poorly informed about the environment, there are too many risks to allowing uninformed views to hold sway over decisions'.

In this study, the "attitude gap" between scientific and local views was pronounced. Of the various reasons why this may be so, a disparity in the level of knowledge of participants about ecosystem processes, benefits and biodiversity in general, is likely to account for much of this. The assessments indicate that BPA may be a useful format for helping environmental managers and policy makers understand where local views stray from scientific views about ecosystem services and risks and how the coastline should be managed. Such a procedure could be an ideal "first step" in any coastal management initiative. Clearly, there are two states of thought regarding environmental decision making; one favouring consumer preferences (local), the other preferring reliance on expert opinion for strategic development. Be it for the purposes of attempting to make optimal trade-offs between coastal development, conservation, risk and return, or simply to categorise the differences in outlook between local and strategic views, there is scope for development and application of BPA. In situations where strategic and local objectives are closely aligned, BPA is especially suitable for application. There is also scope in future applications of the BPA framework to apply non-market values from the literature to biomes as a measure of return (instead of using Likert scale values) in a benefit type transfer type exercise.

5. The impact of quota changes on the discrete fishing site choice of vessels in Irish demersal otter trawl fisheries

5.1. Introduction

Today's fishery policy is largely based on conventions developed in the field of fisheries economics from the 1950s onwards (e.g. Gordon, 1954; Clark and Munro, 1975; Munro 1979) and has historically been predominantly concerned with attempting to determine the "optimal" harvest rate of a fishery using a bioeconomic modelling approach. This format implicitly assumes policy makers and fishery managers can successfully control fishery inputs and/or outputs to achieve desired targets. Often however this is not the case and repeated failures to constrain fishery inputs and outputs have led to major fisheries crises internationally (Bockstael and Opulach, 1983).

Recently however, more spatially and behaviourally concerned fisheries economic research has emerged, specifically, the use of discrete choice econometrics to investigate fishermen's choices. In this vein of research, economists investigate the influence of economically relevant factors and the impact of policy changes on fishermen's choice(s) across a finite number of discrete options which fishermen have available to them. Papers in this area have analysed the choice of fishery (Bockstael and Opaluch, 1983), the entry-exit decision within a fishery (Ward and Sutinen, 1994), fishing location choice (Eales and Wilen, 1986; Dupont, 1993; Larson et al., 1999; Curtis and Hicks, 2000; Mistiaen and Strand, 2000; Smith, 2005), repeated participation decisions (Smith and Wilen, 2005; Smith et al., 2008) and choice of gear (Eggert and Tevertas, 2004). Berman et al. (1997), Smith, (2002) and Smith and Zhang (2007) focus on multiple discrete decisions, combining fishing location choice with either the participation decision or target species decision.

The new round of CFP reforms will affect technical measures, quotas, available fishing grounds, fleet capacity and see the advent of new supports for small scale fisheries[17] as well as a host of other changes. All of these changes will have financial and operational consequences for fishermen, thereby influencing the fishing-related decisions that they make and in turn the ecological ramifications and effectiveness of these new policies. It makes sense then that research in this area should focus on the potential impacts which changes to the CFP may have on fishermen's behaviour. By improving the understanding of the factors that determine fishermen's decisions, we can move towards a situation where fishery managers can better predict the actual outcomes of conservational measures.

In this paper, I seek to understand the potential impact of the CFP reform on the decision making behaviour of fishermen in Irish waters. Specifically, I investigate how the now imminent changes to EU quota levels could affect the fishing location choice of Irish vessels in bottom trawl fisheries based in Irish waters.

This is a potentially useful line of inquiry for a number of reasons. Firstly, it is important for fisheries managers to be able to speculate on the potential behavioural responses of fishermen to policy changes, since these responses can themselves have an ecological impact (Daw et al. 2012). Beyond fisheries management alone, CFP reform and the Commission's green paper focus on the importance of spatial management and coherence with the Marine Strategy Framework Directive, maritime spatial planning, and the Habitats and Birds Directives. The CFP and the aforementioned spatially and environmentally orientated directives prioritise good ecological status including the protection of biodiversity and ecosystem functioning.

[17] Small scale usually refers to vessels less than 12 metres in length not using towed gear. Definitions of what this support should actually involve are sparse, but the general allusion is toward financial support and more lenient quota regulation.

The topic of fishermen's targeting behaviour is of particular relevance to the problem of by-catch. In mixed fisheries, where species selectivity of fishing gear is limited, the main determinant of the species harvest mix will be the choice of fishing location. This means that policy alterations that have a major impact on the spatial decision making of fishermen, will automatically have connotations for the species harvest mix of the fishery, and therefore the level of by-catch. The ability to inform fishery managers about the likely direction of reallocated fishing effort, once displaced, can assist them in determining where increased by-catch in certain areas may be a problem, and to anticipate such a negative ecological outcome using pre-emptive policy instruments. The major barrier to such anticipation however is the lack of by-catch estimates that exist. For example, the Marine Work Group Ireland (MWGI) report for most fisheries around Ireland, a lack of monitoring and assessment means it is generally not possible to determine annual cetacean by-catch mortality rates. OSPAR (2000) reports that "improved estimates of population sizes and knowledge of stock identity and migration are required to enable more accurate assessments of the impact of by-catches on cetacean populations". This situation applies to the majority of fish stocks in EU waters. However it is very important to highlight that given such estimates of various by-catch rates for various species across time and space, the usefulness of spatial discrete choice models would be greatly magnified, especially in mixed fisheries where selective gear and targeting of species is limited.

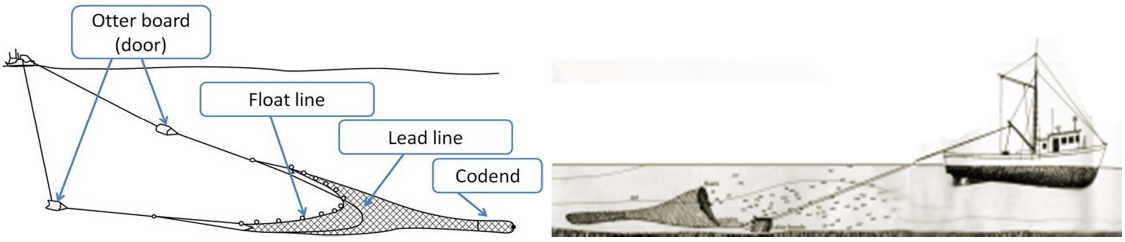
Section 5.2 below provides a discussion of the Irish otter trawl fleet, fishery and management policy. Section 5.3 reviews previous literature applying discrete choice analysis to fisheries related problems. Section 5.4 discusses the Vessel monitoring systems (VMS) data used in the analysis.. It also discusses how fishing location alternatives were defined. The theoretical model is described in section 5.5 and the practical method by which it is applied is explained in section 5.6. In section 5.7 the results are presented as well as an analysis of the findings. Finally, section 5.8 includes a discussion and the conclusion of the chapter.

5.2. Background

The Irish otter trawl fleet comprises 275 trawlers, both demersal and pelagic, and operates in a complex multi-species multi-gear fishery over a large geographic area around the Irish coast. Various gear types are utilised amongst the fleet to land over 100 species from various species assemblages which in 2006 amounted to 210,000 tonnes live weight, worth approximately €250 million at first sale. This made up 75% of annual Irish landings in terms of economic value (Davie and Lordan, 2011). Clearly then, this is a very economically significant component of the Irish fishing fleet. Because of the multiple species targeted within this fishery and the heterogeneity across vessels within the fleet, analysing fleet dynamics as a single unit can lead to erroneous conclusions. Davie and Lordan (2011) combine factorial analysis with multivariate clustering analysis (see Pelletier & Ferraris, 2000; Holley & Marchal, 2004; Ulrich & Andersen, 2004; Campos et al., 2007) to segment the entire Irish otter trawl fleet into a homogeneous subdivisions or métiers. These metiers are defined according to species assemblage, vessel characteristics, fishing gear, mesh size, vessel length, fishing grounds and fishing season.

The results of Davie and Lordan (2011) indicated that métiers exploiting demersal species are characterised by single-vessel bottom otter trawls, usually less than 24 metres in length and with mesh sizes of 70mm or more. Trawling requires a means of holding the mouth of the net open while towing, and a system of wires to connect the net and gear to the vessel (Galbraith and Strange, 2004). Vessels are fitted with winches on deck to move and store the trawling wires or warps and otter boards' are used to spread these connecting wires and hold the net open, horizontally (see figure 1). Large spherical floats, built to withstand implosion at extreme fishing depths are attached to the upper edge of the net mouth (floatline) and these provide vertical lift to the net while weight is placed on the edge of the net in contact with the seabed (footrope). The net itself is usually funnel shaped with extended sides that form wings for guiding fish into the net. Bottom trawl nets sometimes have a top canopy to prevent fish from escaping over the top of the net.

Figure 5.1: Otter Trawl Rig System



Demersal otter trawl landings are found to account for just 13% of total weight landed by the fleet, a much larger proportion of otter trawl caught species being landed by the pelagic segment. Nevertheless, demersal otter trawls account for the vast majority of commercially harvested demersal species which tend to be of higher value commercially (Gillespie and Hynes, 2011). In Davie and Lordan (2011), demersal métiers tended to exhibit higher diversity of species in catch compositions, which attests to the high level of mixed fishery interactions that take place in the demersal otter trawl fisheries. This makes demersal otter trawls far more susceptible to by-catch and chance-catch issues and highlights the importance of choke species[18] in demersal trawl fisheries. In the case of stock rebuilding efforts, Cod

18 "choke species" is a term used to describe a low quota species, which, if reached, would lead to vessels having to tie up even if they still had quota for other species

being the primary example, the impact of both targeting and non-targeting segments of the fleet are relevant since the species selectivity of fishing gear is limited.

As discussed in Chapter 2, Irish fishing waters have been governed as part of the European Union's Common Fisheries Policy (CFP) since 1983. The CFP is a collaborative means of governance across EU member states and tries to ensure sustainable fishing practice by setting Total Allowable Catch limits (TAC) limiting the number of days at sea (fishing effort), restricting the use of certain fishing gear (Technical Conservation Measures (TCM)) and reducing overcapacity in the EU fishing fleet (through fleet decommissioning)

While management of the Irish pelagic fleet entails diverse management measures, such as determining open and closed seasons and the establishment of "no-fishing" zones, management of the Irish demersal fleet is almost exclusively quota based (OECD, 2013). With conservation and management of Irish demersal fisheries largely taking the form of annual quotas imposed on all principal commercially targeted species, the objective of fisheries management is to regulate and maximise the catching, sale and processing of fish within set limits. In addition to these objectives, conservation is a key concern of the reformed CFP. To determine the status of various commercially targeted fish stocks in EU waters, the European Commission consults with the Scientific, Technical and Economic Committee for Fisheries (STECF) and the International Council for Exploration of the Sea (ICES) (EC, 2014). In the case of Irish demersal fisheries, which are largely regulated using annual species TACs and vessel specific quotas, scientific advice informs prospective changes to quota allocations for species which have been identified as overfished. The proposals of various scientific institutes inform the EU Council of Ministers, made up of national ministers from member states, which have the final authority to negotiate and formulate fishery regulations and species TACs.

Given the annually allocated national quotas, the Irish department of Communications, Marine and Natural Resources consults with industry stakeholders to determine catch limitations per vessel within a given fishery for each month. In particular key white fish species in Ireland are controlled by means of separate Ministerial Orders which restrict the fisheries as necessary, by setting catch limits per boat, according to the size of the vessel and based on recommendations of the committee. The monthly quota allocations also take into account the uptake of the available quota throughout the year to maintain access on an equitable basis amongst the fleet. Key stocks for which monthly catch limits per boat are set include Cod, Haddock, Whiting, Hake, Monkfish, Megrim, Nephrops, Sole and Plaice as well as other deep sea species[19]. Vessel adherence to the monthly quota allocations is enforced by Sea Fisheries Protection Authorities officers on land and the Naval Service at sea.

5.3. Literature Review

Smith (2010) asserts, “a purely empirical approach to historical patterns of fishing behaviour over time and space will mislead analysts about how fishing fleets will react to policy changes… a fisheries analyst must therefore employ structural models in order to predict outcomes”. Yet historically, fisheries economics research has primarily focused on bioeconomic modelling approaches where determining an “optimal” fishery harvest level is the priority and long term investment decisions are the focus. Eales and Wilen (1986) posit this to be understandable since “earliest works in the field were aimed at demonstrating how open access attracts less capital. Subsequent work refined the comparison of open-access equilibria with optimal equilibria”.

[19] Black scabbardfish, greater silver smelt, tusk, roundnose grenadier, orange roughy, ling species and red seabream

Bockstael and Opaluch (1983) took the view that while bioeconomic literature was particularly useful in characterising the nature of optimal solutions, a lack of control over fishing effort (to ensure harvest rates are at levels deemed optimal by bioeconomic models) prevented it from being presently operational for fisheries management. Bockstael and Opaluch (1983) and Eales and Wilen (1986) were amongst the first to use discrete choice micro econometric techniques to analyse the qualitative decisions of fishermen and used fairly straightforward multinomial methods to construct their analyses. Specifically, Bockstael and Opaluch (1983) investigated fishermen's fishery choices given a total of 13 discrete fishery options in New England waters, from 1975 to 1976. Particular attention was paid to what the authors described as the "economic and noneconomic inertia" that prevented instantaneous movement between fisheries on the basis of highest available profits. The influence of uncertainty on the fishery decision was considered by incorporating expectations about means and variances of returns into the analysis. Finally, fishermen's decisions were partially determined by their initial wealth, which was categorised as the value of their fishing vessel. The logit model produced a matrix of what the authors termed "predicted transition probabilities" which indicated the probability of a vessel moving from the initially chosen fishery, into each alternative subsequent fishery. The results of the analysis corresponded with economic theory; fishermen responded positively to increased returns and negatively to increased variability of returns, but exhibited a bias toward remaining in the same fishery over time, thus behavioural responses were slow.

Eales and Wilen (1986) incorporated a "nested" format into their multinomial logit model to assess the fishing location choices of 17 vessels in the pink shrimp fishery off the northern California coast. Eight discrete fishing locations that fishermen could choose to fish were defined and logbooks were used to relate each "set of the net" to the quantity of shrimp caught. The determinants of fishermen's location choice decisions, that is, the regressands on site-choice that were used in the multinomial equations, were expected catch and distance from home port. Nesting the 8 distinct fishing locations into three larger subgroups led to a marginal improvement in the accuracy of the model's results, as compared to a standard multinomial logit where the 8 discrete location alternatives were decided between simultaneously.

Following the work of Bockstael and Opaluch (1983), Dupont (1993) uses a random utility model (RUM) to assess the impact of profit uncertainty on the location choices of fishermen in the British Columbia salmon fishery. Fishermen were assumed to form expectations about future prices though an ARIMA type process[20]. These price forecasts were then used as instrumental variables in the estimation of seasonally variable expected profit functions for each fishing site. Two RUMs were run; in the first, only expected values of profit and variance are permitted to influence/predict fishermen's fishing site choice. In the second, expected wealth and its variability, inclusive of future expected profitability, influence/determine site choice. Dupont finds that expected wealth, profits and the variability of each contribute to the fishing location decision. Ward and Sutinen (1994) use cross-sectional data from 1965 to 1975 and a multinomial logit model to analyse the entry-exit decision of fishermen in the Gulf of Mexico shrimp fishery. The potential determinants/influents of the decision included in the analysis include price and abundance of shrimp in the fishery, changes to fleet size and vessel specific characteristics such as size and mobility. The analysis identified a strongly positive relationship between fleet size and the decision to exit the fishery, but despite an increasing number of vessels in the fishery over

[20] In statistics and econometrics, and in particular in time series analysis, an autoregressive integrated moving average (ARIMA) model is a generalization of an autoregressive moving average (ARMA) model. These models are fitted to time series data either to better understand the data or to predict future points in the series (forecasting). They are applied in some cases where data show evidence of non-stationarity, where an initial differencing step (corresponding to the "integrated" part of the model) can be applied to remove the non-stationarity.

the sample period, the impact of the "crowding externality" was reduced due to an improvement in shrimp landings. The results also supported the "resistance to change" findings of Bockstael and Opaluch (1983).

Larson, Sutton and Terry (1999) model the weekly fishery choice in the trawl fishery of the Bering Sea/Aleutian Islands using a sample of weekly data from 1991. They use a RUM to measure the impact of risk and individual vessel performance on the selection of fishing site across 12 discrete alternatives. The analysis also estimates the impact of the distribution of vessels within fisheries on weekly fishery choice and seasonal affects. All of the aforementioned variables prove to have qualitative implications for vessels' weekly choice of fishery. Holland and Sutinen (2000) use a nested multinomial logit where fishermen are modelled as first deciding upon fishery and zone choice and then upon an area within this larger zone. The analysis was conducted using data from 484 large trawlers in the New England area from 1990 to 1993 and many of independent variables used in the model were lagged values. The results provided strong evidence of "habit" in fishermen's location decisions, that is, historical fishing patterns played a strong role in determining future ones despite revenue incentives in other regions which might incentivise fishers to deviate more readily from historical choices. Once information about successful catches and revenue generation in alternate locations is disseminated however, Holland and Sutinen (2000) find that vessels will respond by more readily changing fishing location.

Mistiaen and Strand (2000) propose a flexible utility specification that allows for the testing of risk preferences and heterogeneity in fishermen's risk preferences, specifically, the random-parameters logit RPL[21]. What this means is that the initial IIA assumptions of the conditional logit model used by previous studies no longer apply, since the model is generalised so that variable coefficients vary randomly across individuals. Employing the RPL and data from the North Atlantic high migration species (HMS) fishery in 1996, Mistiaen and Strand (2000) assume fishermen select fishing location on the basis of utility expectations associated with each site and found that risk preferences in their sample were highly heterogeneous.

Curtis and Hicks (2000) evaluate the impact of area closures designed to protect sea turtles in the Hawaiian Pelagic longliner fishery. Employing the RUM they estimated the compensation needed for different types of fishermen given the financial impact of various area closure systems. Under full seasonal closures recommended by the National Marine Fisheries Service (NMFS), tuna fishermen would need to be compensated $8,735 per trip but only $4,066 if the fishery was exempt from a seasonal closure. In the longline fishery, compensation would need to run at $20,000 regardless of area closure system used. The estimated cost of reducing longline interactions with sea turtles was $41,262 per turtle.

Smith (2002) models the spatial distribution of fishing trips in the California red sea urchin fishery. The choice of whether to participate in the fishery and which location to fish at are modelled simultaneously. Two approaches are employed by the author. The first is an econometric model where a count data regression model estimates trips per month as a function of open days per month, weather conditions, revenue, variance and economic opportunities outside the fishery. The second is a RUM (nested logit specification) in which the decision to participate (or not) in different spatial locations is a function of weather, revenues, variance and travel cost. Smith (2002) found that both in sample and out, the count data model is more accurate than the RUM in predicting participation. Smith and Wilen

[21] Also known as mixed logit, random-coefficients logit and error components logit.

(2003) predict the economic and biological impacts of a marine reserve in the sea urchin fishery of Northern California by constructing a spatially orientated repeated nested logit (RNL) using logbook/landings data and then integrating this model with a bioeconomic model of the sea urchin fishery. Results of the RNL are economically consistent with previous studies in that divers responded to returns across the discrete locations positively, and negatively to weather risk and travel distance. The results of the bioeconomic model, which is able to treat fishing opportunities as 11 discrete location choices (via the RNL), indicate that the biological impact of marine reserves are often over-estimated by ignoring the displaced fishing effort that is directed to neighbouring locations.

Eggert and Tvertas (2004) model fishermen's gear choice in the Swedish trawler fishery using a 2 stage econometric analysis. Firstly, they estimate stochastic revenue functions with fixed effects for each gear type and use these coefficients to create predictions of revenue (and standard deviation) for all gear, for each trip of each vessel. In the second stage, the revenues and standard deviations estimated in the previous step along with actual gear choices are used in a RPL model that accounts for heterogeneous attitudes towards risk. The results are consistent with earlier studies in that fishermen respond positively to profit and negatively to increased standard deviation. A strong inertia effect was also observed.

Smith and Wilen (2005) analyse fishermen's response to multiple risks in unison in the California sea urchin dive fishery. They use discrete choice models of daily participation given a number of risk trade-offs to analyse fishermen's attitudes to risk. Risks considered include physical risks like shark attacks and fatal accidents and financial risks like price and resource variability. A pooled model assuming equivalence in attitudes to risk is estimated, as well as a model allowing for heterogeneity in fishermen's risk aversion. The results indicate risk aversion in both physical and financial risks as well as significant heterogeneity across individuals. Bootstrapping is used to detect correlation in risk aversion to different factors, physical and financial. While correlation is found, it is not significant and is only significant at the 10% level once seasonal affects are considered.

Smith and Zhang (2007) analyse the impact of marine reserves in the Gulf of Mexico reef fishery by modelling fishing location choice and choice of target species jointly in a RUM. The data set used includes daily federal logbook data on the Gulf of Mexico reef fishery, federal commercial fishing permit data, species level price data from Florida landing tickets and a social survey of reef fish. They draw upon the industrial and public economics literature by using product differentiation and spatial sorting techniques to estimate individual-specific structural coefficients based on observable individual characteristics and choice-specific constants using contraction mapping. Again, the results are consistent with previous studies; higher species prices increase propensity to select a species as a target species and a higher CPUE increases the probability of fishing in a specific location.

Valcic (2008) investigates the motivations behind fishermen's fishing location choices and the displacement of fishing effort that comes about following site closure as a management option. The study focuses on the Oregon bottom trawl groundfish fishery and the formation of a rockfish conservation area. Valcic points out that fishing locations in the fishery do not form geographically distinct clusters and that there exists large numbers of distinct alternatives in fishermen's location choice set. This negated the use of a nested logit, multinomial probit or mixed logit approach. Instead Valcic employs a heteroskedastic extreme value (HEV) model of fishing location choice, which is used to estimate the impacts of a spatial closure, in terms of changes in the probabilities of fishing site choices that arise. Predictive variables used in the model included expected revenue (logarithmic form), distance, individual previous experience frequency and the logarithmic form of revenue variance. Apart from revenue variance, all variables were statistically significantly different from zero and the 95% confidence interval and were economically consistent. Valcic (2008) then simulates an areaclosure in the fishery. The simulation suggests significant changes to fishing effort in certain locations, as much as 10% in some, and provides useful information for policy design.

5.4. Data

5.4.1. Vessel monitoring system and logbook data

The analysis of the fishing location decision at a highly detailed spatial level carried out in this chapter, is made possible via the advent of modern Vessel monitoring system transponders (VMS), which since 2005, all European Union (EU) fishing vessels ≥15m in overall length must be fitted with (EC, 2003; Gerritsen et al. 2012). These transponders transmit a fishing vessel’s position every two hours whilst at sea. By using the distance between a vessel’s sequential location points to calculate travel speeds, analysts can calculate which elements of a vessels time at sea are dedicated to travelling to and from fishing locations and which elements are actually spent fishing Gerritsen and Lordan (2011).

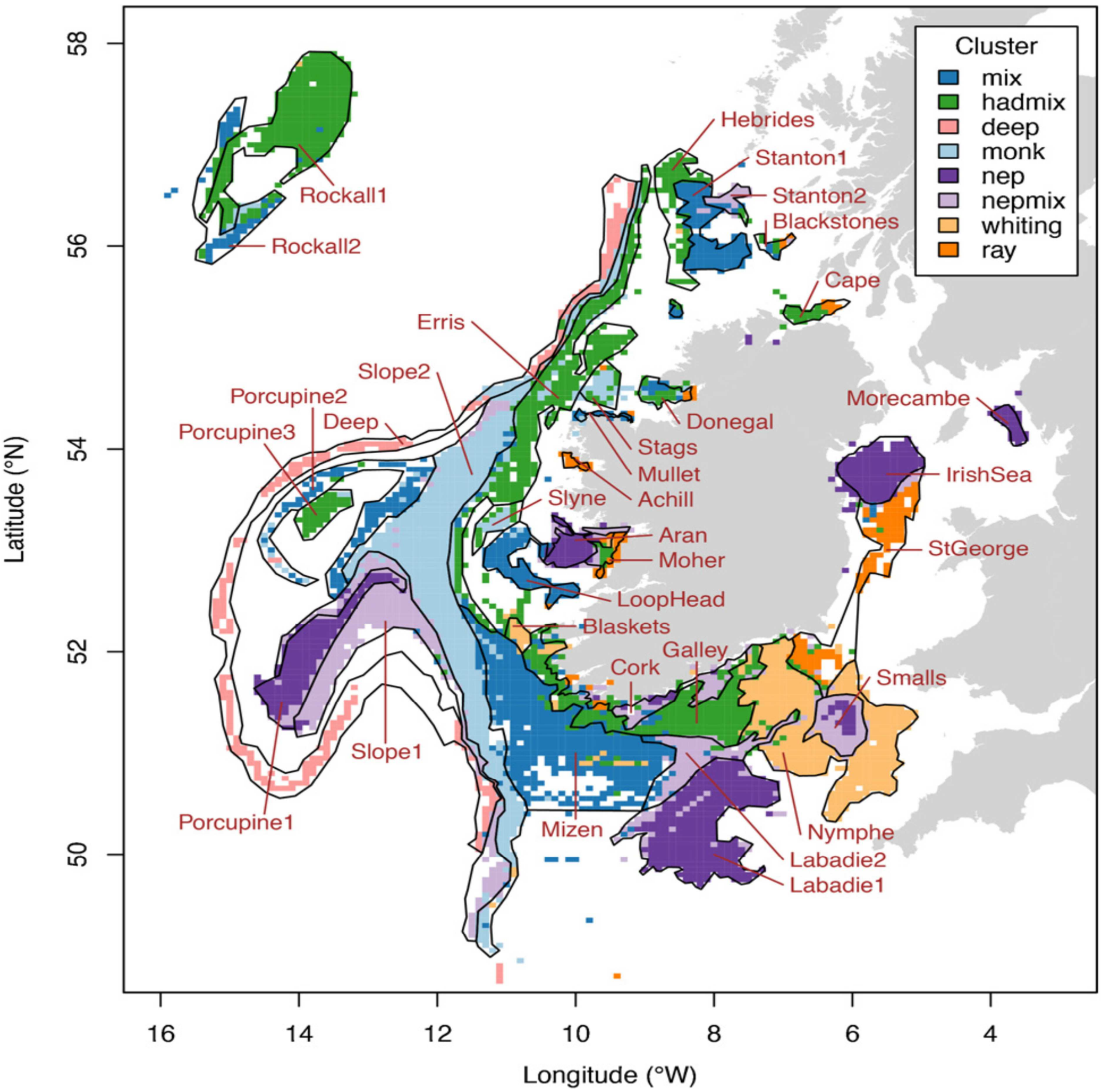
All vessels within the EU ≥10m are also required to record their retained catches on a daily basis in standard logbooks (EEC, 1983). EU logbooks are completed by the masters of fishing vessels upon landing all catch. Once log sheets are filled out they are submitted to the local Port Office. The data is then entered into the Integrated Fisheries Information System (IFIS) database by SFPA staff which is then provided by the Irish Department of Agriculture, Fisheries and Food (Gillespie and Hynes, 2011). In addition, sales notes data is available which is electronically submitted to the IIFS database by the buyer at the first sale of the fish. The sales notes provide data on the price per kg received for landings by a vessel.

By combining positional data of vessels' VMS records with electronic logbook data for each vessel, analysts can create an unprecedentedly detailed representation of the spatial distribution of historical species catches as well as the corresponding fishing location choices made by fishermen. The result is information which permits the analyst to consider the impact of economically-relevant, decision-influencing factors, such as distance to landing port and expected catch values of different species, on the fishing location decision, and also the impact of variation in these factors, such as management initiatives like area closures and quota changes or environmental factors like changes in stock biomass.

5.4.2. Defining distinct fishing location alternatives

Naturally, to analyse 'discrete' fishing location choices, such alternatives need to be defined, and furthermore, the defined alternatives must closely reflect the actual distinctions which fishermen make between fishing locations. Using VMS and logbook data for Irish demersal otter trawl vessels, Gerritsen et al. (2012) use hierarchical cluster analysis to define 8 clusters within the fishery that exhibit similar species compositions. These clusters were then used to define 34 distinct spatial regions in Irish waters. Given the fine spatial scale at which it is possible to observe fishing activity and historical catch rates using VMS data and frequently collated logbook data, the defined fishing location alternatives closely reflect the actual discrete alternatives which fishermen in the Irish demersal otter trawl fishery weigh up when making their fishing location choice decision. Figure 4.1 shows the resulting 34 discrete location alternatives that emerged from the hierarchical cluster analysis. The process used by Gerritsen et al. (2012) to define these 34 fishing location alternatives is discussed in greater detail in section 6.2.

Figure 5.2: Discretely defined demersal otter trawl fishing sites surrounding the Irish coast



From Gerritsen et al. (2012): The spatial distribution of the grid cells assigned to 8 clusters; the colour of each pixel corresponds to the cluster it has been assigned to. The boundaries were drawn manually to define 34 regions. The regions were given names of fishing grounds or geographical features in order to identify them.

5.5. Methodology

5.5.1. Discrete choice models

Discrete choice models provide a useful analytical capacity in situations where individuals choose between a set of mutually exclusive, exhaustive and finite alternatives (Golder, 2014). They are usually developed in a random utility model (RUM) framework in which utility, though unobservable, is assumed to be maximised by the choices that individuals make, each potential alternative having a positive, negative or zero implication for an individual's utility. For individual n, faced with a set of J alternatives, the analyst can specify,

$$j \epsilon U_{nj} \subseteq U, j = 1, \ldots, J.$$

where j is a specific alternative, U_{nj} is the utility derived from the n^{th} individuals choice j, and U is the utility derived from the universal set of alternatives. As there are aspects of individual n's utility which is unobservable, the utility of individual n from choosing alternative $j \epsilon U_{nj}$ is defined,

$$U_{njt} = V_{njt} + \varepsilon_{njt}$$

where V_{njt} is the observable part of U_{njt} at time t and ε_{njt} represents the unobservable component of U_{njt}, i.e., these terms represent the systematic and stochastic components of U_{njt} respectively. The analyst then assumes any individual chooses the alternative with the highest connoted utility, that is, individual n chooses alternative $i \in U_{nj}$ if and only if,

$$U_{nit} > U_{njt} \; \forall j \neq i$$

The joint density of the random vector $\varepsilon_n = \{\varepsilon_n 1, \ldots \varepsilon_n J\}$ is written $f(\varepsilon_n)$. This description of the stochastic component of utility allows one to make probability statements about an individual's prospective choices. Thus the probability that individual n chooses alternative i at time t can be written,

$$P_{nit} = Prob(U_{nit} > U_{njt} \forall j \neq i)$$

$$= Prob(V_{nit} + \varepsilon_{nit} > V_{njt} + \varepsilon_{njt} \forall j \neq i)$$

$$= Prob(\varepsilon_{njt} + \varepsilon_{nit} < V_{nit} + V_{nit} \forall j \neq i).$$

As the probability that each random term $\varepsilon_{njt} - \varepsilon_{nit}$ is below the observed quantity $V_{nit} - V_{njt}$, the distribution can be described as cumulative, and given the density $f(\varepsilon_n)$, it can be written as:

$$P_{ni} = Prob\left(\varepsilon_{njt} - \varepsilon_{nit} < V_{nit} - V_{njt}\right)$$

$$= \int_{\varepsilon} I\left(\varepsilon_{njt} - \varepsilon_{nit} < V_{nit} - V_{njt}\right) f(\varepsilon_{nt}) d\varepsilon_{nt}$$

where $I(.)$ is the indicator function equal to 1 when the expression in parenthesis is true, and 0 otherwise. This representation of the cumulative probability distribution is a multidimensional integral over the density of the unobserved portion of utility, $f(\varepsilon_{nt})$. The specification of the probability distribution can be altered by making different assumptions about the distribution of the unobservable component of utility, which in turn leads to different types of discrete choice models. The specification and underlying assumptions about the distribution are of critical importance, because they influence the means by which the probability that individual n will choose alternative j is calculated, and therefore the accuracy of the analyst's predictions.

Discrete choice models that arise from different assumptions about the probability distribution include for example the logit, multinomial logit, nested logit, mixed logit, and probit. All of these models are suited to different real world applications. With respect to the standard logit, there is a closed form solution, due to the fact that one of the core assumptions of the model is that the unobserved portion of utility is independently and identically distributed (iid). This contrasts starkly with the probit for example, where $f(.)$, is assumed multivariate normal, or the mixed logit, where one assumes the unobserved portion of utility comprises a part that follows any distribution that the analyst wishes and a part that is iid extreme value. This means that for both of these specifications, the integral has no closed form solution and is evaluated numerically through simulation. The best choice of assumptions about the probability distribution, i.e. the best discrete choice model to be used in a particular case, depends upon the physical circumstances of the system being analysed. This is explained in more detail in the next section by taking an in-depth look at some different discrete choice models and the assumptions on which they are based.

5.5.2. Discrete choice model alternatives

Two commonly used discrete choice models which follow the form of the standard logit are the conditional and multinomial logit (MNL). According to Princeton (2014), like the standard logit, these are derived using a closed form solution, i.e. the unobserved portion of utility ε_{njt} is assumed to be independently and identically distributed (iid). This assumption makes the model more convenient to solve and therefore more popular in the literature. For the assumption to be true/satisfied however, more general assumptions we make about the real world scenario under analysis must hold. In particular, it requires that:

1. any unobserved factors contributing to utility are uncorrelated over alternatives and have uniform variance across alternatives;

2. any unobserved factors contributing to utility are uncorrelated across time and have uniform variance across individuals;
3. there are no underlying/unobserved characteristics in decision-makers, that play a role in the potential to attain utility, that vary across individuals.

In the case of the Irish bottom trawl fishery, there are instances in which violation of these assumptions is imaginable. For example assumption 1, also known as the independence of irrelevant alternatives (IIA), suggests that the ratio of two logit probabilities is independent of any other alternative. In other words a decision makers' perception of the attractiveness of one alternative over another will not be influenced by the availability or attractiveness of other alternatives. The problem with this assumption for the Irish bottom trawl fishery, where fishing location choice is likely to be heavily influenced by the travel distance to fishing grounds, is that the attractiveness of one fishing location over another will be partially determined on the basis of the locations' proximity to other fishing grounds, where further fishing may take place. This problem is overcome in the analysis by only considering trips that resulted in one site visit. It is also not as big an issue when considering the impact of quota changes only, as opposed to the impact of removing a site alternative. (i.e. simulating an area closure as is the case in other studies).

Assumption 2 may be violated if fishing locations possess unobservable heterogeneous characteristics, for example, if they exhibit differences in the geophysical make-up of the sea-bed. Likewise, assumption 3 will be violated if vessels or skippers themselves possess unobserved heterogeneous characteristics, such as skipper skill, or unique sets/streams of fishing relevant information.

Such violation of any or all of the three core assumptions of the conditional or multinomial logit models has the potential to compromise the efficiency of the model estimator and create an upward or downward bias on the probability estimates of fishing location choice leading to erroneous predictions. While models that allow for the relaxation of the IIA assumptions (so as to deal with autocorrelation and heteroskedasticity) exist, they are not without their problems. Often these models may not suit the particular economic situation under analysis, or if they do, the nature of the collected data will not be sufficient to allow the model to converge and provide model estimates (Valcic, 2008). The following section discusses alternative discrete choice models which allow for a relaxation of the IIA assumptions.

5.5.3. Discrete choice model alternatives with relaxed IIA assumptions

When economic and physical circumstances do not subscribe to the assumption that the unobserved portion of utility ε_{njt} is independently and identically distributed (iid), other discrete choice models, which relax the IIA assumption, can be employed.

In the case of the nested logit, the analyst need not deviate from the format of assuming IIA, however this assumption is limited to particular groups of alternatives, so that variance is allowed to differ across nests, but within any one nest, the unobserved factors of utility have the same correlation. However, as outlined by Valcic (2008), the nested logit applies best to situations where a clear distinction of alternatives allows them to be defined as "geographically distinct clusters". As in the fishery analysed by Valcic (2008), the fishing location alternatives of the Irish bottom trawl fishery cannot be easily categorised in such a way, making it impossible to "define meaningful nest structures and subsets of alternatives".

Two other discrete choice models which allow for a relaxing of the IIA assumptions are the multinomial probit and the mixed logit or random parameters logit (RPL). These allow the IIA assumptions to be relaxed by allowing correlation in the unobserved factors of utility, or heteroskedasticity in the variance associated with any particular alternative at any time in a panel data set, to be factored into the model and its estimates. Of the two models, the RPL is more suited to a study of location choice in the Irish bottom trawl fishery because the multinomial probit requires a small number of alternatives; Valcic points out the work of Greene (2002) who claims the model becomes difficult to estimate once more than 4 alternatives are considered. The study area in this paper includes a total of 34 sites and is thus well outside the bounds of what the multinomial logit can accommodate; Valcic (2008) alludes to the fact that having 20 alternatives in a probit model is really possible in principle only. A further reason for using the multinomial logit rather than a mixed logit was due to a lack of convergence when the model was run using the latter approach.

Hensher and Greene (2001) have referred to the mixed logit model as "the most promising state of the art discrete choice model currently available". Despite this, many studies using discrete choice methods to analyse fishing choices in commercial fisheries have shied away from the use of these models, employing instead, MNL and the nested logit; (Eales and Wilen 1986; DuPont 1993; Holland and Suttinen 2000; Smith and Wilen 2003; Curtis and McConnell 2004; Hicks et al. 2004; Strand Jr. 2004). This seems unusual considering the fact that the nested logit has a limited capacity to deal with heteroskedasticity and autocorrelation and the MNL does not deal with these issues at all. However, the prominence of these models in the literature is largely explained by the fact that alternatives like the mixed logit are highly complex and difficult to estimate. Hensher and Greene (2001) allude to this, saying that 'although the theory is relatively clear, estimation and data issues are far from clear'.

The main undertaking of the RPL is to allow for the possibility that unobserved information that influences decision-makers' choices is sufficiently rich to induce correlation across alternatives in each choice situation, and across multiple choice occasions. It does this by partitioning the stochastic component of utility into uncorrelated parts. In one part, there is correlation over alternatives and heteroskedasticity exists, while in the other, the iid assumption across individuals and alternatives holds. Thus the utility expression,

$$U_{qj} = \beta_q X_{qj} + \varepsilon_{qj}$$

becomes,

$$U_{jq} = \beta' x_{jq} + \left[\eta_{jq} + \varepsilon_{jq}\right]$$

where η_{jq} is a random term with zero mean whose distribution over individuals and alternatives depends on underlying parameters and observed data relating to alternative j and individual q. The distribution of η can follow a normal, lognormal, triangular etc. distribution. ε_{iq} is a random term with zero mean that is iid over alternatives and does not depend on underlying parameters or data. It follows an iid extreme value distribution. As the variance of ε_{iq} cannot be separately identified from β, it is normalised to set the scale of utility. Hesnsher and Greene (2001) denote the density of η by $f(\eta|\Omega)$ where Ω represents the fixed parameter values of the distribution. For a certain value of η, the conditional choice probability that arises is the standard logit, since the remaining error term is iid extreme value:

$$L_j(\eta) = \exp\left(\beta' x_j + \eta_j\right) / \sum_j \exp(\beta' x_j + \eta_j)$$

The choice probability in the mixed logit model is simply this logit formula integrated over all values of η weighted by the density of η:

$$P_i = \int L_i(\eta) f(\eta|\Omega) d\eta$$

It is from here that the model gets its name, the mixed logit; the choice probability is made up of various logits with f as the mixing distribution. The IIA assumption does not apply to the probabilities and different substitution patterns are obtained by appropriate specification of f. The random parameter specification of the model specifies each $\beta_q$associated with an attribute of an alternative as having both a mean and a standard deviation. This contrasts with the standard logit or MNL whereby the standard deviation of each individual q's β is defined to be zero, such that all behavioural information is captured by the mean.

Because each $\beta_q$in the mixed logit has an associated standard deviation, the model allows for preference heterogeneity. While the distribution of these parameters is unknown, making selection of the correct distribution difficult, this can be overcome by retrieving the individual specific preferences by deriving an individual's conditional distribution on their choices recorded in the sample data. Bayes rule allows this conditional distribution to be defined:

$$H_q(\beta|\theta) = L_q(\beta) g(\beta|\theta) / P_q(\theta)$$

$L_q(\beta)$ is the likelihood of an individual's choice if they had this specific β, $g(\beta|\theta)$ is the distribution in the population of β's and $P_q(\theta)$ is the choice probability function defined in open form as:

$$P_q(\theta) = \int L_q(\beta) g(\beta|\theta)\, d\beta$$

Recall earlier in the discussion on discrete choice models, the point that specification of the probability distribution can be altered by making different assumptions about the distribution of the unobservable component of utility, which in turn leads to different types of discrete choice models. In this case, the model, or probability estimate, is arrived at via the open form solution,$\int L_q(\beta) g(\beta|\theta)\, d\beta$. Thus, in order for the integral to be approximated, it is evaluated numerically through simulation. Assuming the value of the parameters has been estimated, a value of η is drawn from its distribution, upon which, the logit formula $L_j(\eta)$ is calculated. This process is carried out many times, producing a mean value for $L_j(\eta)$, which is used as the approximate choice probability estimated by the model. This process can be written,

$$SP_j = (1/R) \sum\nolimits_{r=1,\dots,R} L_j(\eta^r)$$

where R is the number of draws of η, η^r is the r^{th} draw and SP_i is the simulated probability that an individual chooses alternative j.

In this analysis, attempts to operate the mixed logit using the available data were made, however, due the large number of choice occasions per vessel being combined with a large number of alternatives, the procedure proved impossible i.e. the model would not converge. This is because the logit formula must be calculated not just once for each decision maker, but once for each choice occasion for each decision maker, making the model overly cumbersome, and very difficult to estimate. Additionally, the mixed logit or RPL requires very high frequency of observations in the data. As pointed out by Valcic (2008), the failure to converge can occur when it is impossible to find the optimum of the maximum likelihood function, which can arise as a result of the function being too flat or otherwise irregular, i.e., undermined by data inconsistencies, such as too many zeroes for some variables. In the case of the data used in this study, many species are not caught in *most* of the sites, and even the most prominently harvested species are predominantly found in a small number of key fishing locations. This means that by its nature, the data exhibits a high level of zeroes for various species at various locations for each choice occasion. I believe this may have contributed to a flat or irregular maximum likelihood function. In an ideal setting, some amount of each species would be recorded as harvested at each of the 34 fishing locations. Unfortunately, due to the fact that this is not the case, and due to the cumbersome nature of the model when a large number of alternatives is combined with repeated calculation of the logit formula for each choice occasion, the mixed logit is not estimated, despite its obvious advantages with respect to efficiency in its estimator in the face of autocorrelation and heteroskedasticity over the chosen conditional logit model.

5.6. Methods

5.6.1. Model Variables

As earlier mentioned in the methodology section, the discrete choice model finally selected to analyse fishing location choice in Irish demersal otter trawl fisheries was the conditional logit. To specify the model, variables had to be constructed using the collected VMS location, logbook and sales notes data. These variables were defined as follows:

i. *Choice:*

$$(Y_{nit}): Y_{nit} = \begin{cases} 1\ if\ for\ trip\ t\ the\ fishing\ vessel\ n\ chooses\ to\ fish\ in\ location\ i \\ 0\ if\ the\ fishing\ location\ is\ not\ selected \end{cases}$$

ii. *Distance* $(X1_{nit})$:

This variable is measured as the km distance from the port which vessel n departs, to the midpoint of each fishing location i for choice occasion t. It is important to note that there is an observation for the distance to all sites on each choice occasion and therefore all site distances are used as explanatory variables, even if not the selected site. The variable therefore varies according to the port of departure of any vessel. Like Valcic (2008) I expect the sign on the estimated coefficient of this variable to be negative since a greater travel distance will be associated with a higher fuel (variable) cost, diminishing expected profits and a fisherman's expected utility, therefore the larger this variable for a particular fishing location, the lower the probability of a fisherman choosing to fish there.

iii. *Total annual kg live-weight of species j harvested at location i* $(X2_{ij})$*:*

Many studies analysing the fishing location choice use expected revenue at location i on choice occasion t as predictive variable in the discrete choice model. Naturally, expected revenue of any site will act as a central determinant of fishing location choice since the higher the expected revenue possible at any particular site, the greater a contribution that site can make to fishermen's expected utility and therefore the higher is the probability that fishermen will choose it as their selected fishing location for any trip. Usually studies that include this variable in a discrete choice model are focused on the behavioural impact of an MPA, the entry exit decision etc. However, this study has a specific focus on the impact of quotas on fishing location choice. Fishing vessels in Irish demersal otter trawl fisheries are heavily restricted by quotas, and evaluate a fishing sites potential upon this basis. This is partially an expected revenue based decision since revenue will depend on the quantity harvested, but given the strict legislative constraints placed upon fishermen in these fisheries, quota is the deciding factor in how and where to fish, and therefore focusing on the expected live-weight catch of species j at location i for the entire fleet is likely to be the closest way to approximate a fisherman's actual objective function. This seems fair because demersal otter trawl fishermen are very much price takers, and even in a landscape of shifting prices, their targeting behaviour and therefore their decision on where to fish will primarily be determined on the basis of quota permissibility. The species live-weight variable indicates the most likely place that fishermen expect to locate the species for which they have available quota. The targeting choice, and therefore location choice, will be determined on this basis primarily.

While the expected revenue at each fishing site, scaled according to vessel characteristics like vessel length and engine power could be used as a predictive variable, it would be highly likely that the model would suffer mutlicollinearity since expected revenue is calculated using the historical catch of multiple species at alternative fishing sites, and this information is already used in the model for individual species live-weight variables. Having experimented with using the expected revenue at each site as a predictive variable when also using species live-weights across alternative sites as a predictive variable, this was indeed the case (i.e. multicollinearity resulted). In fact the estimated coefficient of expected revenue became increasingly smaller, and eventually became negative, when more species live-weight variables were added to the model.

5.6.2. Defining fishing location alternatives

The definition of fishing location alternatives in this paper arises out of the work of Gerritsen et al. (2012). Indeed the data used to construct fishing location alternatives in that study is the same data used in this analysis. This section details how these fishing location alternatives were decided upon and constructed. It is particularly relevant since when I go on to discuss the results of my analysis, that is, the change to fishing site visitation probabilities that arise out of simulated quota changes in the predictive model, I place particular emphasis on the impact of the quotas on groups of locations which are linked by the dominant species with which they are associated in the Gerritsen et al. (2012) study. Therefore it is important that the reader have an understanding of how these links and associations are defined.

The first step in the process was to use the VMS location data to determine which areas of the open seas were bonafide fishing locations. As described by Gerritsen and Lordan (2011), VMS automatically collect positional data from fishing vessels by transmitting a signal via satellite every two hours. A speed rule is then applied to this data, such that vessel-speed criteria can be used to infer whether a VMS record corresponds to fishing activity. In the case of demersal otter trawl fisheries, the speed at which vessels travel while fishing can lie anywhere between 1.5 and 4.5 knots. Therefore the data was filtered for location observations with travel speeds within these boundaries. According to Gerritsen and Lordan (2011), vessel speed can distinguish fishing activity with an accuracy of 88% and most errors (both false-positive and false-negative) occurred around the start and end of fishing operations.

Once areas of fishing activity are determined using VMS data and the speed-rule, daily retained catch data could be applied equally to such areas for each vessel on each date. The resulting catch data could then be aggregated to the grid of 0.10° longitude $\times$ 0.05° latitude. In this case, the researchers had to balance between achieving a good level of spatial resolution and allowing for a sufficient number of data points within each grid cell. The data used was from 2006-2009 and grid cells with less than 5VMS location observations that corresponded with fishing activity, or grid cells with less than 100kg retained catch were omitted from the analysis.

Once historical retained catch data had been combined with location information it was possible to define regions according to the dominant species harvested within them. To do this the researchers applied a hierarchical cluster analysis to the now spatially-defined historical catch records. Importantly, 10 key species types were used to define clusters according to similarities in historically retained catch species compositions at various locations. A table from the study defining these species types and their historical contribution to retained catch is shown in Table 5.1. Naturally, a host of species will be harvested and retained across fishing locations, but to define areas accurately, focus was given to species which represented the bulk of commercially harvested species. According to the study, these 10 species categories accounted for 90% of total demersal landings in the study period and all other species were grouped into an 11th category ('other'). The retained catch weights by species category in each grid cell were converted to proportions. Next, a dissimilarity matrix was constructed by calculating the Euclidian distance between the cells using the proportions of the 11 species categories to define their location in 11-dimensional Euclidian space. A hierarchical cluster analysis, using Ward's minimum variance clustering algorithm (Gordon, 1987), was then applied to this matrix.

Table 5.1: The 10 most common species classes in the catches which were used in the cluster analysis and their contribution to the landings.

Code	Common name	Scientific name	Landings
Cod	Cod	*Gadus morhua*	3%
deep	All deepwater species	*Mostly Phycis blennoides, Heliocolenus dactyloptreus, Apogonidae, Coryphaenoides rupestris, Aphanopus carbo and Lepidopus caudatus*	4%
had	Haddock	*Melanogrammus aeglefinus*	10%
hke	Hake	*Merluccius merluccius*	5%
meg	Megrim	*Lepidorhombus whiffiagonis and L. boscii*	7%
mon	Monkfish	*Lophius piscatorius and L. budegassa*	14%
nep	Nephrops	*Nephrops norvegicus*	30%
pok	Saithe	*Pollachius virens*	2%
ray	All rays and skates	*Generally not identified to species level in the logbooks database*	4%
whg	Whiting	*Merlangius merlangus*	12%
other	All other demersal species		10%

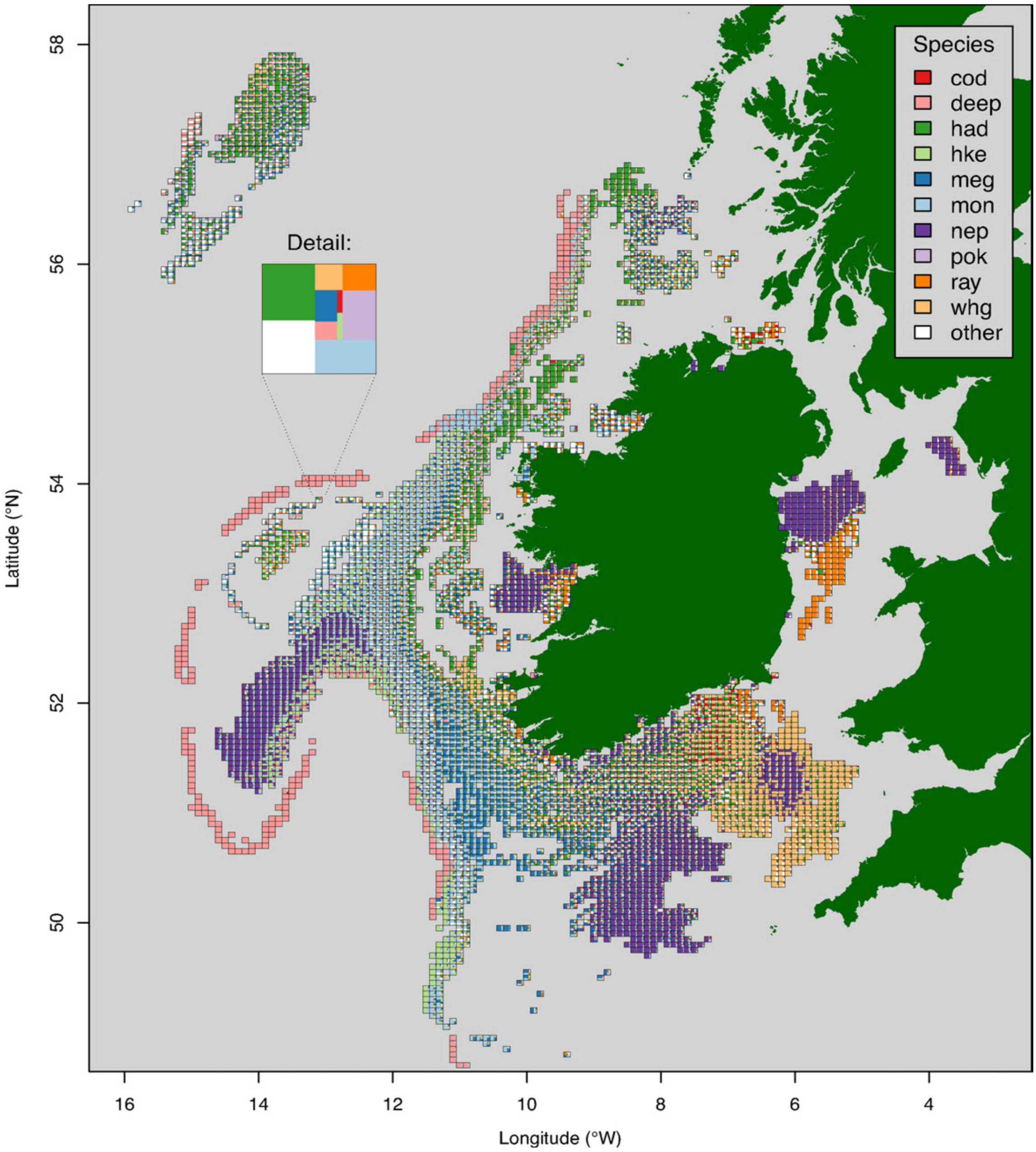
Adapted from Gerritsen et al. (2012).

The researchers (Gerritsen and Lordan, 2011) had to decide how many clusters would allow them to best reflect the alternative location choices that fishermen actually faced: if the number of clusters decided upon was too low, then species that occur in distinct habitats would have been grouped together but if the number of clusters was too high, then similar fisheries would have been assigned to different clusters. After expert consultation to decide on an appropriate number of clusters, the spatial distribution of these clusters was mapped and regions with cells of the same cluster that were contiguous were defined by manually drawing polygons around these regions. To improve the clarity of fishing location alternatives, depth contours and information on bottom type were used to improve the boundary definition. Furthermore, while the clustering technique led to 11 distinct clusters (see Fig. 5.1), a larger number of regions than clusters was eventually defined, due to the possibility that one species composition cluster might occur on two or more spatially distinct fishing grounds. Boundaries were drawn manually around regions with cells of the same cluster. The resulting regions were given names of main fishing grounds or geographical features in order to identify them. The resulting 34 fishing regions of the cluster analysis and geographically defined boundary depictions can be viewed in Fig 4.1[22]. The summary statistics for each of the 34 regions provided by Gerritsen et al. (2012) are reported in that study. Due to similarities in proximity of areas Labadie 1 and Labadie 2 and the similar live-weight retained catch profiles of each location, these were treated as one fishing location alternative in this particular study and thus there were a total of 33 discrete alternatives modelled in the analysis.

To summarise, the clustering algorithm identified clusters that were spatially discrete. Therefore it was possible to identify a number of regions that were largely objectively defined, i.e. requiring only a small amount of expert interpretation. For regions with a patchy distribution (e.g. the deep cluster), additional information like depth contours or bottom type were needed to draw appropriate boundaries.

[22] The cells of the Deep cluster were not contiguous in some places, but it was decided to include all these into a single region. Depth contours were used to determine the boundaries of this region in areas where data was sparse. In the Smalls region (to the south-east of Ireland), the Nephrops and Nephrops-mixed clusters were merged as these correspond to a single Nephrops fishery taking place on a well-defined mud patch. Similarly, small and spatially distinct fishing grounds like Blackstones and Cape (to the north of Ireland) were also defined as single regions even though they contained cells from a number of different clusters.

Figure 5.3: Species composition of the landings of Irish demersal otter trawlers during 2006–2009 in Irish and UK waters.



Each grid cell contains a 2-dimensional barplot in which the area of each colour in the grid cell is proportional to the species composition by weight (the plot was created using the R package "mapplots"). The cells are 0.10◦ longitude × 0.05◦ latitude in size.

5.7. Analysis and results

As earlier mentioned in the methodology section, the discrete choice model finally selected to carry out the analysis was the conditional logit model. The total number of observations in the data set for all otter trawl demersal fishing vessels fitted with VMS devices in 2009 was 205,564, (6,046 trips each with a potential outcome of one of 34 fishing location choices).

5.7.1. Interpretation of estimated coefficients of conditional logit

Table 5.2 shows the list of variables used in the model and Table 5.3 shows the model estimation results produced by the conditional logit model for the entire data set.

Table 5.2: Model Variables

Model Variables	Model Symbol	Estimation Acronym
Distance	X_{nit1}	DIST
Liveweight of:		
Cod	X_{nit2}	COD_KG
Deepwater species	X_{nit3}	DEEP_KG
Haddock	X_{nit4}	HAD_KG
Hake	X_{nit5}	HKE_HG
Megrim	X_{nit6}	MEG_KG
Monkfish	X_{nit7}	MON_KG
Nephrops	X_{nit8}	NEP_KG
Rays and skates	X_{nit9}	RAY_KG
Saithe	X_{nit10}	POK_KG
Whiting	X_{nit11}	WHG_KG
Other demersal species	X_{nit12}	OTHER_KG

Results displayed include the mean estimated coefficient for each species, standard deviation of each coefficient, and goodness of fit measures. All variables are statistically significantly different from zero at the 95% confidence level. For individual species variables, there is some variation in the signs on the coefficients but this is not unusual or undermining for a conditional logit analysis.

The signs on COD_KG, HAD_KG, HKE_KG, MEG_KG, MON_KG, NEP_KG, RAY_KG AND POK_KG are all consistent with economic theory and of a positive sign. This means that an increase in the retainable live-weight catch of any of these species at an area will increase the probability of that area being chosen as a fishing alternative at the outset of a trip. Distance, or DIST, exhibits a negative sign; thus an increase in the distance a vessel must travel to reach a fishing ground will have a negative effect on the probability a particular site being chosen.

In the case of negative coefficients, it is important to recognise that the coefficients represent on average outcomes only. For example, where more Hake is being caught fishers are less likely to visit on average, so the sign is negative, yet within the fleet, if we were to dig down deeper into the data, there would be individuals who are more likely to visit a site that has a higher abundance of Hake. However the conditional logit methodology prevents such further analysis.

The results of the conditional logit regression are consistent with findings from other studies. In particular, Eales and Wilen (1986) also used vessels' own live-weight estimates of retained catch at different locations and found that this variable had a positive relationship with the probability of an area being chosen as the location at which to fish. Valcic (2008), Smith and Wilen (2003) and Holland and Suttinen (2000) all found that the greater the expected revenue associated with an area, the greater the probability that this area would be chosen out of the total set of fishing location alternatives. Of the above studies, three (Smith and Wilen, 2003; Valcic 2008; Eales and Wilen 1986) found that when using distance as proxy for expected travelling cost (instead of using a measure of travelling cost as a variable in the model specifically), the relationship was negative, that is, distance had a negative effect on the probability of vessels selecting a particular fishing location.

The results of Valcic (2008), Holland and Sutinen (2000) and Dupont (1993) appear to support my omission of the standard deviation of species live-weight as a predictive variable of site-choice probability. Valcic (2008) found this variable to be statistically insignificant even at the 50% confidence level. The appearance of risk-seeking behaviour in fishermen is also documented by Dupont (1993) in salmon fishing vessels in British Columbia.

In Irish demersal otter trawl fisheries, the primary concern in fishermen's species targeting choice is the amount of quota available for any species. If allowed to do so, fishermen will continue to target these species until they have reached their quota, regardless of how much catch variability they face in doing so. Vessels in this fishery operate in a highly leveraged fashion, in areas with few other economic opportunities, and therefore vessel operators must simply accept whatever catch variability is associated with a species catch potential, which is primarily determined by quotas.

Holland and Sutinen (2000) find evidence of risk-neutrality and risk-seeking behaviour in fishermen in New-England groundfish fishermen yet claim there is evidence that fishermen in multi-species fisheries reduce risks in other ways, such as targeting a variety of species. This idea resembles a suggestion by Hilborn (2001) who discusses the possibility of fisheries management initiatives using species portfolio theory to allow fishermen to broaden and diversify their set of target species so as to smooth incoming revenues over time. However quotas are a problem for such forms of risk management. By being forced to rigidly subscribe to species quotas, fishermen may well be forced to accept high levels of variation in the catch of species for which they have available quota.

In the future even this form of fishing behaviour may be forced to change due to the current phasing in of a discard ban. The EU discard ban will attempt to minimise the catching of juvenile fish and unwanted species and make the landing of any unwanted species a legal necessity. Initially it will apply to pelagic and surface fish but will be relevant to demersal fisheries from January 2016 (RTE, 2013). Because it will negate excessive catch of unwanted by-catch species that have stock levels below MSY, it may involve the reduction of fishing effort for species that have stock levels above MSY that could otherwise be caught freely. In such an instance, these biologically sensitive and by-caught species become known as "choke" species, since they prevent the catching of other potentially sustainably harvested species until such a time as more selective gear can be developed. It is worth noting that the discrete choice methodology applied here, which will attempt to quantify the redirection of displaced effort to other fishing areas following quota changes, would be useful in such an instance, informing fishery managers where new fishing effort will arise, and where sensitive by-catch species are likely to be further affected by fishing effort following quota changes. It will also be useful to inform authorities and stakeholders where choke species are likely to arise and restrain fishery rents, following quota changes.

Table 5.3: Conditional Logit Results

	Coef. Est.	Std. Err.
Model Variable		
DIST	-0.0012263*	0.0000791
CODKG	0.0000071*	0.0000014
DEEPKG	-0.0006173*	0.0000494
HADKG	0.0000084*	0.0000009
HKEHG	-0.0000066*	0.0000010
MEGKG	0.0000026*	0.0000005
MONKG	0.0000124*	0.0000011
NEPKG	0.0000009*	0.0000001
RAYKG	0.0000126*	0.0000013
POKKG	0.0000123*	0.0000011
WHGKG	-0.0000013*	0.0000002
OTHERKG	-0.0000188*	0.0000027
Goodness of Fit		
Number of observations		205,564 (6,046 trips x 34 location-alternatives)
Log likelihood function (LL(β̂))		-17,728.779
Restricted log likelihood		-21,320.376
Chi-squared		7,183.15
LR Chi-squared (12) 1-LL(β̂)/LL(0)		0.16846

Rather than relying on the R-squared statistic[23] to determine the goodness of fit of the model I use the likelihood ratio index test which involves estimating two conditional logit models under different circumstances and comparing them. The first model is run using the estimated parameters based on the entire data set and the second is run when all parameters are assumed to be zero. This produces two log likelihood ratios, $LL(\hat{\beta})$ and $LL(0)$ respectively. Table 5.3 shows the likelihood ratio index, which is defined as $1 - LL(\hat{\beta})/LL(0)$. This likelihood ratio index test compares the log likelihoods of the two models and tests whether this difference is statistically significant. The likelihood ratio index of the estimated model is 0.1685 and thus

[23] I do not use the R-squared statistic to interpret the goodness of fit of the model as logistic regression does not have an equivalent to the R-squared that is found in OLS regression and therefore it can be misleading to do so.

the results of the test show that the difference between the two models is statistically significant, i.e., that the less restrictive model based on the estimated parameters $\hat{\beta}$ fits the data significantly better than the more restrictive model, where all parameters are assumed zero. The statistic has a chi-squared distribution with degrees of freedom equal to the difference in the number of degrees of freedom between the two models (in this case, 12; distance, 10 species variables and the 'other' species category). Experimenting with the Wald test also produced results consistent with the LR test.

5.7.2. Model validation: comparing areas' estimated and actual percentage share of trips

The reason for estimating variable coefficients in the previous section is to use these model parameters to predict the fishing location choice of vessels in Irish demersal otter trawl fisheries. Such a model can then be used to simulate the impact of quota changes on the fishing location choice. This is the key advantage to using discrete choice models; it allows the analyst to predict the impact of a policy before the policy has been implemented. If the model performs well, and allows the parameters to accurately predict vessels' fishing location choice, then there should be a good degree of similarity in the percentage share of trips to all areas that is actually observed in the data and the predicted percentage share of trips posited by the model. If this is the case, then it will be easier to rely on changes to site-choice probabilities that the model predicts when quota constraints are added to the simulation, i.e. the model will be a better predictor of the impact of a policy change. This form of model validation is rare and to the best of my knowledge the main example is from Valcic (2008). In this section then, I compare the actual percentage of trips made to the 34 fishing sites with the percentages estimated by the model.

From section 5, recall the equation:

$$P_{ni} = Prob(\varepsilon_{njt} - \varepsilon_{nit} < V_{nit} - V_{njt})$$

$$= \int_{\varepsilon} I(\varepsilon_{njt} - \varepsilon_{nit} < V_{nit} - V_{njt}) f(\varepsilon_{nt}) d\varepsilon_{nt}$$

Using the models estimated parameter values and the collected data allows the analyst to approximate the above probability equation, i.e. the probability that fisherman n will choose area i on choice occasion t. The resulting probabilities of this exercise and the actually observed (in the data) percentage share of trips to each site are shown in Table 5.4. This information is also displayed graphically in Fig. 5.3.

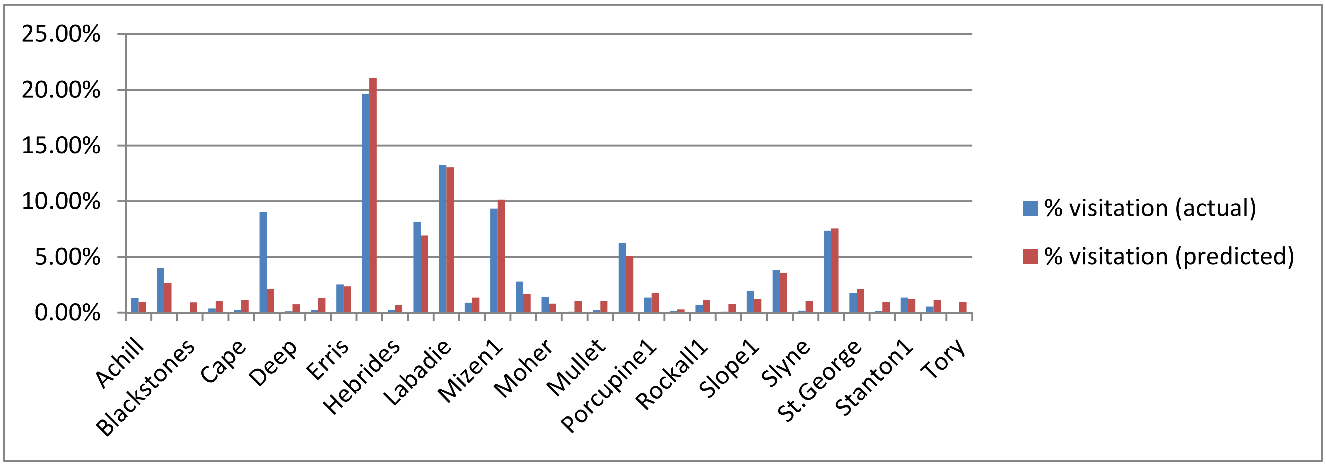
Table 5.4 shows the difference between predicted and actual area-visitation percentages. The difference between the two is less than 0.5% for 12 out of the 33 fishing areas, and between 0.5% and 1% for 14 others. For a further 6 fishing areas the difference between actual and predicted site-visitation lies between 1% and 2%. The only real outlier is fishing area Cork, where the difference between predicted and actual area-visitation is 6.97%. It is difficult to suggest reasons why this is the case. The probability of visiting a site is overestimated in 22 cases and underestimated 12, albeit by less than 1% in 26 out of 33 cases.

Table 5.4: Actual and predicted site visitation percentages for sample period

Fishing Area	% visitation (actual)	% visitation (predicted)	Predicted - Actual
Achill	1.29%	0.96%	-0.33%
Aran	4.02%	2.67%	-1.35%
Blackstones	0.07%	0.92%	0.86%
Blaskets	0.38%	1.06%	0.68%
Cape	0.28%	1.16%	0.88%
Cork	9.06%	2.10%	-6.97%
Deep	0.13%	0.76%	0.63%
Donegal	0.28%	1.29%	1.01%
Erris	2.55%	2.36%	-0.18%
Galley	19.63%	21.05%	1.41%
Hebrides	0.26%	0.70%	0.44%
Irishsea	8.17%	6.92%	-1.25%
Labadie	13.26%	13.04%	-0.22%
LoopHead	0.91%	1.35%	0.45%
Mizen1	9.35%	10.15%	0.80%
Mizen2	2.80%	1.70%	-1.10%
Moher	1.41%	0.81%	-0.60%
Morecambe	0.08%	1.05%	0.96%
Mullet	0.25%	1.05%	0.80%
Nymphe	6.25%	5.08%	-1.18%
Porcupine1	1.36%	1.79%	0.44%
Porcupine2	0.15%	0.30%	0.16%
Rockall1	0.71%	1.14%	0.43%
Rockall2	0.10%	0.78%	0.69%
Slope1	1.95%	1.23%	-0.72%
Slope2	3.84%	3.55%	-0.29%
Slyne	0.18%	1.05%	0.87%
Smalls	7.36%	7.55%	0.19%
St.George	1.80%	2.14%	0.33%
Stags	0.15%	0.99%	0.84%
Stanton1	1.37%	1.22%	-0.15%
Stanton2	0.55%	1.12%	0.57%
Tory	0.05%	0.94%	0.89%

Note that only 33 sites are shown because sites Labadie 1 and 2 were combined

Figure 5.4: Bar chart of actual and predicted site visitation percentages for sample period



As pointed out by Valcic (2008), one reason for differences between the predicted and estimated percentage shares of fishing effort may be the fact that the model does not implicitly account for the proportion of habitat type in each area. Area characteristics like ground type or depth therefore act as omitted variables, skewing the estimation somewhat and causing bias in the model's betas. The impact of this omitted variable bias is likely to be less significant in this study than in the Valcic study however because Gerritsen manually drew some of the area boundaries on the basis of ground-type and furthermore used depth contours and information on bottom type to improve the boundary definition. It is likely however that superior measures of habitat type, depth and other area-specific characteristics would reduce the possible bias in the model's estimators, leading to better predictions of the spatial-behavioural impact of policy changes on fleet dynamics.

5.8. *Policy application: Simulating the impact of quota changes on fishing location choice*

5.8.1. Results for all fishing areas

As discussed in section 5.3, most discrete choice models used to simulate fishing-based decision-making have focused on the impact of area closures or profit uncertainty on the fishing location decision (Eales and Wilen, 1986; Bockstael and Opaluch, 1983; Dupoint, 1993; Larson, Sutton and Terry, 1999; Holland and Sutinen, 2000; Mistiaen and Strand, 2000; Curtis and Hicks, 2000; Smith and Wilen, 2003; Smith and Zhang, 2007; Valcic, 2008. Eggert and Tvertas (2004) use estimates of revenue and the standard deviation of revenue to model gear choice while others model the entry exit decision for a single fishery using risk trade-off measures (Smith and Wilen, 2005) or variables such as price, abundance of target-species in the fishery and changes to fleet size (Ward and Sutinen, 1994). At time of writing, I am unaware of any discrete choice analysis which has modelled the impact of quota changes on effort displacement and the fishing location decision.

The first step in simulating quota changes in Irish demersal otter trawl fisheries was to determine realistic or potential future changes to species quotas. In December 2013 the European Fisheries Council announced changes to quota for several species relevant to the Irish demersal otter trawl fleet; see Table 5.5. For six species categories, there was no change in species quota, for a further four the change in quota was negative and for only one species (Monkfish) did the changes result in an increase in quota.

Table 5.5: EU Fisheries Council changes to species quotas for 2014

Species	Quota change
Cod	None
All deepwater species	None
Haddock	-33%
Hake	None
Megrim	None
Monkfish	15%
Nephrops	-9%
Saithe	None
All rays and skates	-10%
Whiting	-23%
Other demersal species	None

Unlike the simulation of an area closure, which involves removing closed areas from the fishermen's choice set (e.g. Valcic, 2008) there is no change to fishermen's location choice set when simulating quota changes. Rather, the potential of each location to produce species for which quotas have changed is now changed (as far as fishermen's bottom line is concerned) by regulation. For this reason fishers will reduce fishing effort in areas where species with reduced quotas are dominant and vice versa. From a by-catch perspective, the important detail is that the redirected effort will result in a change to discard patterns. To reiterate an earlier made point, the only barrier to determining what the change in by-catch levels may be is attaining estimates of by-catch in each area. Armed with estimates of by-catch of demersal otter trawling for different species in each of the 33 areas and the predicted effort reallocations of the present conditional logit model, fisheries scientists may be able to predict some of the changes in by-catch in the fishery that result from the quota changes. As earlier alluded to however such by-catch estimates are scarce.

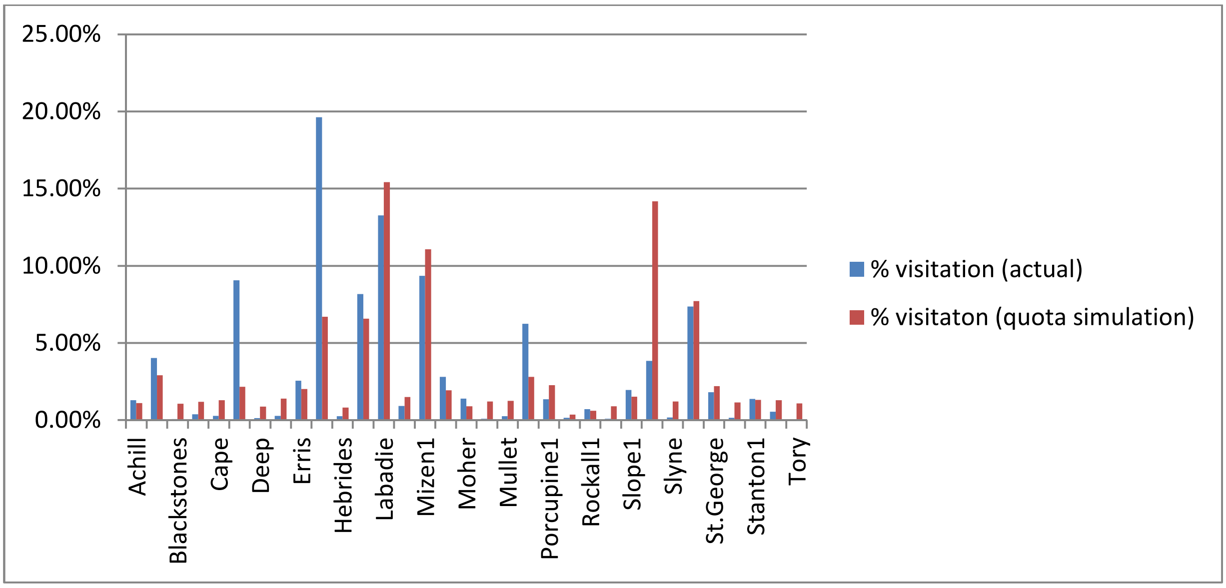
Proportionately restricting (or increasing) the capacity of species with changed quotas to contribute to the live-weight retained catch at each of the 33 fishing areas, and recalculating the probability of site visitation at the outset of each trip, produces the post-quota estimates of percentage share of fishing trips to each area shown in Table. 5.5 and Fig. 5.4. Overall, the simulation suggests significant decreases in percentage site visitation in some areas and significant reallocation of displaced effort into alternative fishing locations. Note that an assumption of the model is that the number of trips made in the Irish demersal otter trawl fisheries will remain unchanged following quota changes. While it is possible that quota changes may cause fishermen to exit the fishery to pursue other economic opportunities, this is not reflected in the model. The degree of displacement predicted by the model will be exaggerated if this is the case.

Table 5.6: Actual site visitation percentages and predicted percentages following the quota change for the sample period

Fishing area	% visitation (actual)	% visitaton (quota simulation)	Quota-sim minus Actual
Achill	1.29%	1.10%	-0.19%
Aran	4.02%	2.91%	-1.11%
Blackstones	0.07%	1.07%	1.00%
Blaskets	0.38%	1.19%	0.81%
Cape	0.28%	1.30%	1.02%
Cork	9.06%	2.16%	-6.90%
Deep	0.13%	0.88%	0.75%
Donegal	0.28%	1.40%	1.12%
Erris	2.55%	2.02%	-0.53%
Galley	19.63%	6.69%	-12.94%
Hebrides	0.26%	0.81%	0.54%
Irishsea	8.17%	6.57%	-1.60%
Labadie	13.26%	15.42%	2.16%
LoopHead	0.91%	1.49%	0.58%
Mizen1	9.35%	11.08%	1.73%
Mizen2	2.80%	1.94%	-0.85%
Moher	1.41%	0.90%	-0.51%
Morecambe	0.08%	1.21%	1.13%
Mullet	0.25%	1.24%	0.99%
Nymphe	6.25%	2.81%	-3.45%
Porcupine1	1.36%	2.26%	0.90%
Porcupine2	0.15%	0.36%	0.21%
Rockall1	0.71%	0.62%	-0.09%
Rockall2	0.10%	0.89%	0.79%
Slope1	1.95%	1.53%	-0.43%
Slope2	3.84%	14.18%	10.34%
Slyne	0.18%	1.22%	1.04%
Smalls	7.36%	7.71%	0.35%
St.George	1.80%	2.20%	0.40%
Stags	0.15%	1.14%	0.99%
Stanton1	1.37%	1.31%	-0.06%
Stanton2	0.55%	1.29%	0.74%
Tory	0.05%	1.09%	1.04%

Differences or estimated spatial impact of the quota changes is also shown in column 3

Figure 5.5: Bar chart of actual and post-quota change predicted site visitation percentages for sample period



5.8.2. Analysis of results by location cluster

To interpret the results of the simulation in more detail, I analyse the impact of quota changes according to groups or clusters of areas defined by Gerritsen et al. (2012) and as demonstrated in Fig. 5.2. These area-clusters are determined according to similarities in the retained live-weight catch of species. For example, “hadmix” forms one cluster in which Haddock is the dominant species harvested, and includes areas such as Rockall 1, Galley, Erris and Hebrides. I analyse the impact of the quota changes in terms of changes in percentage share of trips to each location in the cluster. This is done for all clusters and makes interpretation of the results more clear.

The simulated impact of the quota changes on the percentage share of trips made to each area in the hadmix cluster is shown in Fig. 5.5. Of particular note is the impact of the quota change on the percentage share of trips made to the Galley site (from 19.63% to 6.69%). The Galley site was the area with the largest percentage share of Irish demersal otter trawl trips during the sample period (19.63%) and was the most productive area for the harvest of Haddock; (553,918 kg, or 49.36% of all Haddock caught in the hadmix cluster). Given the fact that this site also accounts for a large harvest of Whiting and Nephrops, it is not surprising that the estimated percentage share of trips fell in the simulations as the quota was reduced for all of these species. As this location also accounts for a sizeable catch of Monkfish, it is likely the impact would have been more significant had quota not been increased on this species. The percentage share of trips made to Rockall 1 and Erris also fell, albeit by a small percentage. It is worth noting that in the case of Rockall 1 and the Galley site, there was a tendency in the model towards upward bias, that is, the model over-predicted percentage shares at each of these sites. This means that for these sites the actual impact in reality may be greater than that predicted by the model. For Erris, Donegal, Hebrides and Porucpine 2 (for which the model displayed upward bias in the predicted site visitation probabilities) the opposite is true; i.e. the model may be overestimating the percentage share of trips to these areas after the quota changes.

Figure 5.6: Comparison of actual and simulated site visitation percentage share of trips within the hadmix cluster

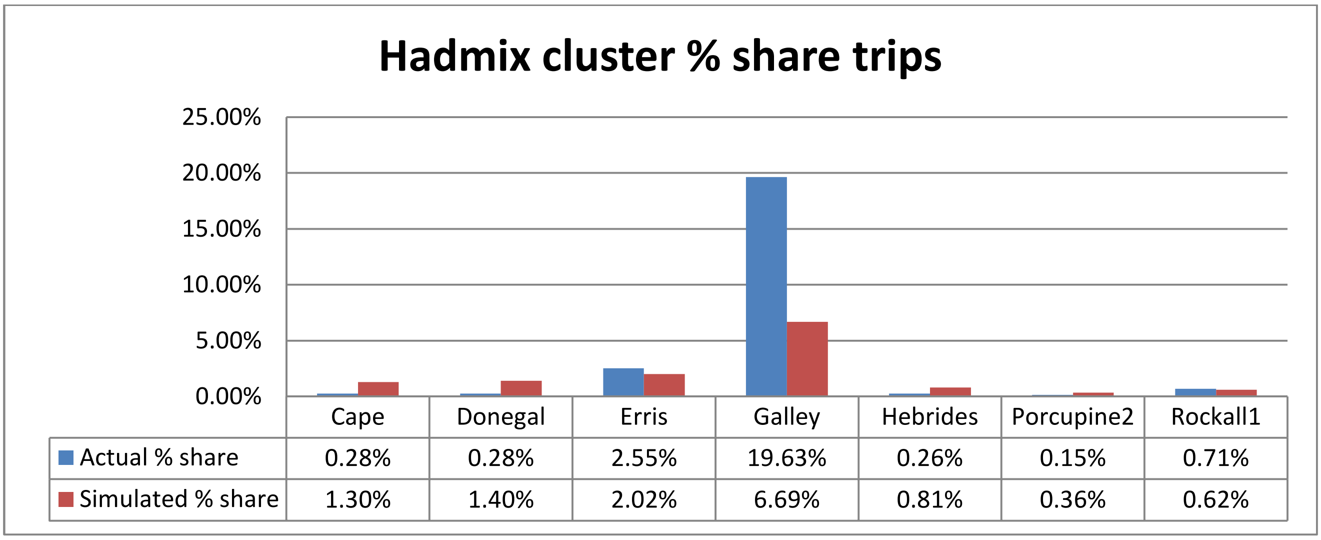
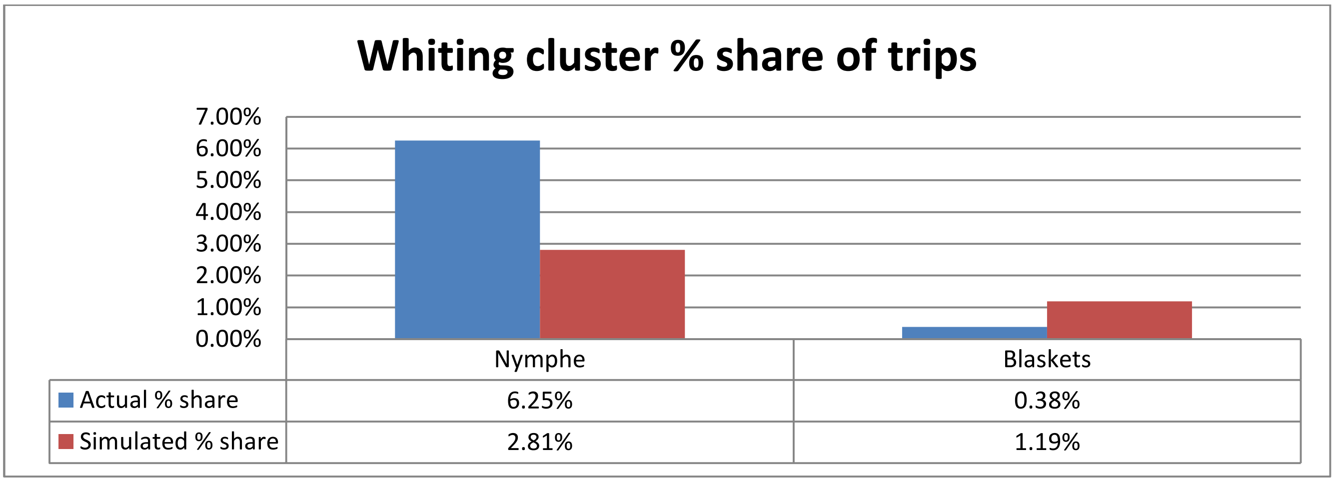


Figure 5.7: Comparison of actual and simulated site visitation percentage share of trips within the Whiting cluster



From Gerritsen et al. (2012) only two areas were included in the Whiting cluster; Nymphe and Blaskets. The results of the simulation indicate significant displacement of effort arising out of the quota changes in the Nymphe area, and an increase in effort in Blaskets. However, the pre-quota model estimations under-estimated the percentage share of trips to the Nymphe site and over-estimated it for Blaskets. Regardless, the extent of the change for the post-quota change simulations are significant enough to suggest that a pattern similar to that observed in the simulations will/would occur in practice.

Changes to the percentage share of trips made to areas in the nephrops and nepmix clusters (see Fig. 5.7 and 5.8) were not as significant as changes within other clusters. Indeed following the quota changes, simulation results indicated that for a number of key nephrops producing sites such as Labadie, Smalls and Stanton 2, the percentage share of trips actually increased. This can be explained by considering two factors; firstly, the level of displacement arising out of areas in the hadmix and Whiting clusters, and secondly, due to increases in the quota on Monkfish (which are caught in significant numbers in many key nephrops sites). In particular, despite the percentage share of trips estimated by the model for Labadie exhibiting a downward bias, the level of visitation is predicted to *increase* significantly following the quota changes. For other sites, while a fall in predicted site visitation after the quota changes is predicted, for these locations, there is a significant existence of downward bias. This is especially true for the Cork site, to the degree that the prediction of the model for percentage share of trips to Cork following the quota changes must be interpreted with caution.

Figure 5.8: Comparison of actual and simulated site visitation percentage share of trips within the neprhops cluster

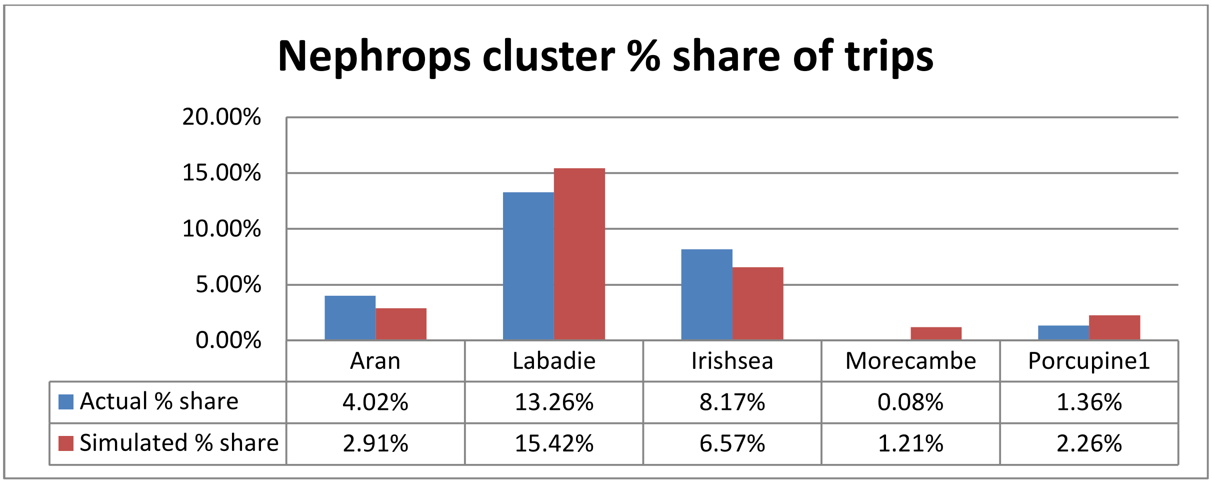
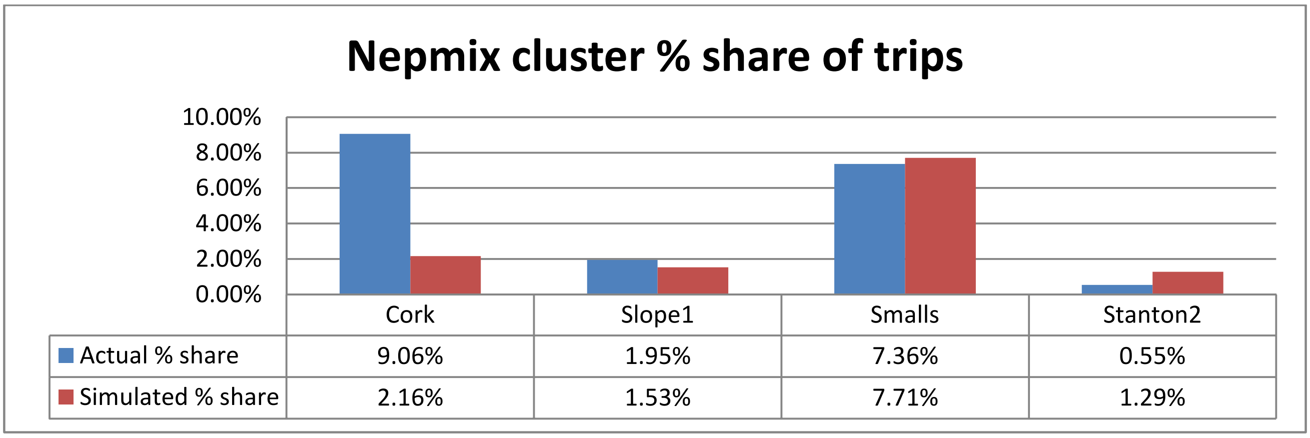


Figure 5.9: Comparison of actual and simulated site visitation percentage share of trips within the nepmix cluster

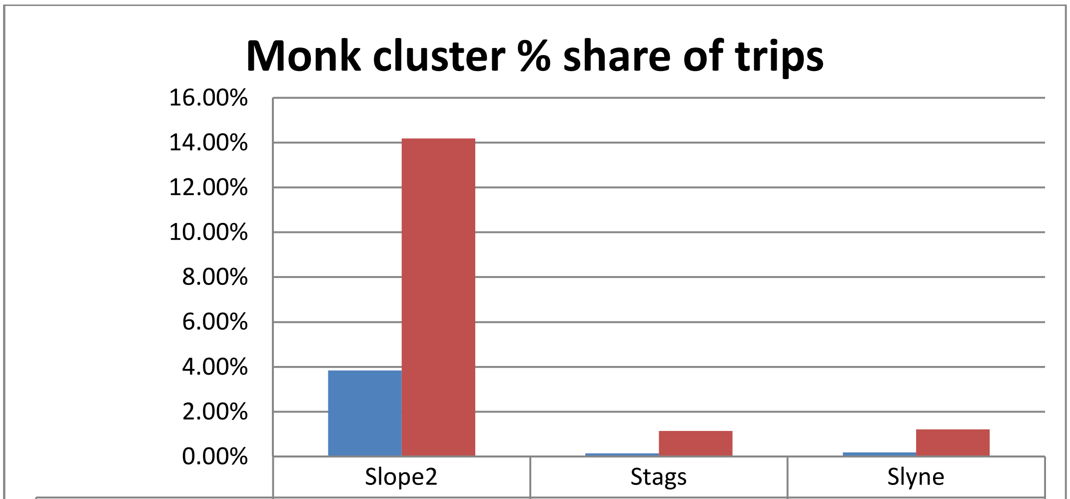


The simulation indicates widespread increase in percentage share of trips in almost all areas in the mixed cluster. While the model's pre-quota change estimates of percentage share of trips to many of these areas were upwardly biased, there remains an indication that effort displaced from the hadmix and Whiting clusters is being absorbed by these areas.

For the ray cluster, the impact is more ambiguous, since the predicted post-quota change percentage share of trips increases for some areas and falls for others. Most of these areas are small and produce only small catches of ray and skate. The most important ray and skate fishery within the cluster is St. George, and the simulation results indicate that despite the quota decrease for ray and skate species, there is a predicted increase in percentage share of trips to this area. Again, this is most likely a result of the high levels of displacement exhibited by the hadmix and Whiting clusters.

Monkfish were the only species for which the quota granted to fishermen in the Irish demersal otter trawl segment was increased in 2014. While Monkfish are caught at many sites in Irish waters, the Gerritsen et al. (2012) monk cluster included sites Slope 2, Stags and Slyne. An increase in percentage shares of trips to all of these areas is predicted in the post-quota change simulation (see Fig. 5.9). However, the change is most notable for the Slope 2 site, which is the main producer of Monkfish across all 33 fishing location alternatives. The suggestion then is that while in other fisheries (for example in the hadmix, Whiting , nephrops and nepmix clusters), the increase in quota on Monkfish may thwart to some degree the displacement of effort from these regions, in sites like Slope 2, Stages and Slyne, it absorbs displaced effort from other fisheries.

Figure 5.10: Comparison of actual and simulated site visitation percentage share of trips within the monk cluster



	Slope2	Stags	Slyne
Actual % share	3.84%	0.15%	0.18%
Simulated % share	14.18%	1.14%	1.22%

5.9. Discussion and Conclusion

Since ecosystem-based fisheries management requires a multispecies perspective, empirical methodologies that can fulfil this criterion are in demand. Typically, empirical multispecies analysis have followed one of two formats; a bio-economic model which determines the optimal harvest rate of more than one species using estimated predator-prey or competitor parameters, or structural ecosystem models that can be used to determine optimal Total Allowable Catches (TACs) across multiple species. The methodology used in this paper provided a third alternative to model the impact of management changes on multiple species by using realistic fishing site options within a discrete choice modelling framework.

The use of such an approach can provide policy makers with an assessment of the ecological, economic and potentially social implications of different designation strategies in order to meet the requirements of policies such as the Marine Strategy Framework Directive (MSFD), the Habitats Directive and the reformed Common Fisheries Policy (CFP) and also in helping to decide on potential conflicts in the establishment of networks of MPAs in European waters. As pointed out by Katsanevakis et al. (2011), many of these conflicts will only be resolved through the use of ecosystem-based marine spatial management, which is now seen as the most appropriate approach for the integrated management of the sea.

With these considerations in mind, this study combined fishing vessel positional data with records of retained catch and economic information, using this information to empirically model the spatial targeting decisions of fishing fleet in the Irish demersal otter trawl fishery. The application of the RUM to this data set allowed for an analysis which sought to estimate the impact of quota changes on fishing site choice. Tentatively, it appears that employing discrete choice modelling techniques in this way can assist fishery managers in estimating the changes in the spatial distribution of fishing effort that arise from policy changes. More specifically, the results indicate that the quotas imposed on the Irish demersal otter trawl fishery will lead to substantial changes in the percentage share of fishing trips made to each area in the fishery. Overall, the results of the multinomial logit used in the analysis were economically consistent and coherent with economic theory and other studies in the area.

Certain caveats, standard when employing such an empirical model, should be observed. Upward and downward bias in the model’s estimator will be exacerbated by the omission of information which influences the fishing location decision, such as skipper skill or habitat type information. The nature of the model’s assumptions must also be borne in mind. One assumption (that fishermen will remain in the fishery after quota changes), if false, will exaggerate the degree of effort displacement to other locations, albeit a small exaggeration.

An important issue in global fisheries management at present is that of the by-catch problem. The relevance in this context is that displacing effort from one fishing site to another will have a connotation for the level of by-catch. Sadly, due to the phenomenon of discards, this information is not recorded and it is therefore difficult to estimate the degree of by-catch taking place in different fishing locations and for different fishing methods and gear types. With the advent of the discard ban however, this situation will change and the degree of by-catch in EU waters will be recorded. The possibility to predict the changes in by-catch that arise from fleet displacement following policy changes is perhaps the most promising area for future research. Further research could also use the fisheries site choice model developed in this paper to investigate the impact of the formation of one or more MPAs on the fishing location decision of demersal otter trawl vessels. Under CFP regulation the formation of MPAs falls under the jurisdiction of member states and the insights garnered from such a study would be helpful in the Irish MPA design process. In addition, the inclusion of cost data in future analyses and the expansion of the dataset to include pelagic trawls would provide a more complete picture of spatial behavioural dynamics in the fishery.

Overall, the model developed here does achieve its objective of providing an insight into the factors driving the fishing site choice decision of the Irish demersal otter trawlers. I would also argue that the predictive power of the model is greatly bolstered by the fact that due to the degree of detail in the VMS data and the spatial mapping techniques used by Gerritsen et al. (2012), fishing site alternatives defined in the study are more in line with fishermen's actual perception of fishing location alternatives. Most importantly the discrete choice methods applied in the study highlight its potential as a tool for ecosystem based fisheries management, taking account of various EU environmental policy initiatives such as the MSFD, MSP and HBD.

6. Conclusions

This chapter provides a summary of each chapter in the thesis and discusses the key findings that have arisen from the empirical chapters. These findings and the connotations for relevant policy and future research are discussed in the context of the core research objectives of the thesis. Section 6.2 draws attentions to the limitations of each methodology employed and asserts certain caveats for interpretation of the results. Section 6.3 recommends future avenues of research, given, and taking stock of, the findings and limitations of the research demonstrated in this thesis. Finally section 6.4 uses the findings of the three empirical chapters to make policy recommendations for Irish coastal and fishery management.

6.1. Key findings

A major theme in the research carried out in this thesis is the significance of stakeholder behavioural dynamics, and the attitudes of stakeholders themselves toward sustainable management, in successful marine and coastal environmental policy design. Specifically, this thesis provides a detailed analysis of the impact of changes in fishery legislation on fishermen's harvesting and/or spatial decision-making and the significance of this behavioural response for wider ecosystem management goals. Through the use of portfolio theory and the expected utility hypothesis, as well as VMS data and spatial mapping techniques, the thesis makes a number of contributions to the fisheries economics literature. Furthermore, the thesis highlights the significance of differences in social groups' attitudes and beliefs for collective consensus-based environmental management. It's major contribution in this regard lies in the proposition (and demonstration) that spatially-orientated analytically qualitative frameworks be used not only to determine socially optimal spatial management policies, but can also be employed as a metric to understand the differences in attitudes toward natural resource and ecoystem management across groups.

Chapter 2 began by covering the broad spectrum of EU marine and coastal policy that exists; from origins, to evolution, to the most up to date policy changes currently underway that are of relevance to the principal research focus of the thesis. The evolution of early water Directives into the overarching WFD was discussed, as well as the relationship between WFD and other forms of environmental management policy such as the MSFD, HBD, ICZM, MSP, and IMP. The complexity and interconnectedness of freshwater, marine and coastal habitats means that in order to be effective in achieving sustainability and measures of good environmental status, these Directives must work synergistically and in unison to bring about long term societal goals. In this respect, EU marine and coastal policy has moved away from sectoral and disjointed policy initiatives and continues to develop a framework which marries the various Directives into a single overarching environmental management agenda.

Chapter 3 analysed the impact of changes to species quotas on the harvesting decisions of fishing fleet in the Irish Cod, Haddock, Whiting and Hake, Monkfish, Megrim fisheries by using the portfolio theory framework and expected utility hypothesis to approximate the objective function of each fleet. The combination of PT and the expected utility hypothesis has been practised very little in fisheries research and its use in this case was especially novel since it was used to predict changes in the retained live-weight catch of numerous species in a multispecies fishery when quota changes occur. Fleet behaviour was further approximated using an exponential weighting factor which attributed greater weight to more recent observations in the fleet's expectation of species prices and catches.

Simulations of hypothetical changes to species quota were run based on the relevant scientific recommendations after which the respondent changes in the species weights of the resulting harvest portfolios were observed. In general, the results indicated that Irish fishing fleet in these two fisheries are relatively efficient in terms of risk-return trade-offs, since the theoretically optimal point of production was similar to that observed in practice. The main undertaking of the study was to ascertain whether or not precautionary quota constraints on a single species in a multi-species fishery affect the retention of catch of alternative species in the fishery or in neighbouring fisheries. The results indicated that this was indeed so, and the study then set about quantifying the degree to which this was the case. The changes that arose in the contribution each species made to the harvest portfolio could then be used to inform fishery managers which species were likely to experience increased retention in live weight catch following quota changes on any given species.

The simulations highlighted the already identified problem of traditional whitefish fleets entering alternative fisheries, such as the Hake, Monkfish and Megrim fishery. Its focus on predicted behavioural responses to protective measures allowed for the factoring of these changes into management decisions so as to avoid unpredicted changes in fishery targeting behaviour. Under the third hypothetical scenario simulation, wherein precautionary measures were placed on Monkfish, the short term alleviation of fishing effort on the traditional whitefish stocks was reversed, and effort again refocused on stocks like Cod and Whiting in the optimizations. This demonstrated the capacity of the model to play a role in more forward thinking planning when adopting a precautionary approach. Ultimately, there is little benefit to alleviating the strain on one stock by temporarily allowing effort to focus on another if in the long term, decline in the stocks of the "alternative" species led fishers back to their original species targeting behaviour.

Chapter 4 presents an application of Biome Portfolio Analysis (BPA) to the Iarras Aithneach peninsula, an Atlantic coastal region located in Connemara, Co. Galway in the West of Ireland. The undertaking had two main concerns. Firstly, was to apply BPA to a coastal region to develop the framework further, as a tool for ICZM. The requirements for such a framework are very specific in that it must be spatially orientated, allow for the participation of multiple interest groups in the community, permit trade-offs between alternative ecosystem services and risks to be assessed and finally, assist coastal managers in making qualitative decisions between alternative policies.

The second concern of the study was to determine the level of coherence in the stated principals of ICZM. In order to manage ecosystems successfully *and* equitably, ICZM attempts to combine strategic, scientifically founded principals, with local, participative ones. The problem this creates however is that a scientific understanding of ecosystem processes, and a locally orientated stakeholder understanding, can be at odds with each other and lead to different ideas about how ecosystem 'biomes' should be managed, and also which development projects are most desirable for the community and society at large. The major contribution of this study then is the attempt to determine the extent of this 'attitude gap' between scientists and local stakeholders by evaluating each groups opinions about ecosystem services and threats separately using BPA, and using this information to compare attitudes and beliefs related to ecosystems services and their management.

The results indicated that local stakeholders in the Iarras Aithneach peninsula placed greater value on services from biomes which contributed to community livelihoods such as coastal water (fishing), sand beaches (tourism), coastal islands (cultural value), peat bogs (turf-cutting) and agricultural land (farming). While these biomes also received a high valuation in the scientific consultations, it was due not only to the existence of provisioning services (which is what local stakeholders *and* scientists placed value on), but also to supporting and regulating services. The contribution of such ecosystem services to society are less direct, more complex and less obvious to observe without the necessary understanding and it is this disparity between scientists and local stakeholders in understanding environmental processes which creates ideological difference in how coastal areas should be managed. This disparity also caused differences of opinion in the sources and degree of risk which certain biomes, and in turn the services they provided, were subject to.

In the stakeholder consultations, risk was given far lower values than services from equivalent biomes, except in the case of the risk of over-regulation. Clearly then, stakeholders were reacting to what they viewed as excessive regulation of provisioning services, which in part can be explained by an insufficient understanding of the regulating and supporting services which protected areas helped to provide. During the scientific consultations on the other hand, the greater awareness of the participants of the supporting and regulating services of ecosystems, and the sensitivity of these services to human activities, led to far higher risk-values for various biomes.

Because of the attitude gap between both groups, two biome portfolios with entirely different risk return relationships arose from each consultation. The Portfolio Impact Sensitivity measure (PIS) indicates the degree of correlation across biomes in terms of sensitivity to threats; the higher it is, the more likely it is that a single or small number of threats could have a highly negative impact on the ecosystem services arising out of many of the biomes within the management area. A low PIS suggests it is more straightforward to make decisions that yield highest possible biome returns while "containing" risks. The PIS of the biome portfolio following stakeholder consultations was low. This suggests a BPA study of an area using only local participants could indicate to coastal managers that return of the biome portfolio is resilient to development of most of the major biomes, without systemic risks affecting other biomes in the portfolio. However the scientific consultations indicated that the biome portfolio had a high PIS value. The implication of this from a coastal management perspective, is that a decision affecting one biome, which may appear to pose no threat to other biomes in the area (based on the values attained through the stakeholder analysis), could affect the return from other biomes according to more scientifically informed points of view and analyses.

The final empirical paper in Chapter 5 sought to understand the potential impact of recent CFP reform on the fishing behaviour of fishermen in Irish waters. Specifically, it investigated how changes to EU quota levels affect the fishing location choice of Irish vessels in multispecies demersal otter trawl fisheries based in Irish waters. Vessel monitoring system (VMS) information and logbook and sale notes data were used to create a detailed representation of the spatial distribution of historical retained species catches and the corresponding fishing location choices made my fishermen. The core contribution of the paper was the use of VMS data, spatial mapping techniques and the RUM to make predictions about the impact of changes to quotas on the fishing location decision of fishermen. Overall, the results suggested that changes to fishing quotas lead not only (as was shown in Chapter 3) to changes in the species mix of retained catch, but also to significant changes in this fishing location choices of demersal otter trawlers.

In particular, the results indicated that the sizeable reduction in quota on Haddock and Whiting led to a general fall in the percentage share of fishing trips to key sites for these species. For predominantly nephrops orientated sites, despite a reduction in species quota, the model predicted an increase in the percentage share of trips. This latter result was likely due to the level of displacement which had arisen in Haddock and Whiting predominated species-areas. Finally in the Monkfish dominant areas, the model predicted that the quota changes would lead to an increase in fishing effort. This was not a surprising result given the fact that quota had been increased for Monkfish and such areas were also absorbing displaced effort from other regions

In general terms then, the three empirical studies carried out in this thesis provide a detailed analysis of stakeholder responsiveness to changes in environmental regulations, and the importance of stakeholders' attitudes and beliefs in their perception of ecosystem services and management initiatives. It must be stressed however that generalisations about ecosystems per se, and particularly marine and coastal ecosystems (which are highly heterogeneous across space), can lead to erroneous conclusions about the optimal management path for these resources. Rather it is superior to view the analysis and management of both ecological and social systems as ongoing processes requiring the constant application of case studies and specifically adapted methodologies.

The methods demonstrated in this thesis then, are not intended to act as methodological frameworks which can be applied routinely to any general coastal area or fishery, and from which general theories can be drawn, but as steps in the process of developing frameworks that can be adapted and applied on a case by case basis, through continuous testing and improvement. As the review of policy in Chapter 2 demonstrated, the nature of aquatic, coastal and marine management is intrinsically non-sectoral, non-homogenous and requires high levels of integration and adaptability across policy, research and management. Despite the requirement of research in this area to be case-specific and spatially orientated, all three empirical studies carried out in this thesis are consistent with the dominant economic and social theories to which they relate which is testimony to the robustness of the methods employed herein.

6.2. General limitations of the research in this thesis

Chapter 3 demonstrates how the portfolio methodology could be employed to accurately predict how a fleet might actually respond to various catch constraints. To be of real practical use however, further development would be necessary, particularly in terms of data availability. The major data limitations were the lack of data on costs and technical interactions; these factors play a large role in determining the profitability and feasibility of achieving particular combinations of aggregate catch in a multispecies fishery. For the methodology to be developed further into something that could actually assist in multispecies management decisions, such data would have to be included. The model employed in this paper minimised the consequences of omitting cost data by selecting fisheries with similar variable costs amongst species and little to no fixed cost fishery entry barriers

It is also the case that the measure of variance used in the study is the variance in annual total revenue over time which could be related to many time variant factors as opposed to randomly and independently over time. Furthermore, I consider the variance in revenues for the Irish fleet alone, which catch only a fraction of the total species harvest in each fishery. This is likely to compromise the accuracy of the model's predictive power somewhat since it is impossible to know how the Irish measures of variance apply to the wider international fleet in each fishery. Despite these weaknesses, there is scope for the portfolio approach to be improved upon by inclusion of catch data for the entire international fleet and perhaps the inclusion of more frequently observed species catch quantities and prices. Future improvement in the data collection process along these lines would resolve both of these two issues.

Chapter 4 applies BPA to an Irish coastal peninsula characterised by diverse habitat types, and uses this application to assess the degree of coherence in the principals of ICZM. In some respects, the two main objectives of the study (determining whether BPA was feasible as an ICZM tool and determining whether an 'attitude gap' exists in local and strategic viewpoints) drive at the same question. If no attitude gap tends to exist between scientists and stakeholders, that is to say, if there is not a disparate understanding of the natural processes driving ecosystems and their services, then BPA will act as a useful tool for ICZM since attitudes and beliefs about optimal management will be similar, and on a consensus basis, management decisions can be made. The fact that the results indicated a pronounced attitude gap between scientists and local stakeholders then, suggests incoherence in the stated principals of ICZM, and because of this, a methodology such as BPA, is unlikely to be able to determine consensus amongst various interest groups. This is not so much a flaw in the BPA methodology however, but in the incoherent nature of ICZM principals and a lack of conceptualization of how ICZM is to be applied in practice. There is also the scope to use BPA as a measure for the degree of disparity between strategic and local attitudes for any one area. It is recommended that BPA always be carried out by defining categorical groups such as local, strategic, stakeholder, relevant interest group etc, so that the risk-return values of each group for each biome can be compared and assessed, and the sources of biome value can be identified.

An additional point to note from the results presented in this Chapter is that apparently similar biome valuations from both the scientific and local survey participants acquired value from contradicting services, for example, conservation and peat extraction. While commonality in the value assigned to biomes' total return may indicate convergence (which is attractive from a management perspective since it suggests that strategic and local objectives are aligned), similar total biome return values may not reflect consensus at all. In fact, they can represent the very opposite; a completely opposite point of view on the value of the services delivered by the biome, and as a consequence, a completely opposite point of view on how that biome should be managed into the future.

The contribution of a service to biome portfolio return was also found to be heavily affected by the size of the biome it arose out of. Certain ecosystem services, for example carbon sequestration, need to arise out of large spatial areas to provide a meaningful service. In such a case, considering biome size is highly relevant when calculating which biome best provides that service. However, there are many cases where ecosystem services, in particular biodiversity and existence value, are of priority, yet are linked to relatively small biomes. It is not ideal that a methodology designed to evaluate risks and services relating to sensitive ecological areas should understate these risks and services when a biome is small relative to other biomes in the study area. If anything, small biomes are more responsive to threat factors and this should be represented in the methodology.

The final empirical paper of the thesis, Chapter 5, which applied the RUM to Irish demersal otter trawl VMS, catch and sales notes data indicated significant spatial displacement of fleet resulting from recent quota changes. While the model performed well across a number of measures of goodness of fit, the existence of either upward or downward bias in the model's predictor of site visitation probabilities is likely to be more pronounced if information relevant to the fishing location decision is omitted from the analysis. Where possible of course, all available information was used in the analysis, however measures of information such as skipper skill, or highly detailed information about area habitat and bottom type is limited. It is likely that the use of such measures in the RUM would reduce the bias of the model's estimator and improve the accuracy of the models predictions.

Other issues to bear in mind when considering the reliability of the model developed in Chapter 5 are the assumptions made in the analysis. For example, in my analysis I assume that all fishermen will remain in the fishery and respond to the quota changes by continuing to fish, reallocating effort according to an objective function based off of the random utility framework. However, it is possible that fishermen may also choose to exit the fishery. In this case, the model would over-predict the level of effort reallocation arising from the quota changes.

6.3. Future research arising from the findings in this thesis

Each methodology chapter in this thesis was unique, and therefore future research emanating from each chapter will also be largely unrelated. In terms of applying and adapting portfolio theory to fisheries management research, further multi-disciplinary work, such as collaboration with fisheries scientists/managers in the design and application of the portfolio methodology may lead to improvements in the model's usefulness. In addition, the comparison of species' weights resulting from alternative scenarios with the output of structural ecosystem models is another avenue for future research that could yield informative insights for fisheries management. Work involving the extrapolation of cost data from similar sized and comparable fisheries, and the creation of cost functions (as demonstrated by

Rockmann et al. 2009) for data-poor Irish fisheries would also improve the quality of a portfolio approach to fisheries management greatly.

ICZM is a form of coastal management which is very much in a development phase and methodologies which can assist in its application are in high demand. The BPA format addresses many of the requirements of an ICZM framework, in that it is spatially explicit, includes stakeholders in the policy design process, allows for trade-offs in costs and benefits and assists in qualitative decision-making with respect to alternative development projects. Due to the fact that coastal habitats and communities are highly heterogeneous across space, the development of BPA as a tool for ICZM would benefit greatly from further case studies in different areas along the EU coast. For example the BPA process is likely to be far more complex for more densely populated regions and an entirely different format of stakeholder consultation may be required. Only through further case studies and expansion of the methodology on a case by case basis will the potential of BPA as a tool for ICZM be fully understood.

One particular improvement that future work could address is the issue of scaling ecosystem service by biome size. One way of factoring heterogeneity in risk sensitivity (due to biome size differences) into the BPA methodology would be to structure area dependent risk elasticities into the methodology. In this case, the greater the supply of any biome, i.e. the larger it is, the less responsive it would be to risk exposure. Diminishing the size of any biome, or dealing with a biome which is in lesser supply than other biomes, would mean increasing its responsiveness to any threat in the modelling procedure. This adaption of the methodology would be in keeping with economic theory related to the increasing substitution price effect of any good as it diminishes in supply. Such an adaption may be justified on the basis that if a biome is decreased in size, its potential to absorb negative environmental impacts is reduced and the costs of damaging the biome's function (to supply ecosystem services) is likely to be higher, hence risks are higher. The key to structuring risk elasticity into the methodology would be to correctly associate various risk sensitivity levels to particular categories of biome size.

With respect to Chapter 5, a central theme to consider in weighing up the potential usefulness of the analytic methodology demonstrated therein is that of anticipating negative ecosystem outcomes arising from policy changes, and attempting to take regulatory actions to limit them. Specifically I refer to rather substantial problem of by-catch in demersal otter trawl fisheries (indeed many fisheries), and the fact that shifting fishing effort to alternative fishing sites may alter the degree of by-catch that takes places.

Due to the fact that most by-catch is currently 'discarded' and only the level of retained catch in an area is retained, data on the degree of by-catch that arises in various fishing areas is not recorded. Estimates of such by-catch would allow a 'change in by-catch' estimate to be produced for any given level of quota changes for various species. While these estimates do not exist, I strongly contend the importance of developing them and the usefulness of the approach demonstrated in the analysis once they exist. Even in their absence however, expert opinion from fishery managers and stakeholders could provide some tentative indications of changes in by-catch patterns following the quota changes given the changes to fishing site choice presented in this study. This is surely an area where future work would prove fruitful, given the improvements in monitoring technology that now exist as well as increased prominence of the by-catch and discards issue in both policy and public discourse.

Further research using the fisheries site choice model developed in Chapter 5 could also investigate the impact of designating one or more site choice options as marine protected areas effectively ruling them out as options in the choice set. Such area closure simulations have already been demonstrated in the literature but would be of use for regional based management decisions for Irish fisheries. There also appears to be scope to expand the analysis by including data from the approximate 300 vessels in the wider Irish trawl fleet (thereby including pelagic as well as demersal trawl vessels) as the VMS programme is now being expanded upon for Irish fleet. Adding cost data to the analysis would also improve the fit of the model as would survey information relating the attitudes of skippers from each vessel to each of the sites specified. Including habitat characteristics would also improve the accuracy of the model in reflecting changes to the fishing location choice. Encouragingly, there are signs that such data will be more available in the future through the advent of the Irish National Seabed survey and its continued development through the formation of INFOMAR, a project which is improving current habitat data through use of more advanced seabed mapping technology. In the event that area closures become a realistic management option for demersal fisheries, research investigating the opinions of skippers about which site closures are least restrictive would be useful for an integrated management approach.

6.4. Concluding Remarks and Recommendations

From the three separate empirical studies carried out in this thesis, a number of insights arose which permit several key conclusions to be drawn and a number of recommendations in to be made.

Turning first to the empirical analysis in Chapter 3, one of the results was that precautionary constraints in the Cod, Haddock and Whiting fishery could indirectly contribute to an over-capacity in the Hake, Monkfish and Megrim fishery due to displacement of traditional fishing effort. One of the ways in which fishery managers try to overcome the problem of over-capacity is to balance a fleet's harvest levels with stock capacity. Where negative multi-species/multi-fishery impacts of a protective measure are predicted, a similar structural rationalisation may be appropriate in the fisheries affected instead of relying on more precautionary quotas alone. The recent decommissioning of Irish whitefish fleet is a good example of this (Cawley et al. 2006). The fishery portfolio method may be of particular use when considering any measures (such as decommissioning) that can precede the undesirable displacement of fishing effort following a precautionary measure.

In a multispecies fisheries management context, the PT framework may also lend itself well to the task of setting multiple TACs across species. The ability to predict the direction into which fishing fleet will refocus targeting effort once quotas have shifted can help to inform policy makers about quotas required for other relevant species.

Rights-based measures may be warranted to ensure fleet capacity and harvest rates are maintained at sustainable levels. Indeed, the need for a portfolio approach to fisheries management to be combined with clearly defined harvesting/property rights and institutions has already been stressed by some (Hanna 1998; Edwards et al. 2004; 2005). In this analysis, it is much more likely that fishers would seek efficient risk-return outcomes if they acted as a single group maximising the value of the output in the entire fishery.

One issue limiting the usefulness of the portfolio approach is the lack of species specificity in fishing gears. Where a fishing strategy does not differentiate between two or more species, optimal fishing may not coincide with the species weights resulting from portfolio optimizations. Increased species specific fishing gears, which are becoming increasingly emphasised given EU policy developments on discard bans, will therefore improve the usefulness of the portfolio approach and contribute to more ecosystem-based and sustainable fishing practice.

With respect to ICZM in Ireland, the findings of the BPA study in Chapter 4 result in several recommendations for Ireland, and for the wider EU area. While the results indicated that local and strategic principals of ICZM were incoherent, and this fact seemingly undermines the usefulness of BPA in determining optimal management decisions through community consensus, there nevertheless emerged a very specific function which the BPA framework could provide. Specifically, rather than grouping all interest groups in the BPA consultations into one group and seeking to determine consensus, coastal managers can use the framework to evaluate the attitudes and beliefs of different groups to determine the disparities in opinion about ecosystem services, threats and optimal management. Such a process would inform coastal managers and policy makers on which types of provisioning services stakeholders are protective of, and potentially facilitate their expansion where strategically viable. Furthermore, BPA would also provide insight into which important supporting and regulating services were perhaps underappreciated in the local community, due to a lack of understanding of environmental processes. Such a BPA process could then act as a precursor to local conferences, scientific exhibits and education programmes which attempted to increase local understanding of complex ecosystem services arising from local habitat types.

As stressed earlier in the discursive section of Chapter 4, it seems clear that in order for EU coastal management and ICZM to have any real jurisdiction and become effective in a practical sense, clearly defined principals which are not contradictory must be developed. It seems apparent that this will need to take place through a prioritisation of the strategic principals, in a framework that will accommodate local objectives, educate shareholders on the nature of supporting and regulating services, and effectively direct strategic management goals for the coastal environment.

One idea put forward in Chapter 4 was that BPA could support a new direction for ICZM through a red light green light system, in which coastal managers would base management decisions on data from local participatory stakeholder groups *and* data from scientific consultations. Because stakeholder consultations are likely to result in a lower PIS value, any initiatives benefitting provisioning type services that benefit stakeholders would only be given the green light once they were validated according to the PIS values resulting from the scientific consultations. This would allow projects which had the capacity to improve the quality of local stakeholder livelihoods to go ahead once they did not trigger any ‘red lights’ in the more strategic and scientifically founded BPA results.

The final empirical paper of the thesis in Chapter 5 produced extensive results about the impact of quota changes on the fishing location choice of vessels in Irish demersal otter trawl fisheries. How fishery managers and marine ecologists involved in policy design and implementation respond to the results depends upon their understanding of the habitat structures and by-catch levels at each of the alternative sites. To elaborate, if the results indicate substantial displacement of the demersal trawl fleet into areas exhibiting a more coral based bed (as opposed to a less environmentally sensitive sandy bottom for example) further restrictions may need to be placed on the fleet to counteract the negative ecosystem impacts of such an increase in trawling effort. Furthermore, while estimates of by-catch resulting from demersal otter trawling do not exist, expert knowledge about concentrations of typical by-catch species in the various sites included in the study may be of use to fishery managers when deciding supportive policies that may help minimise any negative ecosystem impacts of fishermen's' behavioural response to quota changes. The point is that once the predictions are made about fleet displacement and reallocation of fishing effort, how this information is used must involve the advice of economists and ecologists as well as the real world practical working knowledge of fishery managers.

In a more general sense, the results of Chapter 5, being spatial in nature, can be of great use in amalgamating CFP design with that of the MSFD, MSP and HBD. A key issue in the future management of fish stocks in Irish and European waters, will be understanding the very impact of the changing policy and regulatory landscape on the behaviour of fishing fleet, and in turn, the impact these behavioural changes will impose upon the natural environment. Beyond the goal of fish stock and fishery sustainability which the CFP itself sets out, fisheries policy must also be coherent with other domains of environmental and marine policy which have as their underlying goal, good environmental and ecosystem status, such as the MSFD, the Birds and Habitat Directives, Maritime Spatial Planning and several other important Directives. Thus within the CFP framework and other relevant directives, the spatial component to management and stakeholder activity is critical.

7. References

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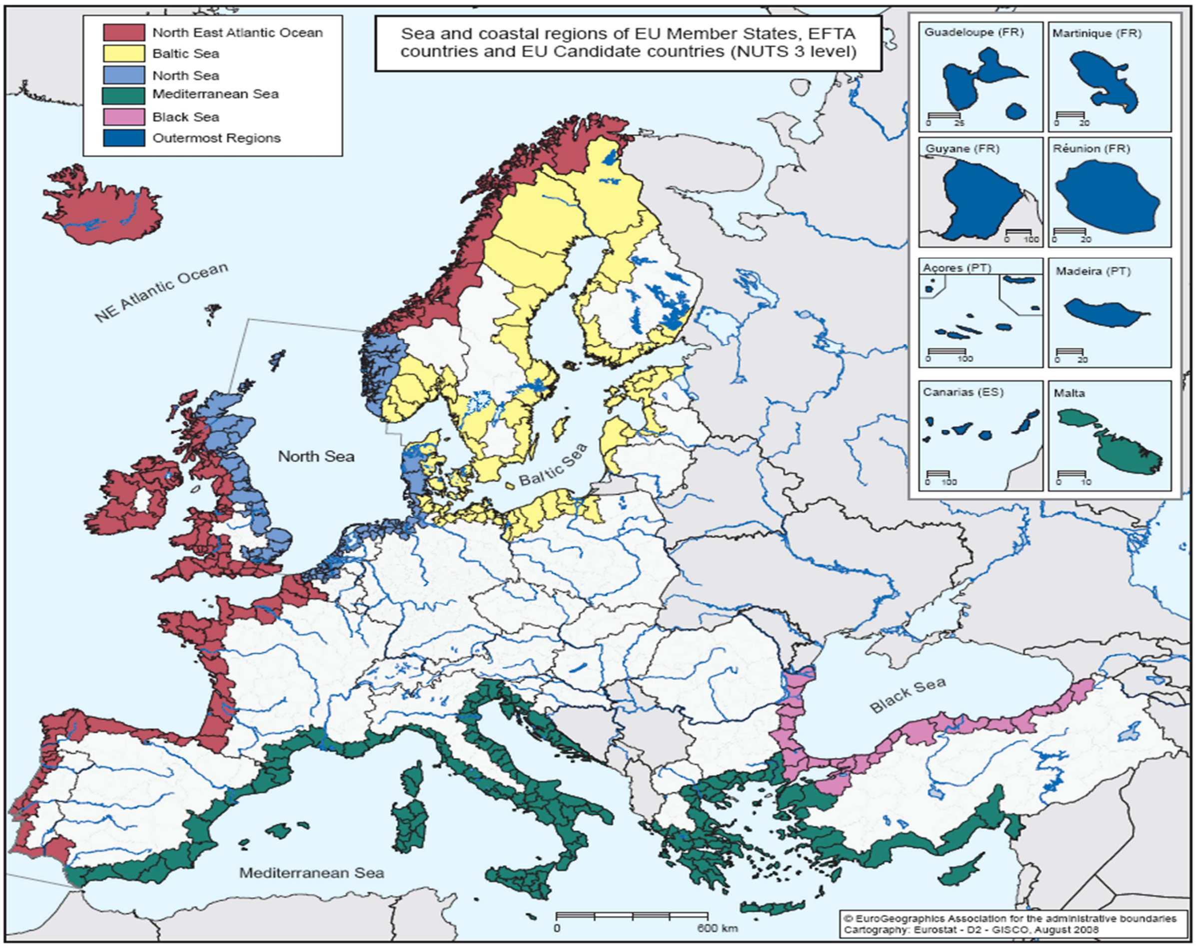
A. Appendix: EU and Irish coastal population statistics

Coastal regions in Europe are defined according to the NUTS level 3[24] statistical and geographical system of classification. Under this system, a coastal region of the EU is one that either has a coastline, or more than half of its population living less than 50 km from the sea. Until July 2013, the 27 EU Member States had 446 such regions, belonging to the 22 Member States which had a coastline (referred to as the EU-22) (Eurostat, 2009). Croatia recently became a member of this group making it the EU-23, but data on Croatian coastal statistics has not yet been published for inclusion in EU coastal statistics. In 2005 European coastal regions accounted for 43 % of the total EU-22 area and population (Eurostat, 2009). The EU-22 coastal regions are occupied by 199 million citizens resulting in an average population density of 100 inhabitants per km² compared to 108 over the entirety of EU-22 countries with a sea coast.

A very high population density (over 200 inhabit./km²) is registered in the coastal regions of Malta, Belgium, Netherlands, United Kingdom, Portugal and Italy. The highest density is observed in Malta with more than 1200 inhabitants per km². The lowest densities are found in Estonia, Sweden and Finland with less than 40 inhabitants per km² (see Figure A.1 for population density of each MS coastal area). The North Sea basin has by far the highest population density with on average nearly 250 inhabitants per km² while the Baltic Sea and the Outermost regions have only around 30 inhabitants per km². Figure A.1 defines coastal spatial area by sea basin type.

[24] The Nomenclature of Territorial Units for Statistics (NUTS) was established by Eurostat more than 30 years ago in order to provide a single uniform breakdown of territorial units for the production of regional statistics for the European Union

Figure A.1: Delimitation of Coastal Zones by Sea Basin



As well as the NUTS 3 definition of a coastal area, Hynes and Farrelly, (2011) develop two other definitions. A "Coastal Counties" definition includes any county that has a shoreline of any length adjacent to an ocean or sea, including estuaries and bays. In Ireland, 15 of the 26 counties in the Republic can be defined using this definition. The second, definition, "Shoreline Electoral Districts" (EDs) includes EDs immediately adjacent to the ocean or sea, including estuaries and bays. There are 3400 EDs in Ireland, of which 630 are on the coast.

Work carried out by SEMRU based on 2011 data estimated that NUTS 3 coastal areas in Ireland, the coastal counties and the shoreline ED's account for 94%, 78% and 27% of the national population respectively (SEMRU, 2014). The population density in coastal regions of Ireland is 69 inhabitants per square km at the EU NUTS 3 coastal area, 76 inhabitants per square km at the coastal county level and 94 inhabitants per square km at the shoreline ED level. The value of the coastal county economy for 2011 is estimated to be approximately €130.8 billion (84% of total GDP in Ireland) while the shoreline ED economy value is estimated to be approximately €44 billion. Table 2.1 and 2.2 show further socioeconomic data for different spatial definitions of coastal areas.

Table A.1: Socioeconomic statistics for Irish coastal EDs

	Shoreline ED	Shoreline ED Rural	Shore line Urban	National Level Average
Male Unemployment Rate (%)	22.47	23.47	19.90	21.71
Females Unemployment Rate (%)	14.28	14.44	13.89	13.86
Male Unemployment Rate (% change 2006 to 2011)	119.12	125.37	103.51	130.00
Females Unemployment Rate (% change 2006 to 2011)	200.12	225.95	133.79	266.67
% Primary Education Only	114.75	121.72	97.35	125.95
% 3rd Level Education	18.73	20.84	13.31	18.84
% Higher & Lower Professionals	29.77	26.22	38.87	25.88
Semi and unskilled Manual Workers	17.94	19.07	15.04	18.26
Population Change (% change 2006 to 2011)	6.29	6.99	4.49	7.79
Age Depending Ratio	35.05	36.13	32.28	34.94
Lone Parent Ratio	17.73	15.47	23.52	16.28
Affluence index score	-0.59	-2.21	3.57	-1.46
Affluence index score (% change 2006 to 2011)	0.75	0.45	1.54	-0.11
Number of Eds	638	459	179	3406

Table A.2: Socioeconomic statistics for Irish coastal EDs, coastal counties and EU NUTS 3 level

	Shoreline ED	Coastal County	EU Coastal
Male Unemployment Rate (%)	22.47	23.41	22.76
Females Unemployment Rate (%)	14.28	15.43	15.17
Male Unemployment Rate (% change 2006 to 2011)	119.12	91.98	92.93
Females Unemployment Rate (% change 2006 to 2011)	200.12	155.38	163.23
% Primary Education Only	114.75	86.88	87.86
% 3rd Level Education	18.73	16.18	16.30
% Higher & Lower Professionals	29.77	30.08	29.33
Semi and unskilled Manual Workers	17.94	18.03	17.97
Population Change (% change 2006 to 2011)	6.29	7.12	8.07
Age Depending Ratio	35.05	33.27	33.50
Lone Parent Ratio	17.73	22.40	20.74
Affluence index score	-0.59	-0.40	-0.36
Affluence index score (% change 2006 to 2011)	0.75	0.46	0.23
Regional units defined as coastal	638 out of 3406	14 out of 26	7 out of 8

B. Appendix: Survey used in Stakeholder and Scientific Consultations

The first part of the survey questionnaire is concerned with the ‘return’ that the selected biomes provide society in the form of various ecosystem services.

Table B.1: Biome returns survey

Service *(Indicate a value between 0-3)* *0=negligible ecosystem service provided* *3=extensive to complete ecosystem service provided*	**Biome Type** **Sea and ocean**	**Shallow water**	**Salt marsh**	**Saltwater Lagoons**	**Water bodies (inc. fresh lochs and rivers)**	**Sand Beaches**	**Coastal Islands**	**Coniferous forest**	**Agricultural land with areas of natural vegetation**	**Peat bogs**	**Other 1**
Agriculture											
Fishing											
Aquaculture											
Intertidal Gathering											
Sand/gravel/rock/peat extraction											
Conservation interest											
Recreation and tourism											
Cultural/educational											
Flood protection /coastal defence											
Nutrient/waste absorption											
Re. Energy generation											
Angling and shooting											
Landtake (carparks/range/causeways)											
Other 1											
Other 2											
Other 3											

The second part of the survey questionnaire is concerned with the risks that threaten the sustainability of the selected biomes and in particular, threaten the functioning of each biome with respect to specific ecosystem services:

Table B.2: Biome risk survey

Risk/threat *(Indicate a value between 0-3)* *0=threat factor has no impact* *3=threat could destroy the biome function*	Biome Type										
	Sea and ocean	Shallow water	Salt marsh	Saltwater Lagoons	Water bodies (inc. fresh lochs and rivers)	Sand Beaches	Coastal Islands	Coniferous forest	Agricultural land with areas of natural vegetation	Peat bogs	Other 1
Climate Change											
Erosion											
Flooding (inc. Sea level rise)											
Saline intrusion											
Tourism and recreation impact											
New causeways and other infrastructure											
Agricultural change											
Pollution inc. oil spills											
Invasive species											
Marine and terrestrial litter/dumping											
Overgathering of shellfish/overfishing											
Exhaustion through traditional usage											
Other 1 (Over-regulation)											
Other 2 (Salt damage)											
Other 3											

C. Appendix: Definition of Coastal Biomes

1. **Coastal Water:** The zone of maximum interaction between humans and critical biological resources; the intertidal zone to four meters below Mean Low Water.
2. **Sand open beach:** A beach is a geological landform along the shoreline of an ocean, sea, lake or river. It usually consists of loose particles which are often composed of rock, such as sand, gravel, shingle, pebbles or cobblestones. The particles of which the beach is composed can sometimes instead primarily be of biological origins, such as whole or fragmentary mollusc shells or fragments of coralline algae. In this study it also encompasses
   - 2.1. *Sand inlet:* A recess, such as a bay or cove made up of sand, along the coast)
   - 2.2. *Shingles:* A beach which is armoured with pebbles or small- to medium-sized cobbles. Typically, the stone composition may grade from characteristic sizes ranging from two to 200 mm
   - 2.3. *Rock Platform:* The ancient, stable, interior layer of a continental craton composed of igneous or metamorphic rocks covered by a thin layer of sedimentary rock. Rock platforms are flat, expansive eroded regions that lie at the base of rocky headlands. They are important habitats, as they contain a huge variety of plants and animals that cope with unique physical stresses of waves, fluctuating weather conditions and two complete tide cycles per day. Rock platforms are the most accessible of all marine habitats and an important resource for recreation and education
   - 2.4. *Sand Dunes:* a ridge of sand created by the wind; found in deserts or near lakes and oceans
3. **Coastal/Saltwater Lagoons:** Natural saline lagoons are areas of typically (but not exclusively) shallow coastal saline water, wholly or partially separated from the sea by

sandbanks, shingle rock or other barrier such as hard substrata. They retain some sea water at low tide and vary in salinity from slightly saltier than fresh water (brackish) to saltier than sea water (hyper-saline). Sea water exchange can occur through a natural or artificial channel or by percolation either through or over the barrier. More diffuse freshwater inputs (e.g. percolation, groundwater seepage) can affect the lagoon's salinity. Lagoons that are highly modified or are of artificial origin, such as those that occur behind a seawall, can still provide a similar habitat to that of natural lagoons, with a comparable range of specialised species.

4. **Water Bodies:** Natural or artificial stretches of water including rivers.
5. **Coastal Islands:** Any substantial land masses on the coast of Iarras Aithneach coast captured by the boundaries of the study site
6. **Coniferous Forest:** Vegetation formation composed principally of trees, including shrub and bush understories, where coniferous species predominate. Also included the categories:
7. **Agricultural land:** Areas principally occupied by agriculture, interspersed with significant natural areas. Also included the category:

    7.1. *Pastures*: Dense, predominantly graminoid grass cover, of floral composition, not under a rotation system. Mainly used for grazing, but the fodder may be harvested mechanically. Includes areas with hedges (bocage).

8. **Peat Bogs:** Peatland consisting mainly of decomposed moss and vegetable matter. May or may not be exploited.