RECONCILING SCIENCE AND POLITICS IN MARINE RESOURCE MANAGEMENT

Bethan Christine O'Leary

Thesis submitted for the Degree of

Doctor of Philosophy

University of York

Environment Department

February 2012

Abstract

Fishery resources are in principle renewable, but modern fisheries are typically characterised by excessive fishing effort, fleet overcapacity, illegal, unregulated and unreported (IUU) fishing along with deficient governance. This has led to growing trends of unsustainability. Fisheries management is a social and political process which aims to regulate human activities within the constraints of the biological ecosystem in which it operates. But the incorporation of sustainability into fishery practices around the world has to date generally failed. In this thesis I explore the relationship between science and politics in several different spheres of marine resource management.

Analysis of the extent to which European politicians have adhered to scientific recommendations on annual total allowable catches (TACs) from 1987 to 2011 for 11 stocks revealed that in 68% of decisions TACs were set higher than recommendations. Politically-adjusted TACs averaged 33-37% above scientifically advised levels. A simple stochastic model indicated that such politically-driven decision-making dramatically reduces stock sustainability. With 88% of European fish stocks overexploited relative to maximum sustainable yield targets, I conclude that political mismanagement must bear a considerable share of the responsibility for this decline.

Whilst the practice of political adjustment of scientific advice reveals the negative political impact on management and its failure to integrate science into management, the establishment of marine protected areas (MPAs) shows the relationship between science and politics in a more positive light. MPAs are increasingly being established to protect and rebuild coastal and marine ecosystems. However, the process of establishing these areas is not simple, particularly in areas beyond national jurisdiction (ABNJ) where few MPAs currently exist. Nevertheless, in 2010 the OSPAR Commission successfully established six MPAs forming the world's first network of MPAs in ABNJ. I summarise how this network was created, identify the main challenges, and offer a series of key lessons learned, highlighting approaches that may also be effective for similar efforts in the future. This success story was driven by strong political commitment and based on the best available science, and

serves as an example of the positive integration of science into management by politicians.

The difficult relationship between science and politics is illustrated clearly by the story of the Atlantic bluefin tuna (*Thunnus thynnus*). This species has become the quintessential example of overfishing and general mismanagement of the world's fisheries. An age-structured spatial model of the two stocks of Atlantic bluefin tuna highlighted the importance of taking area and stock movement into consideration when determining total allowable catches for the Atlantic bluefin tuna fisheries. The western bluefin stock was found to be more sensitive to assumptions of stock movement and mixing than the eastern populations, corroborating previous research. My results also indicated that to maximise the total catches of bluefin in perpetuity, it may be better to cease fishing in the western Atlantic and to only target individuals in the eastern Atlantic. The estimated timeframes for recovery are found to be medium to long term if fishing were halted today (within 20 years for both stocks to attain their B_{MSY}) and it is estimated that a 34% reduction in fishing mortality on both stocks is the minimum required decrease to ensure recovery. The aim of this model is to further research on the integration of science into a political management system in order to create a sustainable fishery.

In this thesis I identify several important requirements for sustainable fisheries management, namely: the need for a sound scientific basis, stakeholder engagement and cooperation, and strong political commitment and willingness.

List of Contents

List of Tables

List of Figures

List of Accompanying Material

Acknowledgements

Firstly, I would like to thank my two supervisors Prof. Callum Roberts and Dr. Jon Pitchford. Their wide knowledge, personal guidance and support have been invaluable for this work. I would also like to express my thanks to Dr. Jim Smart for his tireless patience and enthusiasm during my initial foray into the academic world.

I would like to acknowledge my funders, the Economic and Social Research Council (ESRC) and the National Environment Research Council (NERC).

This work is greatly indebted to my family, friends and the staff of the Environment Department who provided invaluable assistance, support, solidarity and inspiration. To Kathryn Addison, Claire Hughes, Maroula Hill and Dave Hay, thank you for your administrative and technical support as well as your friendly chats. To Rachel Brown and Ruth Thurstan, thank you for sharing the ups and downs of the PhD experience and for some great conference holidays! To Julie Hawkins and Bryce Beukers-Stewart, thank you for your initial training during my MSc and for your constant support and friendship since.

A special "thank you" goes to Simon Croft and Samik Datta for their mathematical and modelling expertise as well as their incredible patience.

To those family and friends outside of academia, thank you so much for your support throughout my PhD. For rejoicing with me, consoling me and giving me pep talks when needed I am eternally grateful.

In particular, many thanks to my parents, Alice and Mike O'Leary for your unconditional and undying belief in me, for undertaking the unenviable task of proof-reading my thesis and for becoming 'experts' in my field - I can't imagine you ever thought you would care so much about fish! This thesis is dedicated to you. My love and gratitude also goes to my sisters, Emma and Helen, as well as David Hunt for always being there and listening.

Finally to Dean. I could not have completed this research without your love, support, and inspiration. Thank you.

Author’s Declaration

Chapter 2 is based on a dataset originally examined by Fiona Neale, Julie Hawkins, Amy Milman and Stephanie Newman. A paper was published in Marine Pollution Bulletin based on the analysis described in Chapter 2 and the model developed in Chapter 3 entitled ‘Fisheries Mismanagement’ in which all co-authors have been identified (Appendix 2). The full reference of this paper is: O’Leary, B.C., Smart, J.C.R., Neale, F.C. Hawkins, J.P., Newman, S. Milman, A.C., Roberts, C.M., 2011. Fisheries Mismanagement. Mar. Pollut. Bull.. 62, 2642-2648. Appendix 3 reproduces a letter published in Nature based on the work presented in chapters 2 and 3 of this thesis and published in response to Froese (2011). The reference of this article is: O’Leary, B.C., Roberts, C.M., 2011. Fishery reform: ban political haggling. Nature. 475: 454 (Appendix 3).

The work contributing to Chapter 4 was completed as part of a project with two members of the University of York Marine Research Team, Rachel Brown and Callum Roberts, and taken forward by the OSPAR Convention under the guidance of four key persons: David Johnson, Henning von Nordheim, Jeff Ardron and Tim Packeiser. Chapter 4 constitutes a paper published in Marine Policy written and coordinated by myself which has received comments from all these persons. The full reference of this paper is: O’Leary, B.C., Brown, R.L., Johnson, D.E., von Nordheim, H., Ardron, J., Packeiser, T., Roberts, C.M., 2012. The first network of marine protected areas (MPAs) in the high seas: the process, the challenges, and where next. Mar. Policy. 36, 598-605.

I hereby declare that all data analysis and writing of this thesis is my own work and any previously published material included here has been acknowledged as such.

Bethan Christine O’Leary

Chapter 1.

Introduction and Main Concepts

1.1. Introduction

"The future of everything we have accomplished since our intelligence evolved will depend on the wisdom of our actions over the next few years" (Wright 2004)

Fishery resources are renewable but limited by the environmental capacity of the ecosystem they inhabit. Fisheries are based on the extraction of wildlife and as a result fishing pressure has to be controlled within the limits of the ecosystem. If fishing effort rises above sustainable levels it will at best increase to the point of economic unprofitability. At worst it may result in a total stock collapse and drive ecosystem phase shifts. Modern fisheries are characterised by overcapacity of fishing fleets and the current scale of exploitation is considered to be far too intensive to be sustainable in many fisheries (Villasante and Sumaila 2010; Standal and Utne 2011). Despite large economic and social investments, without marine living resources the fishing sector would cease to exist. Therefore to disregard the state of marine ecosystems and organisms when investing and legislating on fishing and aquaculture would be very short-sighted.

The declining trends seen in world fisheries have become a major cause for concern. Much evidence now indicates the overexploitation of the global oceans and continuous absolute and relative increases in collapsed stocks are being predicted (Froese and Kesner-Reyes 2002; Worm *et al.* 2006; Froese *et al.* 2009). Currently 28% of all fish stocks monitored are considered to be overexploited, depleted (3%) or recovering from depletion (1%) while 53% are fully exploited with no scope for further expansion (FAO 2010b). It has been estimated that the global reservoir of unexploited fishable stocks is likely to be exhausted by 2020 (Froese *et al.* 2009). The FAO[1] (2010b) considers that the maximum potential for wild capture fisheries has probably now been reached and is advocating stronger management to ensure ecosystems are not degraded further.

With millions of people around the world depending on fisheries for their livelihoods and as a major protein source (FAO 2010b) the potential collapse of fish stocks and the continuing degradation of the marine environment would have far-reaching, and devastating, consequences. The growing evidence of the impacts of fishing has led to

[1] Food and Agriculture Organisation

the conclusion that many marine ecosystems are overfished and improvements in management are essential (Sutinen and Soboil 2001). As part of the reform of management holistic approaches incorporating industry, conservation and other stakeholders of the oceans are necessary to rebuild stocks. In this introduction I will highlight the impacts of fishing on the marine environment, discuss current fisheries management practices and introduce the concepts of this research thesis.

1.2. Impacts of fishing on the marine environment

Modern day commercial fishing has been recognised as a leading environmental and socio-economic problem, having far-reaching impacts in the marine realm (Jackson *et al.* 2001; Lotze *et al.* 2006; Worm *et al.* 2006; Daskalov 2008; Worm *et al.* 2009).

The most straightforward effect that fishing has on exploited populations is to directly reduce the abundance of targeted species through harvesting. In addition however, exploited species have been shown to exhibit a higher temporal variability in abundance than unexploited species (Hsieh *et al.* 2006; Anderson *et al.* 2008). Fisheries selectively remove large and old individuals which results in a truncation of age-structure and this is increases population variability. Older and larger fish produce an increased quantity and quality of eggs than smaller and younger individuals (Berkeley *et al.* 2004). Consequently, selectively removing these individuals acts to reduce the reproductive capability of the population. In addition, so as to increase the survival rate of larvae under variable environmental conditions many fish species use bet-hedging strategies. For example, some species exhibit age-related difference in spawning localities and time (Hsieh *et al.* 2006). When fishing undermines these bet-hedging strategies populations lose their ability to dampen environmental stochasticity and populations more closely track short-term environmental variability (Anderson *et al.* 2008). The biomass of species is also affected by fishing. Global large predatory fish biomass is considered to be only ~10% of pre-industrial levels (Myers and Worm 2003, 2005).

Comparisons of fished and unfished areas have consistently shown that diversity and biomass of marine organisms are higher in the unfished areas (Koslow *et al.* 2001; Halpern 2003; Hiddink *et al.* 2006). As well as local diversity effects, predator diversity has declined between 10% and 50% in all oceans over the past 50 years, coinciding with increases in fishing pressure (Worm *et al.* 2005). Biodiversity has been linked to ecosystem resilience (Worm *et al.* 2006) and concern has been raised regarding the loss of some species and the stability of an ecosystem (Hooper *et al.* 2005). How much the loss of a species affects an ecosystem will depend on its functional role; whilst some may be lost without any great change within the ecosystem others may cause ecosystem structure to shift (Roberts 1995). The loss of species diversity as a result of fishing may lead to a reduction in ecological integrity promoting phase shifts to another, perhaps less desirable state (e.g. Steneck *et al.* 2002).

In addition, fishing techniques have been shown to impact the structure of ecosystems. Fishing preferentially targets large, slow-growing species which are the most vulnerable to fishing. As traditional stocks decline fishers move to smaller, previously less favoured species (Kaiser *et al.* 2005), a pattern that has been termed 'fishing down the food web' (Pauly *et al.* 2002). Within global landings there has been a decline of 0.05-0.10 trophic levels[2] per decade indicating the slow removal of large, long-lived fishes from the oceans (Pauly *et al.* 2002). The result is a change to the ecosystem structure which allows the dominance of previously suppressed species which may, or may not, be exploited by humans (Pauly *et al.* 2002). It may also have the effect of driving the evolution of a species to smaller, faster maturing individuals (Caddy and Garibaldi 2000; Grift *et al.* 2003; Sharpe and Hendry 2009).

Reducing the biomass of large older individuals through fishing also has consequences for the reproductive output of the stock, as this varies with size and age. The fecundity (number of eggs per unit of body mass) of fishes increases exponentially with body size and evidence indicates that the eggs of large, older females are often larger and contain more oil than smaller females leading to 60-80% better survival rates (Berkeley *et al.* 2004; Scott *et al.* 2006; Field *et al.* 2008). The removal of large organisms from size spectrum[3] has been shown to shift the steady state of marine ecosystems from stability to instability (Plank and Law 2011). This follows from observations that it may be deleterious to remove the Big, Old, Fat,

[2] Trophic levels describe the position a fish or invertebrate occupies within the ecosystem. It expresses the number of steps they are removed from algae (which occupies a trophic level of 1). Most food fishes have a trophic level ranging from 3.0 - 4.5 (Pauly *et al.* 2002).

[3] Size spectrum refers to the frequency distribution of all individuals across the spectrum of body mass, irrespective of their taxonomic identity (Datta et al. 2010).

Female Fish (BOFFF) and changes in size and age structures of marine species resulting from fishing may have a profound influence on the potential for future generations (Law 2007). In fact, Rochet and Benoît (2012) have suggested that selective fishing also destabilises community trophic dynamics. There is a theory therefore that rather than simply reducing fishing pressure exploitation should be balanced across trophic levels to maintain ecosystem trophic structure (Law *et al*. 2012, Rochet and Benoît 2012, Bundy *et al.* 2005).

In terms of marine extinctions, few have been documented in the past due to the difficulty of detection (Dulvy *et al.* 2003). In addition, up until the late nineteenth and early twentieth century the belief that the seas were inexhaustible was commonly held (Pauly *et al.* 2003), limiting investigation into marine extinctions. However, evidence suggests that marine extinctions are likely to have been underestimated and are becoming more common (Roberts and Hawkins 1999; Dulvy *et al.* 2003; Hutchings and Reynolds 2004). There is also much evidence of local and regional extirpations indicating the reduction of range for many species (e.g. MacKenzie and Myers 2007; Lotze and Worm 2008; Robinson and Frid 2008). Commercially important species may be fished down to a vulnerable level because of their economic value and non-targeted species may be threatened through bycatch (Cheung *et al.* 2005). The direct (e.g. removal of biomass) and indirect (e.g. habitat destruction) effects of fishing may therefore be placing marine species under greater threat of extirpation and extinction.

On land, habitat loss and fragmentation is the most frequently cited factor in the increasing rate of species becoming threatened or extinct (Pimm *et al.* 1995; Brooks *et al.* 2002). The destruction of marine habitats through fishing and other activities such as coastal habitat destruction and conversion are therefore also likely to contribute reduced marine abundance or diversity. Many fishing techniques have the capability of altering, removing or destroying the complex, three-dimensional physical structure on the seabed through the capture of attached species such as cold-water corals or seafans (Harrington *et al.* 2005; Biju Kumar and Deepthi 2006). This directly removes hard substrata and disturbs ecological communities that have developed sometimes over thousands of years (Roberts 2002). The disturbance to these habitats may affect structural and functional biodiversity and community composition (Biju Kumar and Deepthi 2006).

To a large degree the abundance of an organism is a function of the quality and quantity of suitable habitat available to it either directly, through habitat niches or indirectly, through the quality and quantity of prey. The degradation of benthic habitats, both directly (e.g. through the process of trawling) and indirectly (e.g. siltation) can have negative consequences for fish yield by causing a redistribution of species and by reducing the potential production of that ecosystem (Turner *et al.* 1999).

1.3. Fisheries management

The overall aim of fishery management is to facilitate sustainable fishery systems, i.e. to ensure that conditions necessary for marine resource renewal are provided. Fisheries management is a social and political process that is constrained by the biological systems in which it operates and to date fisheries management has generally failed to incorporate sustainability into fisheries around the world (Sutinen and Soboil 2001). The next section provides a brief overview of the major features and problems of fisheries management.

Single-species tools in a multispecies world

The invention of modern fisheries science is often attributed to Beverton and Holt (1957), who formalised theory of exploitation for different fish life histories and brought together existing ideas about Maximum Sustainable Yield (MSY) and surplus production. MSY is a yield estimation based on the ‘surplus yield’ model where the objective is to take the maximum possible catch that can be maintained under favourable conditions (Hilborn and Walters 1992; Mace 2001). In theory, catching fish can increase the productivity of a stock as at low population densities there is an exponential growth in population size until resources becoming limiting and growth slows, eventually to zero at the carrying capacity of the environment. As fishing effort is applied, the biomass of the population decreases and growth rate therefore increases due to reduced population density and competition for resources. The maximum growth rates occur at intermediate population sizes as at low population densities growth rate may become negative due to Allee effects

(Liermann and Hilborn 2001; Jensen *et al.* 2012). MSY is formed under the assumption that when a population is at half of its unexploited biomass (B_{MSY}) its population growth rates are highest, and consequently it represents the quantity of stock that can be removed in perpetuity without the stock declining or collapsing (Mace 2001). The concept of MSY therefore leads to a conflict between maximisation and sustainability – the higher the harvest rate is set the more fragile the sustainability leaving no margin for error. Should environmental variation and stochastic events occur while the population is at half of its unexploited biomass the stock can quickly be driven towards collapse. Typically, within management B_{MSY} acts as a trigger reference point below which catches are systematically reduced to reach zero at a limit biomass (B_{LIM}) (Froese *et al.* 2011). This is the point at which the reproductive capacity of the stock is endangered. Precautionary target biomasses that are larger than that which produces MSY may therefore be applied in accordance with Annex II of the UN Fish Stocks Agreement (1995)[4].

As a result of the MSY concept, fishery management to date has largely been conducted on a single-species approach aiming to maintain fisheries production and target stocks using a variety of controls on fishing effort and catch (Sutinen and Soboil 2001). Types of regulatory measures include temporal and spatial restrictions on catch or fishing effort, annually adjusted quotas in the form of total allowable catches (TACs), technical restrictions and minimum landing sizes of species (Holland 2003). These measures often result in high grading of target species and the discarding of undersized or over-quota commercially important species, thereby increasing the fishing mortality being imparted on the system (Kristofersson and Rickertsen 2009; Poos *et al.* 2010; Bellido *et al.* 2011).

The majority of fisheries actually operate in a multispecies environment. As species have different intrinsic abilities to accommodate fishing mortality their individual MSY curves are very different. Consequently, there will be a spectrum of responses to fishing within the ecosystem (Kaufman *et al.* 2004; Pinnegar *et al.* 2005). If fishing effort is matched to reach the MSY for the most resilient species the other species (often caught using the same gear) may become overexploited or very overexploited. The least resilient species is known as the 'weakest link' (Pinnegar *et*

[4] Annex II of the United Nations Fish Stocks Agreement (1995) specifies that the fishing mortality which generates MSY should be regarded as the minimum standard for limit reference points.

al. 2005) as, if effort was only matched to this species, then catch of other species would decrease significantly (Kaufman *et al.* 2004).

Traditional management, therefore, often fails to consider the indirect effects of fisheries on the whole ecosystem (e.g. habitat, food supply, non-target species), at least until there is a problem with that fishery (Walters 1998; Sutinen and Soboil 2001; Lindenmayer *et al.* 2007). It has been argued that had science advisors and managers considered more of the ecosystem in decision-making, and tried to minimise the impact of politics on the fishing industry, then mistakes might have been foreseen and avoided (Cardinale and Svedäng 2008; Rice 2008).

The recognition that multispecies considerations need to be incorporated into fishery management is now widespread and is central to the concept of ecosystem-based fishery management (EBFM) (McLeod *et al.* 2005). Despite ecologists frequently stating the importance of modelling ecosystems rather than individual species, the majority of models simulate a single-species world due to the increased complications of modelling multiple dynamic trophic levels (Fleming and Alexander 2003; Armstrong 2007). Whilst multi-species modelling appears to offer greater insight into ecosystems they come with a multitude of problems (Hollowed *et al.* 2000). As the models become more detailed and complex there is a greater potential for serious issues of confounding to appear, there is an increased reliance on statistical (often deficient) data, and the inclusion of multiple potential confounding variables often leads to considerable obfuscation of the results. In practice therefore, single species models are still the dominant tool worldwide for providing timely and reliable scientific advice regarding the management of commercially valuable stocks.

Data deficiency and uncertainty

Data are often deficient within fisheries science and management and estimates of biomass are full of inaccuracies (Walters 1998; Chen *et al.* 2003; Kraak *et al.* 2010). Difficulties in estimating the abundance of fish during scientific assessments from survey and fisheries data, together with the application of these data into simple stock-assessment models will always result in some uncertainty. As Hilborn and Walters (1992) maintained, "you cannot determine the potential yield from a fish stock without overexploiting it". Fisheries management tools require biological information and due to uncertainty these tools are often not used effectively with, for example, excessively high TACs being set by fisheries managers (Walters 1998; Cardinale and Svedäng 2008).

Establishing management strategies such as TACs is often difficult as a result of the inherent uncertainty within science (Kraak *et al.* 2010; Hauge 2011). This uncertainty stems from difficulties in data collection, a lack of knowledge regarding stock-recruitment relationships and connectivity between stocks as well as variable environmental conditions and anthropogenic stresses. These uncertainties are great for those stocks and environments within national exclusive economic zones but are even greater for areas outside of national jurisdiction (the high seas) and are compounded by political difficulties.

The establishment of marine protected areas (MPAs) has been suggested as a way to reduce uncertainty and act as a buffer against unfavourable environmental conditions and poor management (Lauck *et al.* 1998; Grafton and Kompas 2005). A MPA may be defined as “Any area of intertidal or sub-tidal terrain, together with its overlying water and associated flora, fauna, historical and cultural features, which has been reserved by law or other effective means to protect part of all of the enclosed environment” (Kelleher 1999). However, while there are global targets to establish MPAs[5] in practice it is still found that politicians want a strong scientific underpinning in order to designate areas for protection, at least in the high seas (O'Leary *et al.* 2012).

Political adjustment and harvest control rules

Today, MSY provides a reference against which exploitation can be measured (using the fishing mortality and biomass at which MSY (B_{MSY}) is achieved providing a limit reference point to managers (Punt and Smith 2001). Often, as in the European Union (EU), total allowable catches (TACs) are used to control fisheries landings and are

[5] These include commitments to establish representative networks of MPAs by 2012 at the World Summit on Sustainable Development (WSSD) in 2002, and subsequent United Nations General Assembly (UNGA) resolutions and Convention on Biological Diversity (CBD) decisions. In particular the latest target by the CBD is that "By 2020, at least 17 per cent of terrestrial and inland water, and 10 per cent of coastal and marine areas, especially areas of particular importance for biodiversity and ecosystem services, are conserved through effectively and equitably managed, ecologically representative and well-connected systems of protected areas and other effective area-based conservation measures, and integrated into the wider landscapes and seascapes" (target 11, http://www.cbd.int/sp/targets/).

set according to the perceived stock size and the estimated harvest rate (Punt and Smith 2001; Pitchford *et al.* 2007). TACs are often recommended by scientists and are then adjusted and agreed by politicians and managers either on an ad hoc basis or according to harvest control rules (HCRs).

HCRs were developed in order to minimise ad hoc political decisions and to develop the precautionary approach into feasible fisheries management by the scientific working groups of the International Council for the Exploration of the Seas (ICES) and the Northwest Atlantic Fisheries Organisation (NAFO) among others (Serchuk *et al.* 1997; NAFO 2002; ICES 2008). HCRs are sets of well-defined rules that can be used for determining quotas or fishing effort developed according to the status of the resource being managed and the implementation of reference points based on fishing mortality and biomass (Mace 2001; Apostolaki and Hillary 2009). Traditional harvesting strategies are simple HCRs which use only one parameter, i.e. constant harvest rate (del Valle and Astorkiza 2007). The use of increasingly complex, or multi-parameter, HCRs are now being suggested and used by scientists (del Valle and Astorkiza 2007; ICES 2008). HCRs may reduce uncertainty as, if the management policy is expressed as a HCR then it allows the TAC to be determined unambiguously with no "wiggle room" for interpretation by politicians and managers (ICES 2008).

Within European fisheries, politically influenced 'quota-bargaining' is often seen (Daw and Gray 2005; Roberts *et al.* 2005; Pitchford *et al.* 2007) although HCRs are used for some species such as the Northeast Arctic Haddock (Apostolaki and Hillary 2009). Competitive quota-bargaining in Europe often results in TACs being set unrealistically high by politicians (Roberts *et al.* 2005; Cardinale and Svedäng 2008). For example, previous work has estimated that in Europe fishery ministers usually set TACs 15-30% higher than recommended by fisheries scientists (Roberts *et al.* 2005). Further overshooting of TACs through illegal landings, legally permitted overshoot and discarding then occurs which acts to compound the problems of excessive catches (Cardinale and Svedäng 2008). This leaves biomass vulnerable to be driven down towards collapse. In addition, when combined with environmental variation, one or more years of poor environmental conditions combined with continuing effort might bring the population below the biological replenishment level and collapse may then be inevitable (Grafton *et al.* 2005). The inability of scientists to make confident predictions of collapse is often regarded as a justification for taking large harvests and even for increasing harvests (FAO 1996; Punt and Smith 2001). For example, Beverton and Holt identified exploitation rates on many North Sea stocks to be unsustainable as early as 1957 (Beverton and Holt 1957) and yet these high rates have persisted, despite repeated scientific advice that lower exploitation rates would lead to higher yields (Karagiannakos 1996).

Economic discounting

According to Clark (1973a, b, 1990) it is 'economically rational' (the maximisation of discounted net returns) to exploit populations to the point of extinction as a result of discounting (i.e. where future returns are weighted less heavily than present returns, and often the discount factors are small corresponding to a short time horizon). In fact, within economic analyses the discount rate influences the optimal harvesting strategy dramatically and often leads to the collapse or extinction of exploited populations (e.g. Clark 1990, Sethi and Thompson 2000).

To overcome the problems of a fixed discount rate, it has been suggested that the discount rate should decline over time, known as hyperbolic discounting, to protect natural resources (e.g. Cropper and Laibson 1999, Shogren and Settle 2004, Voinov and Farley 2007). In this case the planner would reduce stock levels early on when the discount rate is high and intend to compensate for this by allowing the stock to recover when the discount rate is lower. This acts to increase the weight on benefits in the distant future and may provide greater protection of resources for future generations. Duncan *et al.* (2011) applied hyperbolic discounting to the Peruvian anchovy fishery in the 1970's. They showed that while this can lead to a sustainable fishery, if the planner ends up repeatedly restarting the optimisation this drives the stock down to the point where it becomes optimal to harvest the stock to extinction. This process of re-optimising policy is thought to have contributed to the collapse of the North Atlantic cod (Duncan *et al.* 2011).

The concept of applying economic discounting to life support systems, of which natural resources (and biodiversity) are included, has been criticised, largely on the controversy of the balance between present and future, i.e. intergenerational equity and efficient use of capital (Heal 1997). However, investigations into the effect of discounting shows, and at least partly explains, the dire consequences of ‘business as usual’ in the exploitation of natural resources (Clark 1973a, b, 1990, Duncan *et al*. 2011).

1.4. New directions for marine resource management

The recognition of the ineffectiveness of past and present fisheries management to impart sustainability into marine ecosystems and the fisheries they support is well recognised. Sustainability has been an aim of most governments since the World Summit on Sustainable Development in 2002 where a target was set to restore all fish stocks to their MSY levels by 2015 (UN 2002). It has been estimated that global fisheries contribute at least $50 billion less to the global economy than they would if stocks were returned to their MSY (Arnason *et al.* 2009; Holt 2009). While some stocks are showing signs of recovery after improved management efforts (e.g. Northeast Arctic stocks, Diamond and Beukers-Stewart 2011) it is also increasingly recognised that impacts from climate change, biodiversity loss, pollution, coastal development and habitat loss and fragmentation compound the problems caused by overexploitation and may affect stock recovery. The unprecedented challenges facing sustainable management of the oceans require scientists, practitioners and citizens to embrace a broader vision for marine management encompassing environmental and socio-economic well-being.

Interest in ecosystem-based fisheries management (EBFM) has therefore increased with the aim to incorporate sustainability into fishing activities and development (Cardinale and Svedäng 2008). Early studies of the effects of fishing, and traditional management techniques were founded on short-term dynamics of target fish populations, considered independently of the ecosystem as a whole (Sutinen and Soboil 2001; García *et al.* 2003). The aim of EBFM is to recognise the interconnectedness within and between systems, while integrating ecological, social and economic positions (McLeod *et al.* 2005). Whilst management policies considering all ecosystem components would be extremely data intensive and as such unrealistic, EBFM is considered to be a management tool that needs to be supported by, and based on, the best scientific advice available. EBFM therefore can be used to make an informed decision while invoking the precautionary approach.

EBFM places an ecological priority on fisheries management as this is thought to be essential for the long-term socio-economic sustainability of the fishing industry. It is essentially a new direction in marine resource management where the order of management priorities is reversed to start with the ecosystem rather than the target species. However, it is recognised that an ecological priority will likely clash with the needs of economic and social objectives in the short-term (e.g. CEC 2009; Standal and Utne 2011).

It is likely that the implementation of holistic management under the guise of EBFM will employ a number of tools that emphasise both protection and use and which incorporate the precautionary approach. These tools might include those aimed at creating incentives for stewardship and collaboration (e.g. individual transferable quotas) and area-based management tools (e.g. marine protected areas and ocean zoning).

Accounting for climate change

Based upon current scientific evidence, emissions of greenhouse gases from human activities are projected to cause significant global climate change during the 21st Century (IPCC 2007). This is likely to create novel challenges for marine ecosystems and resource management. Ocean temperature changes will influence organism metabolism, alter ecological processes (e.g. productivity), and expand or contract species' geographical distributions (e.g. Perry *et al.* 2005; Pörtner and Knust 2007; Brander 2010). Increased carbon dioxide concentrations lower pH, which will alter ocean carbonate chemistry (Doney *et al.* 2009). Changes in precipitation and sea-level rise will have consequences for surface runoff and coastal ecosystem (e.g. mangroves) flooding (Hoegh-Guldberg and Bruno 2011). Patterns of wind and water circulation are also likely to change, influencing upwelling, the transfer of nutrients and oxygen, and ocean temperatures (Hoegh-Guldberg and Bruno 2011). Future management and conservation of the marine environment and fisheries will therefore need to incorporate plans to adapt to climate change (Heller and Zavaleta 2009). Broadly speaking, such plans will require improved regional planning and coordination, expanded spatial and temporal perspectives, incorporation of climate change scenarios into all planning, and greater coordination between policy and management institutions to address multiple threats (Heller and Zavaleta 2009; Link *et al.* 2011).

1.5. Organisation of the thesis

The problems faced by fishers and the marine environment alike, i.e. declining catches and degraded ecosystems, have led to the re-evaluation of fisheries management. In order to help stocks recover, rebuild lost ecosystem resilience and ultimately improve the fishing industry, fisheries management needs to evolve - incorporating science-based management with an ecological focus so as to maintain human well-being into the future.

The aim of this thesis was to explore relationships between science and politics in ocean management using statistical analysis, simulation modelling and practical experience. I examine the impact of political decision-making on European fisheries together with the scientific and political process of establishing marine protected areas as one conservation tool within the high seas. I then develop a model that will allow investigation into the sustainable management of the Atlantic bluefin tuna (*Thunnus thynnus*) and ways of bringing together science and politics in this situation.

The following chapters comprise the analytical part of this research. I first present an historical overview and analysis regarding the role of politicians in fisheries regulation, specifically the annual setting of TACs. I then proceed to further this investigation through the development of a deterministic model in order to untangle the role of political adjustment to TACs from additional influencing factors such as discarding and bycatch. I then discuss scientific and political difficulties of establishing marine protected areas (MPAs) in data deficient areas such as the high seas. For this I use OSPAR and the North-East Atlantic as a case study, where the first network of high seas MPAs was established in 2010. Continuing the theme of the high seas, I then develop a spatial model for the high seas straddling stocks of the Atlantic bluefin tuna. This model allows analysis of the impact that political adjustment has had on the status of both the western and eastern stocks as well as contributing to current discussions regarding the application of marine reserves for migratory and far-ranging species. The final part of my thesis brings together the results and discusses the process of bringing together science and politics in marine resource management.

Chapter 2.

The Impact of Political Adjustment of Total Allowable Catches in European Fisheries Management

O'Leary, B.C., Smart, J.C.R., Neale, F.C. Hawkins, J.P., Newman, S. Milman, A.C., Roberts, C.M., 2011. Fisheries Mismanagement. Mar. Pollut. Bull.. 62, 2642-2648.

2.1. Abstract

In this chapter I analyse the extent to which European politicians have adhered to scientific recommendations on annual total allowable catches (TACs) from 1987 to 2011, covering most of the period of the Common Fisheries Policy (CFP). For the 11 stocks examined, I find that TACs were set higher than scientific recommendations in 68% of decisions. These politically-adjusted TACs averaged 33-37% above scientifically recommended levels. In addition, I find no evidence that the 2002 reform of the CFP improved decision-making, as was claimed at the time. For the stocks examined, scientific recommendations advising zero-catch (moratorium) were not followed on any occasion; a TAC was always implemented in contradiction to the scientific advice. The management zone most prone to political adjustment was the Spanish, Portuguese and Bay of Biscay zone (division VIII/IX). The waters around Iceland (division Va) were the least prone to political adjustment.

I find that political decisions to overrule scientific advice are endemic within the fisheries decision-making process in Europe. With increasing numbers of moratoria being advised, this implies that current decision-making practices and policies are failing to impart sustainability into fish stocks. I argue that the annual negotiation and setting of TACs needs to be changed, moving away from political debates between the European Commission and national fisheries ministers. Instead, I advocate that long-term sustainability should be prioritised. When setting TACs scientific advice should be followed and adherence to scientific recommendations should be legally binding to remove the temptation to set TACs higher than the ecosystem can support.

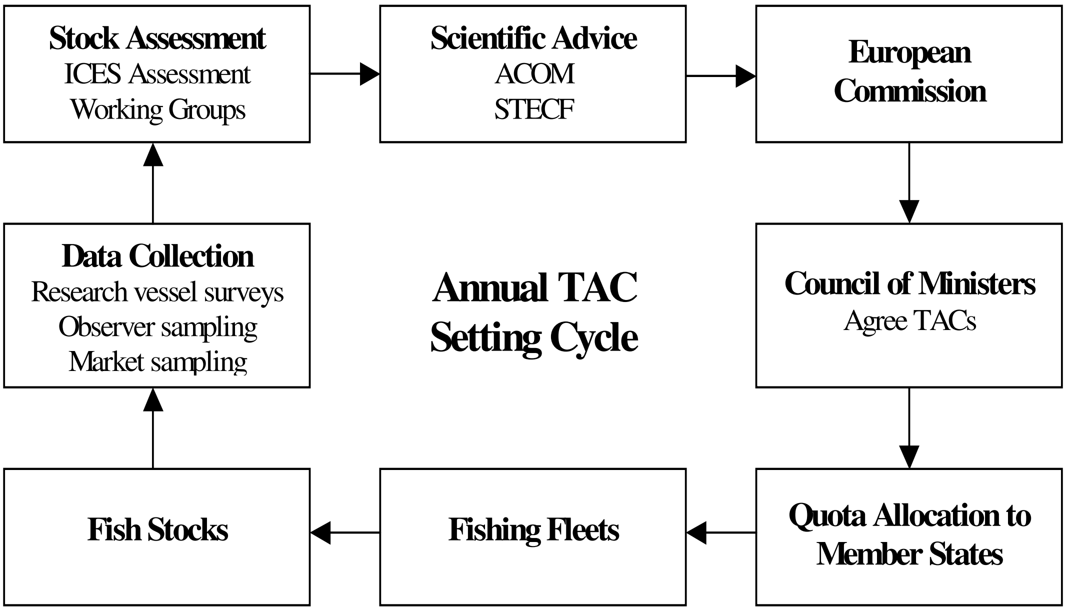
2.2. Introduction

Setting total allowable catches (TACs) underpins resource management and conservation within the Common Fisheries Policy (CFP) (Karagiannakos 1996), under which European fisheries have been managed since 1983 (Roberts 2007). Initially this resulted in major growth in the size of the Community fleet and overall catches increased until the mid-1980s (Pauly and Maclean 2002). At this time, many of the Community's most important fish stocks, such as cod, haddock and whiting, declined (Pauly and Maclean 2002). This led to a reappraisal of the CFP with the aim of rebuilding fish stocks and fisheries yields. As part of this initiative, Community financial aid was redirected towards fleet modernisation, reducing capacity, implementing protected marine areas, and expanding markets to include more species (Council Regulation (EEC) No 4028/86). Twenty years on, despite these provisions, many stocks have continued to decline and many shortcomings of the CFP have been recognised (e.g. Daw and Gray 2005; Khalilian *et al.* 2010). At the turn of the century (2002) the CFP was reviewed with the intention of a radical overhaul and reform of practices. Amongst other issues, the Commission of the European Council highlighted the application of the TAC system as a key factor in the failure of the CFP to incorporate sustainability into fishery resources (CEC 2001). The short term perspective and the failure of the system to follow scientific advice were noted as the major weaknesses. Together with changing policy, the CFP has been affected by the continuing expansion of the European Union placing additional pressures on resources and complicating decision-making processes (Kyriacou 2009). During the implementation of the 2002 reform, ten countries[6] entered the community in May 2004 resulting in the largest single expansion of the EU[7].

In theory, total allowable catches are set according to scientific advice provided by the International Council for the Exploration of the Sea (ICES), and the Scientific, Technical and Economic Committee on Fisheries (STECF) which is appointed by the European Commission. Every year, scientists provide stock assessments for the different management zones in European waters and recommend TACs to the European Commission for each stock and zone; fisheries ministers then set the legally binding TACs and negotiate to divide them amongst fishing nations (Figure 1).

[6] Cyprus, Czech Republic, Estonia, Hungary, Latvia, Lithuania, Malta, Poland, Slovakia and Slovenia
[7] http://ec.europa.eu/enlargement/archives/pdf/press_corner/publications/key_issues_from15_to_25_en.pdf

Figure 1. Flowchart describing the annual TAC setting cycle



Scientific advice is provided by the Advisory Committee (ACOM) of ICES and the Scientific, Technical and Economic Committee on Fisheries (STECF) appointed by the European Commission. Redrawn from Keltz and Bailey (2009)

The aim of this chapter is to explore the relationship between science and politics within this decision-making system by examining the impact of political adjustment, i.e. the degree to which ministers adjust scientifically-recommended TACs within European fisheries. This impact is explored through an analytical appraisal of available data from 1987 to the present.

2.3. Existing research into the TAC management system

Previous research has highlighted the impact of political decisions in the TAC management system. Karagiannakos (1996) examined the importance of TACs as a conservation measure for six demersal North Sea fish stocks; cod, haddock, sole, plaice, whiting and saithe, between 1980-1992. Investigating the scientifically recommended TAC, the TAC agreed by the Council, the actual catch and the spawning stock biomass (SSB), fish landings were found to follow changes in the SSB rather than changes in the TACs. Therefore, the catch was more affected by the condition of the stock rather than the TACs, indicating a weak correlation between TACs and landings. In addition, it was found that the TAC system generates wasteful practices such as high grading and discarding. The lack of fishermen's compliance with the TAC system and political pressures in deciding TACs were also highlighted as failings. For these reasons, Karagiannakos (1996) concluded that the TAC system did not contribute significantly to the sustainability objective of the CFP for the stocks studied.

Daan (1997) investigated the effectiveness of TAC management for two flatfish stocks (sole and plaice) in the North Sea between 1979-1996. In this evaluation, the reliability of the scientific assessment, the deviation between scientifically advised TACs and implemented TACs and enforcement issues were addressed. In all he found that attempts to constrain fishing mortality using TACs had failed. If anything, the implementation of TACs made the situation worse, with exploitation rates increasing after the introduction of the CFP in 1983. Daan (1997) identified the following failings in the TAC system: a) TACs control landings rather than total catches, and in a multispecies fishery they may encourage discarding, b) non-compliance leads to a deterioration in the quality of catch statistics entering stock assessments, and c) political negotiations undermine the TAC setting process. In particular Daan (1997) acknowledged that the ad hoc decision-making process encourages non-compliance as it fails to impart any long-term certainty for the industry on which fishers can base their economic strategy.

Maguire (2001) concluded that it is the uncertainty surrounding stock assessments and the implementation of the TACs that result in the failure of the TAC system to guarantee the sustainability of the resource. Elaborating on the implementation process, Roberts *et al*. (2005) state that fishery ministers usually set TACs 15-30% higher than recommended by fisheries scientists in order to reduce the impacts of TAC reductions on fishing communities. However, whilst this may reduce the impacts in the short term, the impacts may be more strongly felt in the future when greater reductions need to be made in response to declining stocks (Roberts *et al.* 2005). An evaluation by Rice and Cooper (2003) of the management of flatfish fisheries around the globe found that more than any other factor examined, failure to comply with scientific advice greatly increased the risk of unsustainability.

Del Valle and Astorkiza (2007) considered the process of TAC decision-making rather than the discrepancy between advised and agreed TACs. They observed the large number of agents that intervene to decide the agreed TAC, finding that the decision is made in an arena of many actors and institutions, all with their own often conflicting interests. Due to the biological and social complexity involved in the TAC setting process considerable divergence from scientific recommendations may therefore be expected. Within this study they identified a need to minimise the effect of perverse external pressures on the decision-making process through, for example, the use of harvest control rules (HCRs). They argue that HCRs help to remove the final responsibility for decision-making from politicians due to the presence of clear established rules.

Reiss *et al.* (2010) studied the linkage between TACs and fishing effort in the mixed fisheries of the North Sea. They concluded that variation in TACs has minimal impact on fishing effort and that as a result the use of TACs as the principal tool to regulate fishing activity is inadequate due to non-compliance.

These papers all identify the same issues with the TAC management system employed by European fisheries: that of a short-term ad hoc decision-making process that has provided incentives for high-grading, discarding and non-compliance within fisheries. This research aims to expand these studies through a more complete analysis of stocks in European Union waters. With the second reform of the CFP due to take place in 2012 (CEC 2009; Surís-Regueiro *et al.* 2011) it is hoped that this research will provide further evidence to managers and policy makers of the need to reform the current TAC system and more fully integrate science into policy.

2.4. Aims and objectives

The main aim of this research is to examine the history of the TAC setting process for a variety of stocks to determine the degree of discrepancy between advised and agreed quotas. Specifically the following questions will be asked:

1. To what extent are scientifically recommended TACs disregarded in the final TAC setting process?

2. Are some species more prone to higher levels of political adjustment than others?

3. Is political adjustment more evident for some management zones than others?
4. If differences between species and zones are found, what might be driving these differences?
5. What impact have the CFP reform and EU expansion had on political adjustment levels?

2.5. Methodology

2.5.1. Data

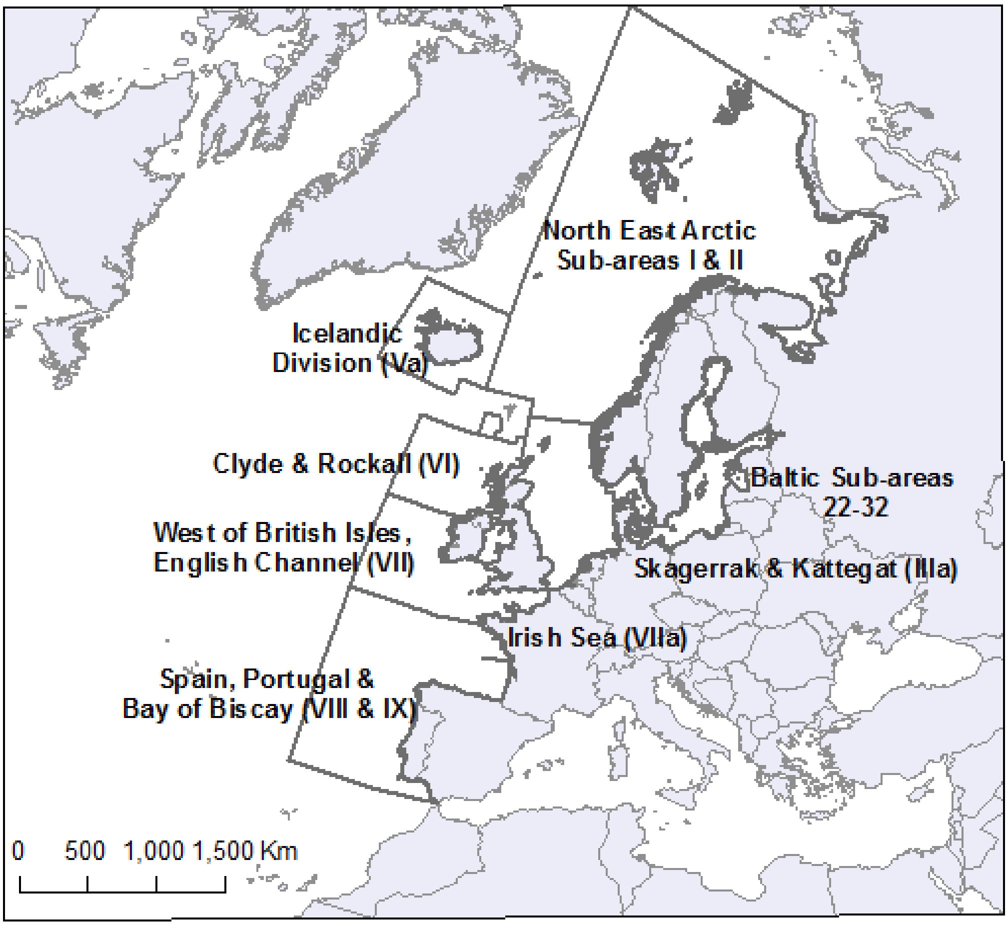
Data on advised and agreed TACs were obtained from a variety of sources including the ICES online advice archives[8] and official EU Council Regulations[9] and bilateral agreements[10]. The ICES data detail the recommended catch corresponding to scientific advice and the TACs implemented by the Council of Ministers. The database currently runs from 1987 to 2011, but it does not always contain continuous time-series for all stocks, because the required information is not always available. Scientific advice may not be consistently given for a variety of reasons, for instance the stocks were not considered, the advice was combined with that of other stocks, or the advice was phrased in text rather than numbers. Council Regulations and the documents relating to bilateral agreements between the EU and other nations were used to obtain data for the 2011 fishery and to validate ICES records.

In total, data were available for 44 stocks of 11 fish species across 9 management zones. The species analysed were; cod (*Gadus morhua*), plaice (*Pleuronectes platessa*), haddock (*Melanogrammus aeglefinus*), megrim (*Lepidorhombus spp.*), saithe (*Pollachius virens*), herring (*Clupea harengus*), sole (*Solea spp.*), hake (*Merluccius merluccius*), nephrops (*Nephrops norvegicus*), sprat (*Sprattus sprattus*) and whiting (*Merlangius merlangus*). These species were chosen because the EU manages them under a TAC system, they have economic importance and they are

[8] http://www.ices.dk/advice/icesadvice.asp
[9] e.g. EU (2010b). COUNCIL REGULATION (EU) No 23/2010 of 14 January 2010 fixing for 2010 the fishing opportunities for certain fish stocks and groups of fish stocks, applicable in EU waters and, for EU vessels, in waters where catch limitations are required and amending Regulations (EC) No 1359/2008, (EC) No 754/2009, (EC) No 1226/2009 and (EC) No 1287/2009. Official Journal of the European Union.
[10] EU (2010a). Agreed record of conclusions of fisheries consultations between the European Union and Norway for 2010. Brussels.

publically recognisable. Stocks for which there was a mismatch between the advice and TAC areas were excluded from the list. For example if the advice referred to area VIId and the TAC was set for area VIId and VIIe these stocks were removed from the calculations. The 9 management zones considered were the Baltic sub-areas 22-32, Skagerrak and Kattegat (IIIa), North Sea (IV), Northeast Arctic sub-areas I & II, Icelandic division Va, Clyde and Rockall (VI), West of the British Isles and the English Channel (VII), Irish Sea (VIIa), Spain, Portugal and the Bay of Biscay (VIII & IX) (Figure 2). These zones were aggregated from smaller management sub-areas for which ICES advised TACs.

Figure 2. Map of the nine management zones used in this study



To attempt to explain any trends that might be found between species or zones, data were also collected on fish price categories from fishbase[11], employment data from

[11] www.fishbase.org, last updated 2010.

Salz and Macfadyen (2007) and national websites[12] and seafood consumption patterns from FAOSTAT[13]. From these data, regional dependency patterns could be identified and compared to political adjustment levels for each zone. In addition, the proportion of stocks assessed in this study for each zone were identified as being within or outside safe biological limits so as to determine whether political adjustment is leading to a depletion of species[14].

2.5.2. Statistical analysis

Political adjustment

A political adjustment index (PAI) was calculated as a measure of the degree to which ministers' adjust scientifically recommended TACs. This was determined by calculating the percentage by which the official TACs differed from the advised TACs for all decisions made for all stocks in all years (PAI = ((agreed TAC – advised TAC)/advised TAC)*100). If science advised a TAC of zero, it was not possible to calculate the deviation from the TAC as dividing an agreed TAC by zero gives an infinite percentage for the PAI. Hence, where a moratorium was advised, the recommended TAC was set to zero and where a positive TAC was implemented in the same year the PAI was arbitrarily set to 100%. In order to account for this arbitrary figure, data were analysed both including and excluding moratoria years. A chi-squared test was used to determine whether decisions subject to adjustment above recommended TACs significantly outweighed those set below recommended TACs or those subjected to no adjustment.

Comparisons of political adjustment amongst species and management zones were made in order to determine whether some species and zones were more prone to adjustment than others. The average adjustment across the stocks of each species was calculated using two indices: a summation index and a mean index (see Table 1 for methodology). In the case of hake, only one stock was included in the analysis and therefore only the summation index was calculated. Trends of political

---

[12] Icelandic data obtained from http://www.statice.is/Statistics/Wages,-income-and-labour-market/Labour-market, Norweigian data from http://www.ssb.no/fiskeri_havbruk_en/, and Russian data from http://www.gks.ru/bgd/regl/b09_12/IssWWW.exe/stg/d01/06-03.htm.
[13] http://faostat.fao.org/site/354/default.aspx, last updated 2010.
[14] Identified from the 2010 ICES advice for each species.

adjustment were also examined for each management zone through the aggregation of species' data for each area.

Table 1. Methodology for calculating the summation and mean PAI

(a) Data on advised and agreed TACs for each species are separated by management zone

The table below is therefore replicated for each zone (i.e. Z_1, Z_2, ... Z_5) data were gathered for.

	Advised TAC	**Agreed TAC**	**PAI**
1987 ↓ **2011**			$\left(\frac{(Agreed\ TAC - Advised\ TAC)}{Advised\ TAC}\right)*100$

(b) Methodology for calculating the summation political adjustment index

Firstly, the data for advised and agreed TACs were added each year across zones.

$$Z_{1,AdvisedTAC,1987} + Z_{2,AdvisedTAC,1987} + \ldots + Z_{5,AdvisedTAC,1987} = \Sigma\text{AdvisedTACs for all zones}$$

$$Z_{1,AgreedTAC,1987} + Z_{2,AgreedTAC,1987} + \ldots + Z_{5,AgreedTAC,1987} = \Sigma\text{AgreedTACs for all zones}$$

This was repeated for all years, 1987-2011.

Secondly, the summation PAI was calculated for each year from these values, i.e.

$$\left(\frac{(\sum Agreed\ TAC_{1987} - \sum Advised\ TAC_{1987})}{\sum Advised\ TAC_{1987}}\right)*100 = \text{Summation PAI}_{1987}$$

This was also repeated for all years, 1987-2011.

Finally, the average summation $\text{PAI}_{1987\text{-}2011}$ was calculated, i.e.

$$\left(\frac{(\sum SummationPAI_{1987-2011})}{Number\ of\ years}\right)$$

(c) Methodology for calculating the mean political adjustment index

To calculate the mean PAI the third column in the table presented in (a) above is used. For each year the PAI for each zone is summed to produce a mean PAI for that year, i.e.

$$\left(\frac{(\sum Z1_{PAI,1987}, + Z2_{PAI,1987}, + \ldots + Z5_{PAI,1987},)}{Number\ of\ Zones}\right) = \text{Mean PAI}_{1987}$$

This was replicated for each year, 1987-2011.

The average mean $\text{PAI}_{1987\text{-}2011}$ was then calculated, i.e. $\left(\frac{(\sum MeanPAI_{1987-2011})}{Number\ of\ years}\right)$

Inferential statistics were applied to test whether there was a significant difference between the amounts of adjustment of TACs for the different species' and management zones'. Both sets of data were tested for normality and homogeneity of variances and were found to violate these assumptions. Consequently the non-parametric Kruskal-Wallis test was used to test for differences among species and the Mann-Whitney U test was used to determine where any differences lay between species or zones.

To determine the effect of the CFP reform in 2002 and the expansion of the EU in 2004 on political adjustment the PAIs were compared for the years prior to 2004 and those of 2004-2011. Whilst the CFP reform was finalised in 2002, it came into force on January 1st 2003 and therefore the first TAC negotiations to take place under the new policy were in December 2003 concerning the 2004 fishery. With 10 new countries joining the EU in May 2004, these two events were considered as one within the analyses as a result of their close succession. The data were again found to violate the assumptions of homogeneity and normality and the Mann-Whitney test was used to test for a difference between the year sets as well as to test the effect of the events on the PAIs of individual species and zones. All statistical tests were implemented using R (R 2008).

2.6. Results

Political adjustment by species

A large variation in the political adjustment of TACs both within, and between different stocks and species was found. Figure 3 and Figure 4[15] illustrate the variation in political adjustment that can be found in the same species, nephrops, within different management sub-areas. Within Division IXa the agreed TACs have been consistently higher than those advised until the last two years in the time series (Figure 3). It appears that the adjustment became smaller as the nephrops stock became more depleted. In contrast, the scientific recommendations were more closely followed for nephrops in Division VIa until 2007 (Figure 4) which appears to have mirrored a more stable stock level. After 2007, both scientifically recommended and agreed TACs increased however the agreed TACs have since

[15] No explicit TACs were recommended for the 2011 fishing season for either the IXa stock. Consequently, the political adjustment level for 2011 could not be calculated.

been set higher than those scientifically recommended. A rapid decline in the level of agreed TACs has been seen since, falling back towards recommended levels. The size of the stocks is not directly comparable however as the area of division VIa is 71,717km^2 greater than that of division IXa.

Figure 3. Advised and agreed TACs for the Division IXa nephrops fishery

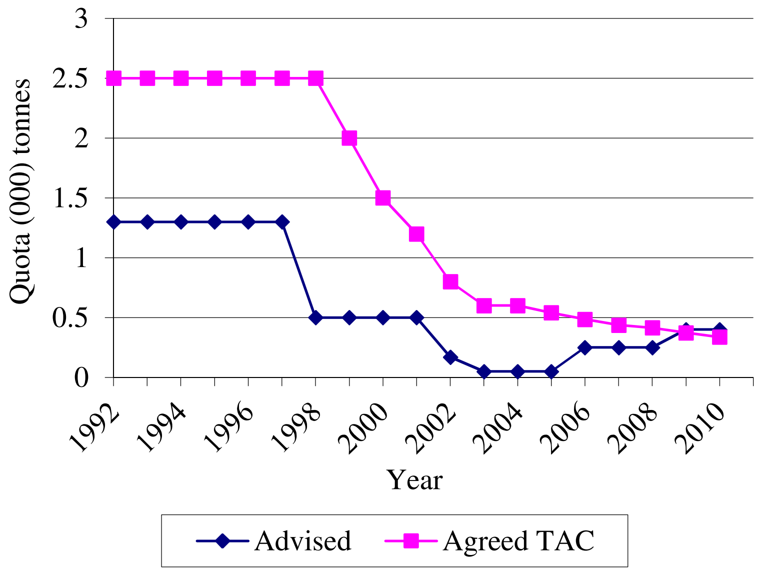
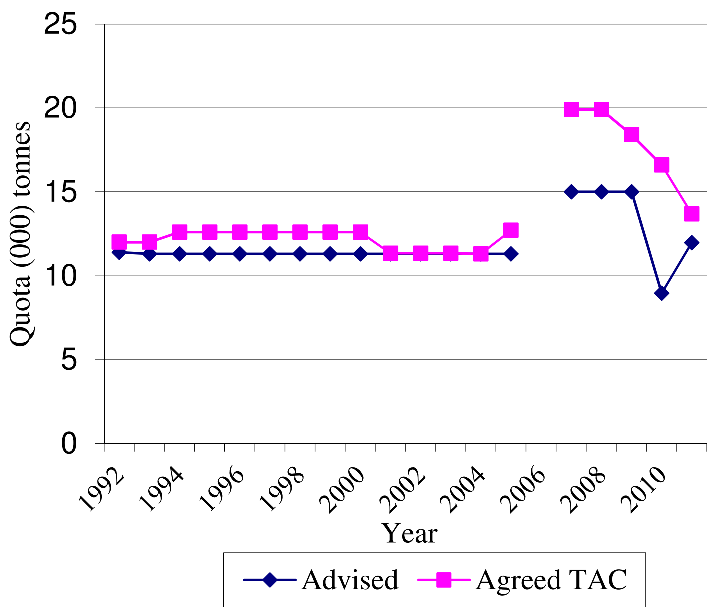


Figure 4. Advised and agreed TACs for the Division VIa nephrops fishery



Overall, in 68% of ministers' decisions TACs were implemented higher than the scientific recommendations. On average the adjusted TACs were set 33-37% (mean index lower end of range, summation index higher end) above catch levels recommended as safe by scientists. Of the remaining decisions, 14% of agreed TACs decisions were set lower than advised, while in 18% of cases the agreed TACs were set equal to those advised. The difference in these proportions is significant, χ^2 (2, N=877) = 480.45, $p < 0.001$, indicating that the process of political adjustment leads to ministers augmenting advised TACs far more often than reducing or accepting the scientific advice. It is interesting to note that in some cases ministers set TACs lower than advised. On average these were set 12% lower than advised from 1987 to 2011. Setting TACs lower than advised may be an attempt to compensate for exceeding the scientific advice elsewhere or it may be driven by industry demand for certain species over others.

The Mann-Whitney U test was used to test whether including the moratoria data years (and therefore the arbitrary 100% PAI value used when advised TACs were zero) in the analysis would significantly affect the results. The influence of including the moratoria years was found to be non-significant (U=317480, $p>0.01$). Scientifically, advising a moratorium is a serious action because it indicates that a stock is badly depleted. Ignoring this advice therefore shows a high level of political intervention. Consequently, the moratoria years have been included in all of the following analyses.

The level of political adjustment varied between species (Figure 5). A significant difference was found between the PAI of all species using the Kruskal-Wallis test (H_{10}=64.28, $p<0.01$). In order to determine where the differences lay the Mann-Whitney U test was applied (Figure 6).

Figure 5. Average political adjustment indices by species with standard error bars

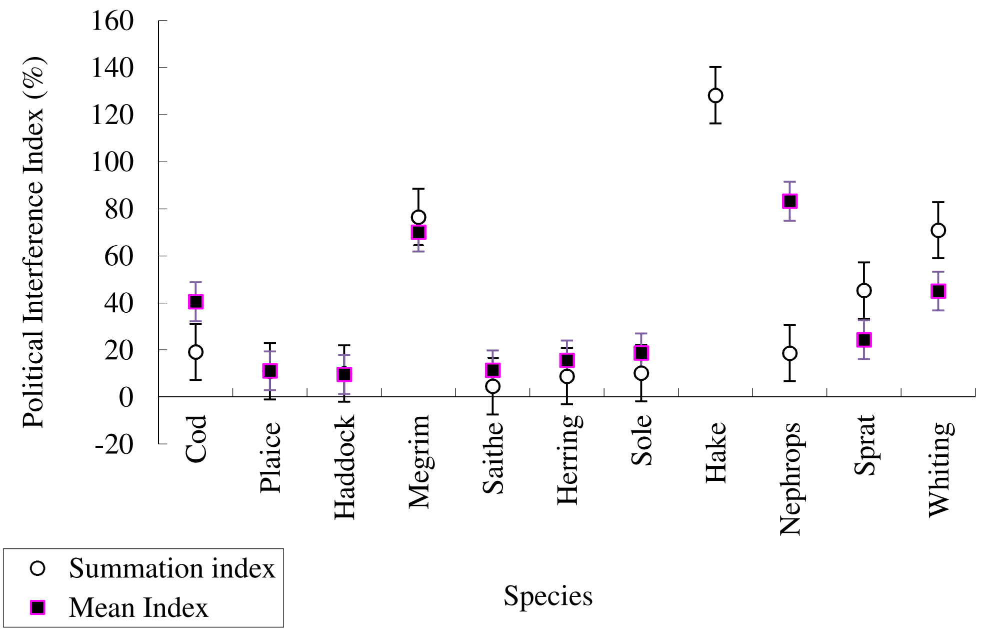
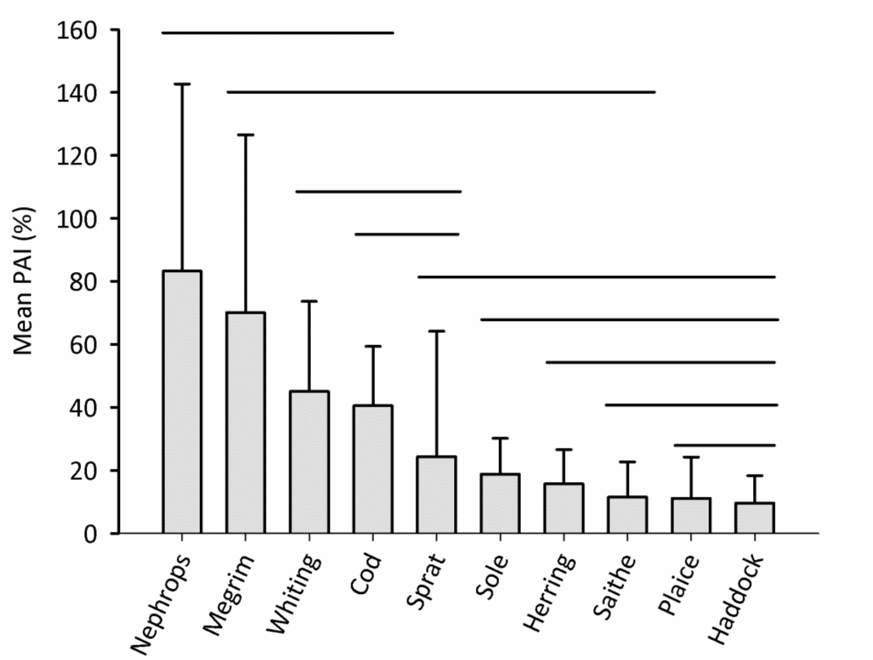


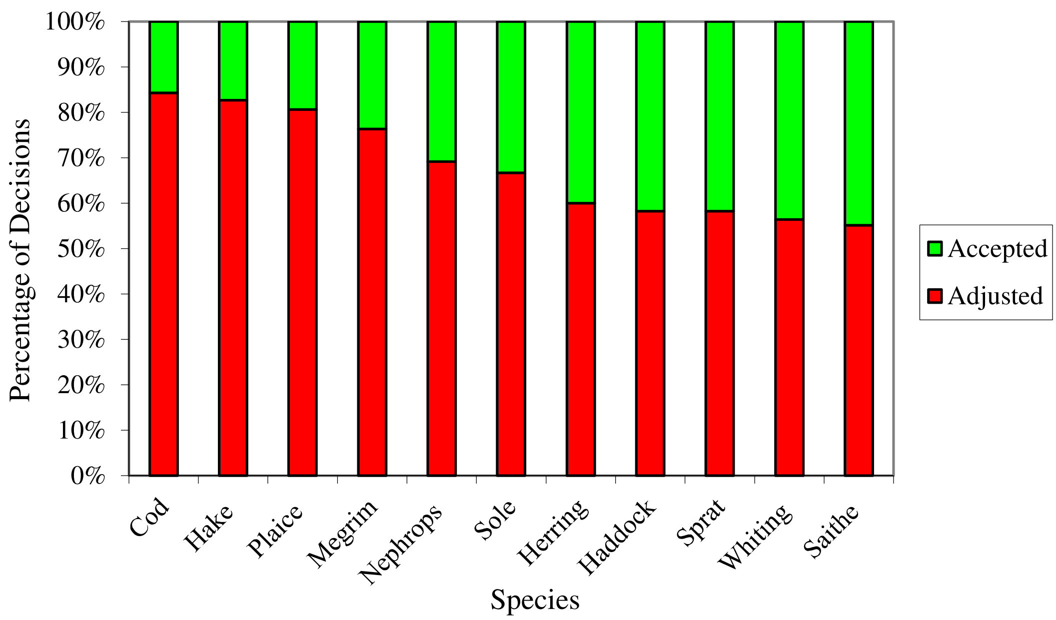
Figure 6. Variance around the mean PAI of each species (1987-2011) and significant differences as tested by Mann-Whitney U (p<0.01)



The shaded bars indicate the mean PAI value for each species. Error bars indicate the variance of values around the mean. Horizontal lines indicate non-significant Mann-Whitney U values between species, i.e. Nephrops does not have a significant number of annual mean political adjustment records larger than those of megrim, whiting, cod or sprat but is significantly larger than sole, herring, saithe, plaice and haddock. Hake is left out of this analysis as only a summation PAI value could be calculated due to limited data. See Appendix 1 for detailed statistical values.

Those species for which scientific advice has been followed least (less than 20% of advice accepted) are cod, hake and plaice (Figure 7). In the case of cod, 84% of advice was overruled and TACs were implemented up to 355% over those advised. Even for saithe, where scientific advice has most often been implemented, 45% of advice was overruled and TACs up to 143% over those advised were set in some areas. However, whilst a high percentage of advice might be adjusted the recommended TACs might only be adjusted slightly. For example, although cod ranks highest in terms of the percentage of scientific advice adjusted (Figure 7), this species has a similar mean PAI (41%) as that of whiting (45%) (Figure 5) for which only 56% of advice has been adjusted over the timeframe studied. Consequently it is important to examine multiple analyses to determine trends.

Figure 7. Percentage of scientific advice accepted or adjusted by the Council of Ministers for each species



An examination of the percentage of advice accepted or adjusted by ministers each year showed no evidence of improvement in decision-making over time or of changes in decision-making after the 2002 reform (Figure 8). The average PAI across all stocks each year of the agreed TAC relative to those scientifically recommended was also examined (Figure 9). No evidence of improvement in decision-making over time was found or of changes in decision-making after the 2002 reform (U=50, N=26, p=0.3).

Figure 8. Percentage of scientific advice accepted or adjusted by the Council of Ministers each year between 1987-2011

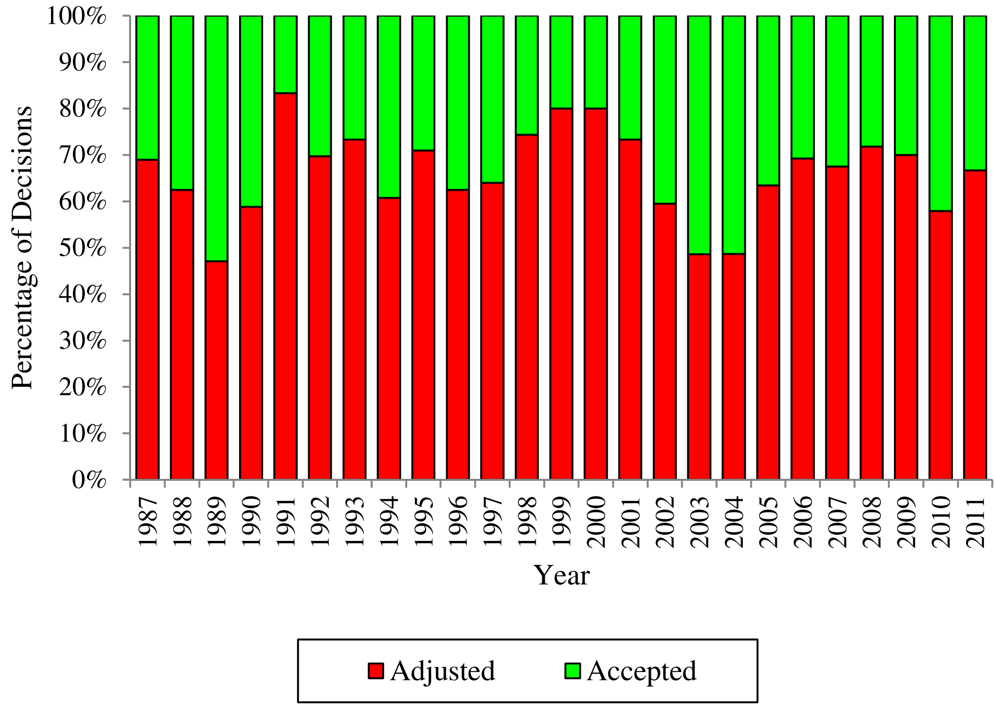
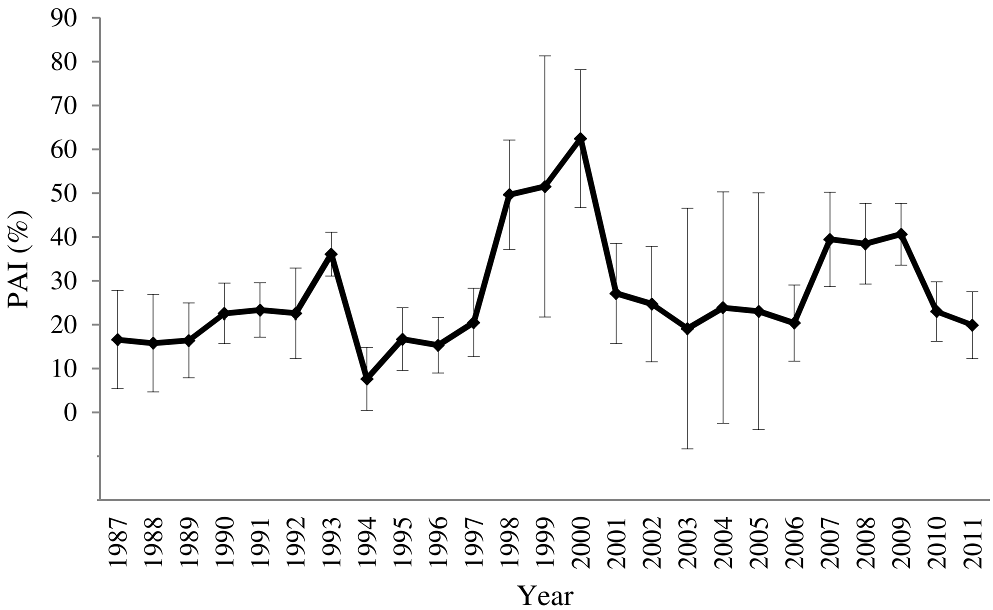
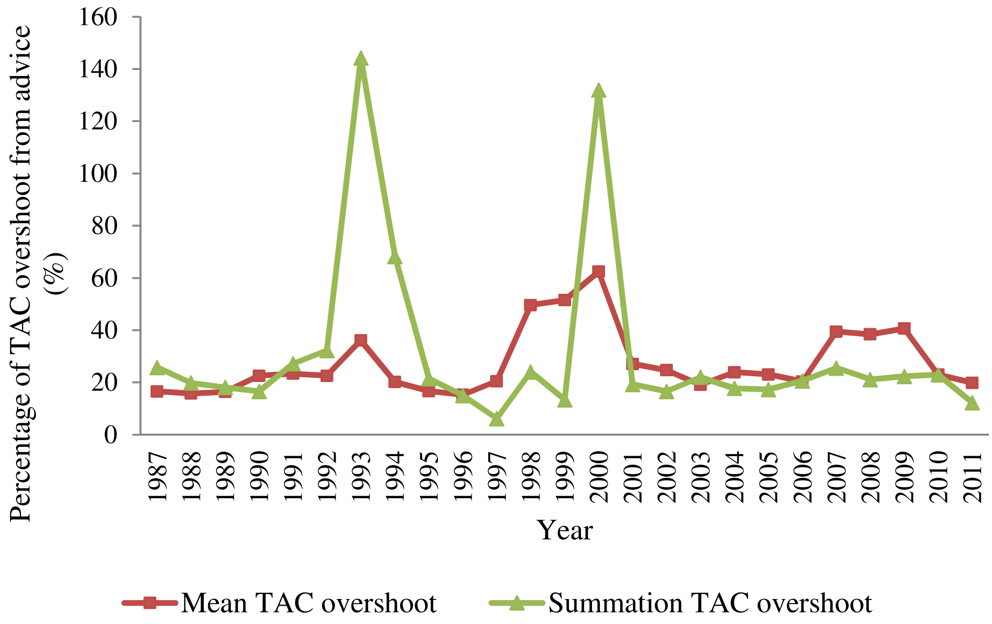


Figure 9. Average political adjustment index between 1987-2011



The average overshoot across all stocks each year of the agreed TAC relative to those scientifically recommended was also examined for both the mean and summation indices (Figure 10). This showed no discernible trend over the period analysed. The peaks shown in the summation index during 1993 and 2000 are the result of anomalously high levels of political adjustment for certain species in specific areas compared with other years. In 1993 the peaks are the result of high adjustment levels for cod (95% in the north-east Arctic), saithe (122% in sub-area VI (later amalgamated with the North Sea)), and hake (1100% in Spain, Portugal and the Bay of Biscay). The 2000 peak is the result of high adjustment levels for cod (255% in the north-east Arctic), haddock (68% in the north-east Arctic) and whiting (167% in Skagerrak & Kattegat). These are not picked up to as great an extent by the mean index due to the relative low levels of political adjustment in other areas balancing the higher values out and creating a lower index. No significant difference can be found after the CFP reform in 2002 (U=50, N=26, p=0.3).

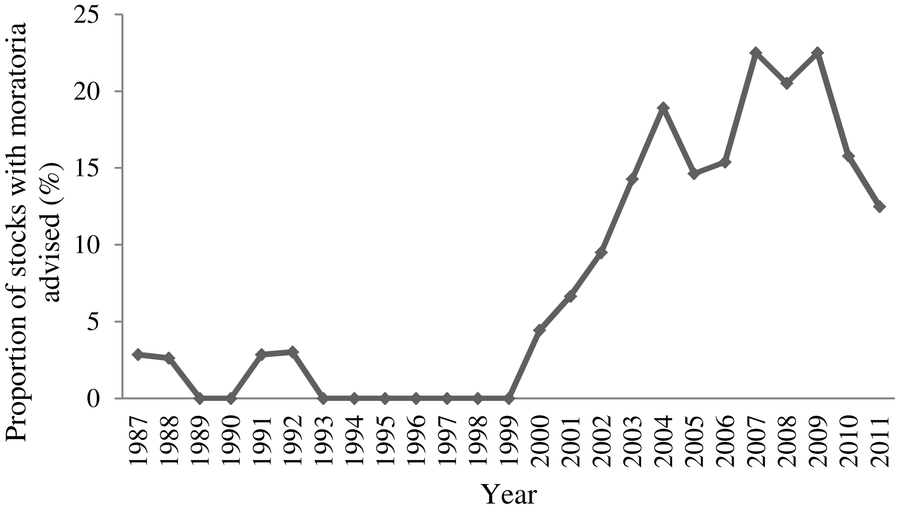
Figure 10. Average agreed TAC overshoot relative to scientific advice



If a moratorium was advised, it was not possible to calculate the political adjustment level and therefore if a TAC was implemented an arbitrary 100% PAI value was applied. Figures for the percentage of stocks per year for which moratorium was advised show that from 2000 the percentage increased from 5% to a maximum of

22% in 2007 and 2009 before falling to 12.5% in 2011 (Figure 11). For the stocks studied no recommendation for moratoria were followed; a TAC was always set.

Figure 11. Proportion of stocks for which a moratorium was advised



Changes in 2011

Since 2010 Maria Damanaki has been the EU Commissioner for Maritime Affairs and Fisheries. Widely known to support stricter determination of fishing quotas and a radical reform of the CFP in 2012 she has been quoted as saying "taking science as our starting point is the only possible approach" (Damanaki 2010). To the potential for limited reform and action she responded in an interview that "First, we will not have fish. Second, there is no second. It is as simple as that" (Rankin 2010). Following such tough words the proposal from the European Commission was expected to be uncompromising and based on scientific advice. The TACs set for the 2010 fishing season and those for the 2011 season for all stocks where data were available were therefore compared. These results are summarised in Table 2. While progress for some stocks has been made, most noticeably for hake in area VIIc and IXa (-82.63%) and nephrops in area VIa (-71.12%), there are still high levels of political adjustment for the 2011 TACs. Maria Damanaki's response to the 2011 TACs was that there is "disappointing news on some quota levels" (Anon 2011). From this it is clear that a reform of TAC decision-making is needed to place a legal requirement on politicians to follow scientific advice or else competitive bargaining will continue.

Table 2. Difference between the PAI for TACs set for the 2010 and 2011 fishing season

Stock	Area	PAI 2010	PAI 2011	PAI Difference Positive change	PAI Difference Negative change
Cod	VIIa	100*	100*		0
	I & II	5.11	0	-5.11	
	Va	11.11	0	-11.11	
	IIIa	100*	100*		0
Plaice	VIIa	1.69	1.69		0
	VIIf-g	36.67	0	-36.67	
	IIIa	-20.47	24.23		44.69
Haddock	Va	10.53	-1.96	-12.49	
	I & II	0	0	0	
Megrim	VIIIc & IXa	43	44.61		1.607
Saithe	Va	47.06	25	-22.06	
	I & II	0	0	0	
Herring	VIIa	0	10		10
Sole	VIId	32.26	0.25	-32.01	
	VIIIa,b	-1.45	1.19		2.64
Hake	VIIIc & IXa	89.8	7.16	-82.63	
Nephrops	IIIa	0	10		10
	VIa	85.41	14.29	-71.12	
	VIIIc	100*	100*		0
	VIIIa,b	14.68	14.68		0
Whiting	IV	89.66	56.13	-33.54	

* refers to PAIs where a moratorium was advised

Those stocks not included in this analysis were: Cod in area IV and 22-32; Plaice in IV; Haddock in IV; Saithe in IV; Herring in Va, VIaN, VIaS and the Norwegian spring-spawn herring; Sole in IIIa, VIIa, VIIe, VIIf-g and IV; Nephrops in IXa; Sprat in IIIa, 22-32 and IV; and Whiting in VIIa and IIIa. These stocks could not be compared for these years due to a TAC not being explicitly advised in one or the other year (advice was based on scenarios rather than an explicit TAC). 'Positive change' in PAI refers to those decisions where the PAI has been reduced or maintained (when following scientific advice). 'Negative change' in PAI refers to those decisions where the PAI has been increased or maintained (when not following scientific advice). These classifications have been put in purely for visual purposes.

Fish prices

Some fish species command a higher market value and are consequently of greater economic importance than others. The market value of each species analysed was obtained in order to investigate whether the higher the value of the species the greater the level of political adjustment. Table 3 categorises fish species in terms of price as calculated by Sumaila *et al.* (2007) for FishBase[16]. The species are ranked in order of decreasing average PAI. PAI tends to decrease with decreasing market value; however sole and plaice represent two anomalies to this trend.

Table 3. Average PAI and price category for each species analysed

Species	Price Category	Average PAI (1987-2010)
Hake	High	128
Megrim	Very high	70
Nephrops	Very high	83
Whiting	Medium	45
Cod	Medium	41
Sprat	Low	24
Sole	Very high	19
Herring	Low	16
Saithe	Low	12
Plaice	Medium	11
Haddock	Low	10

Price category data from FishBase, calculated according to ex-vessel prices (the price that fishers receive when they sell their catch) by Sumaila et al. (2007). The categories are defined on a percentile basis, i.e. the data are sorted from high to low values and subdivided to 20% groups, according to the number of pre-defined categories (e.g. low, medium, high, very high). The group that falls into the highest 20% is assigned to the category 'very high' and the last 20% into the 'low' category. Colour coded by price category. Average PAI refers to the mean index except for hake where only the summation index could be calculated due to limited data.

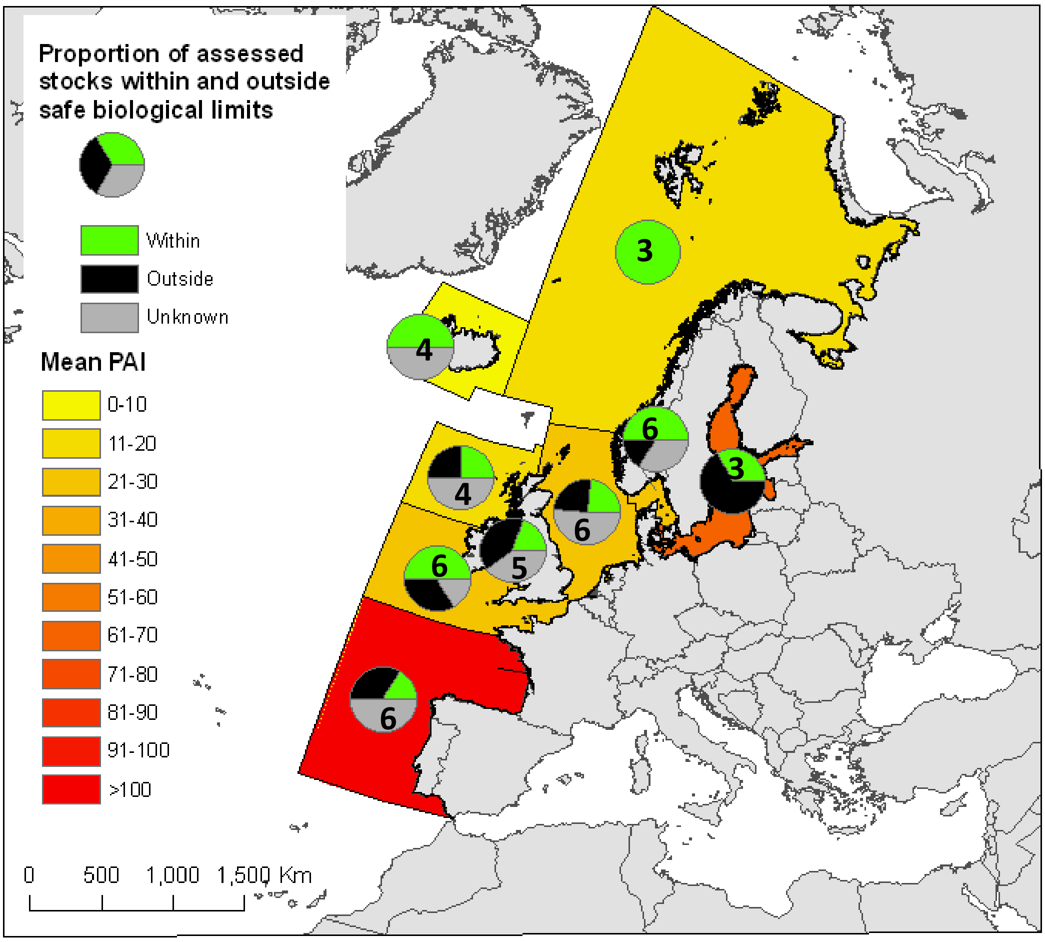
[16] as defined by Fishbase (www.fishbase.org)

Status of assessed stocks

The proportion of stocks assessed for each zone were identified as being within or outside safe biological limits[17] so as to determine whether political adjustment is linked to depletion of species. There are a number of gaps within the data where ICES has not set any reference points for the stocks. Consequently in many of the zones a high proportion of the stocks analysed are assessed as being unknown. In general, the higher the level of political adjustment, the greater the proportion of stocks fall outside safe biological limits (Figure 12). However, the low and uneven number of stocks examined for each area together with the high level of data gaps make it difficult to draw robust conclusions, particularly as some of the most valuable stocks remain unassessed (such as two stocks of nephrops around Spain, Portugal and the Bay of Biscay and cod within the waters of Iceland).

[17] Identified from the 2010 ICES advice for each species

Figure 12. Status of assessed stocks in each management zone

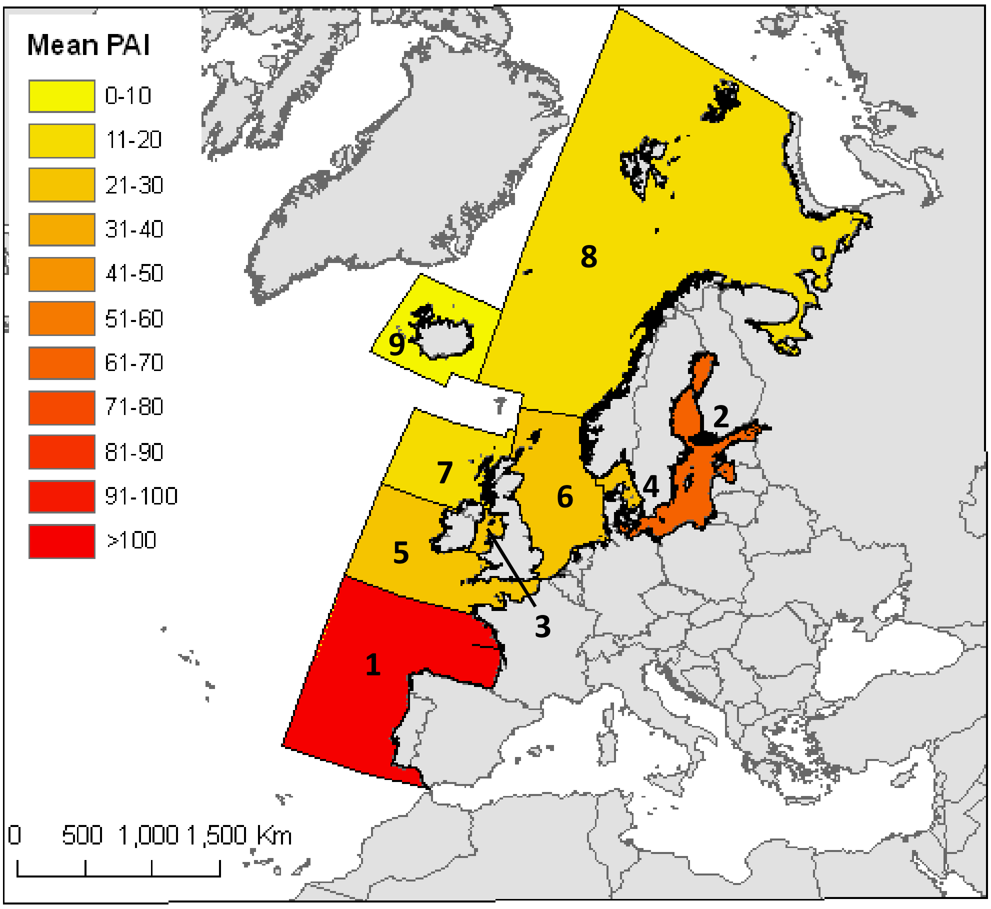


A stock is considered to be outside 'Safe Biological Limits' when the Spawning Stock Biomass is below a biomass precautionary approach reference point (B_{pa}). The status of the stock is stated as unknown if this reference point has been undefined or no data are available. Numbers indicate the number of stocks assessed within this study for each zone.

Political adjustment by management zone

The analysis of the average PAIs by management zone are shown in Figure 13. The Spanish, Portuguese and Bay of Biscay zone (division VIII/IX) has a higher average level of political adjustment than elsewhere. Iceland (division Va) shows the lowest average PAI.

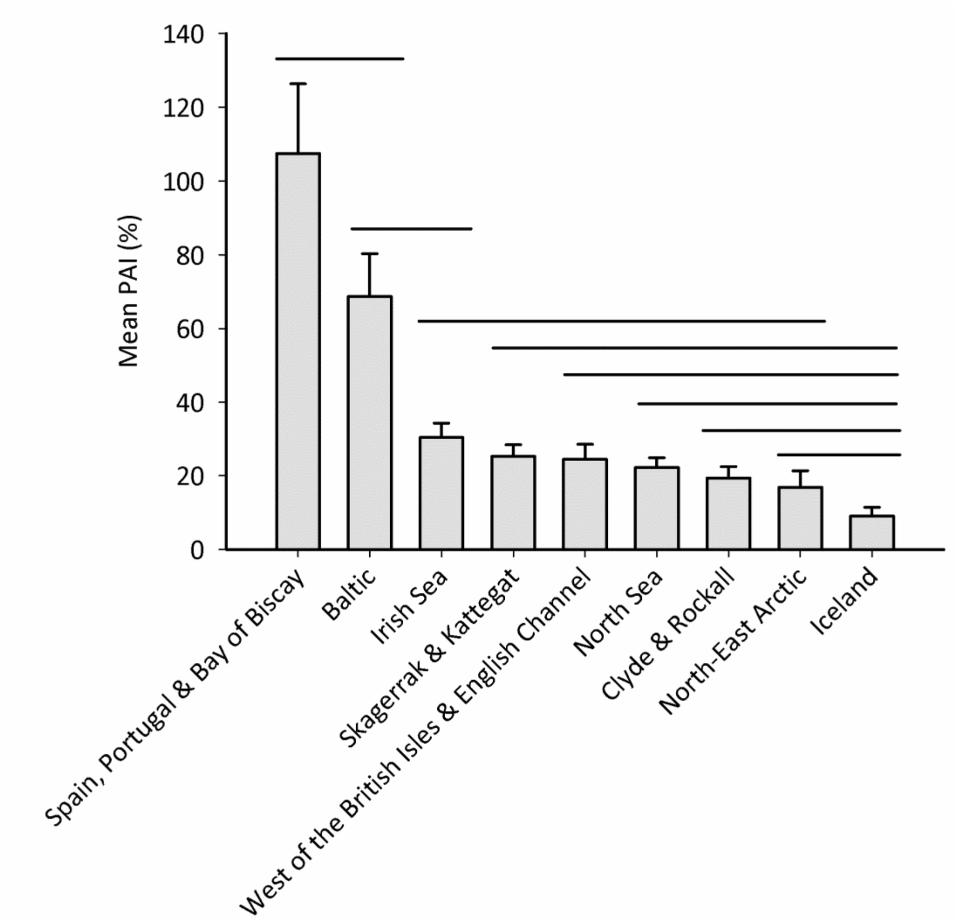
Figure 13. Average political adjustment indices by management zone



Average PAI values for the zones are as follows (in descending order): 1. Spain, Portugal and the Bay of Biscay (division VIII/IX) = 107.47; 2. Baltic (sub-areas 22-32) = 68.68; 3. Irish Sea (VIIa) = 30.48; 4. Skagerrak & Kattegat (division IIIa) = 25.35; 5. West of the British Isles & English Channel (VII) = 24.54; 6. North Sea (IV) = 22.18; 7. Clyde & Rockall (VI) = 19.35; 8. North-East Arctic (sub-areas I & II) = 16.82; 9. Iceland (division Va) = 9.01.

A significant difference was found in the level of political adjustment between zones using the Kruskal-Wallis test (H8=72.52, p<0.01). In order to determine where the differences lie the Mann-Whitney U test was applied. Those zones found to have significant differences are detailed within Figure 14.

Figure 14. Variance around the mean PAI of each management zone (1987-2011) and significant differences as tested by Mann-Whitney U (p<0.01)



The shaded bars indicate the mean PAI value for each management zone. Error bars indicate the variance of values around the mean. Horizontal lines indicate non-significant Mann-Whitney U values between zones, i.e. Spain, Portugal and the Bay of Biscay (division VIII/IX) does not have a significant number of annual mean political adjustment records larger than that of the Baltic but does compared to all other zones. See Appendix 1 for detailed statistical values.

Regional dependency

In order to investigate regional dependency within EU fisheries the EU commissioned a study (Salz and Macfadyen 2007) building of that on Salz *et al.* (2006) regarding employment within the fisheries sector. These studies showed that the fisheries sector accounts for a low share of the total jobs in all Member States (Figure 15 and Table 4). In general those zones with a higher PAI are bordered by countries with greater levels of employment within the fishing sector, although data were not available for Icelandic and Northeast Arctic waters. In terms of employment in the EU fishery sector as a whole, the four most important countries are Spain, France, Italy and Greece, which account for 51% of the EU total (Salz and

Macfadyen 2007). Even in localities which are traditionally highly dependent on fishing, the proportion of people employed within the fisheries sector is low and declining (Salz and Macfadyen 2007; STECF 2009; Villasante 2010). At regional level, fisheries were identified as playing an important role as a source of employment in some key areas; notably, Galicia (Spain), Algarve (Portugal) and North-East Scotland (UK)[18] (Salz and Macfadyen 2007). Iceland was found to support the highest fisheries percentage of total employment for all countries at 4.09%[19].

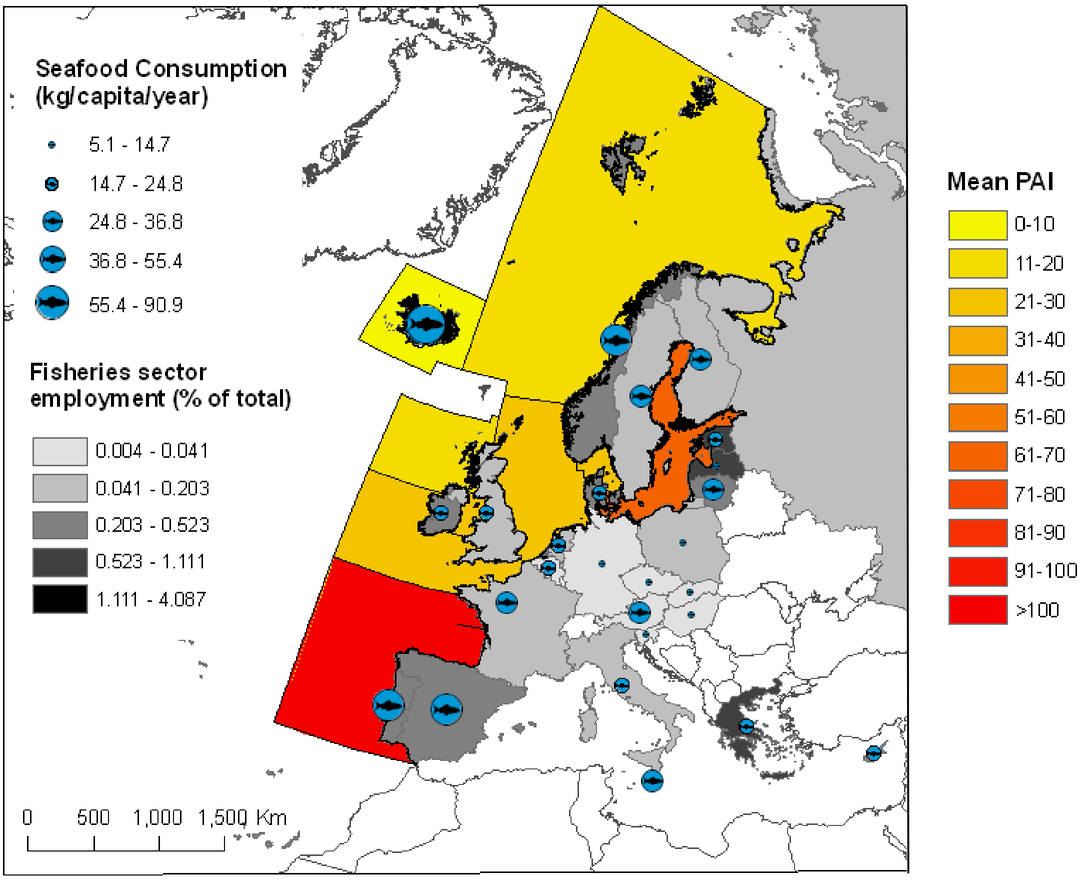
Another measure of regional dependency is the cultural value a country places on their fisheries. At 87.4 kg/year Iceland has the highest per capita seafood consumption with Portugal following at 54.8 kg/year (FAO 2010a). From the data available, there appears to be no correlation between employment and seafood consumption. For example, Estonia has the second highest fisheries employment level but only ranks 20th in terms of consumption. However, with the exception of Icelandic waters, zones with highest average PAIs correspond with the countries surrounding them having higher employment and seafood consumption rates (Figure 15).

While the EU currently has five Mediterranean member states (Spain, France, Italy, Greece and Cyprus), this region has not been considered within this research. In general, catch limits or quotas are not applicable within the Mediterranean, with the exception of limits on bluefin tuna that have been introduced in response to recommendations by the International Commission on Conservation of Atlantic Tuna (ICCAT)[20] (Cacaud 2005). Consequently, while these countries are illustrated on all maps for accuracy, national waters within the Mediterranean are not considered.

---

[18] In 2004 1.9% of populations within the Algarve and Galicia were employed in fishing, the Algarve has a dependency rate on the fisheries sector of 5.1% and Galicia 4.1%. The North-East Scotland has a dependency rate of 2.3% employing 0.7% of its population in fishing (Salz and Macfadyen 2007).
[19] Data obtained from Statistics Iceland: http://www.statice.is/Statistics/Wages,-income-and-labour-market/Labour-market
[20] http://www.eubusiness.com/news-eu/fisheries-management.175

Figure 15. Average political adjustment indices, national employment and consumption rates



Seafood consumption data are from FAOSTAT and is based on the year 2007. Employment data are taken from Salz and Macfadyen (2007) and are based on data from 2005, the most recently available data from the EU.

Table 4. Employment by main region and fisheries sub-sector within the EU

Region Name	Fisheries Sector (total no. employed)	Percentage of regional employment
North Sea	47500	0.2
Baltic Sea	54000	0.4
Atlantic Areas	138000	0.6

Data based on employment within the regions fisheries in 2005, i.e. North Sea data are based on those countries that fish within the area. Reproduced from Salz and Macfadyen (2007)

The effect of the CFP reform and EU expansion

Mann-Whitney U tests comparing the pre-reform and post-reform average PAIs found no significant difference between the years before the reform and expansion, and those after. This was the case both between all species (U=545, N=873, p=0.13) and management zones (U=513, N=825, p=0.13). In addition, no species showed significant differences between pre- and post-reform periods. The PAIs for haddock, megrim, saithe, herring, hake, sprat and whiting were lower after the reform and expansion, while cod, plaice, sole and nephrops had higher PAIs after the events (Table 5).

In addition, none of the management zones were found to have been significantly affected by these events. The PAIs for the Spanish, Portuguese and Bay of Biscay management zone together with the North East Arctic were found to be lower after the reform and expansion, while the PAIs for all other zones were seen to increase (Table 5). These patterns suggest that the reform and expansion events may have had a largely detrimental effect within management zones, compounding the problem of political adjustment.

Table 5. Summary of results comparing the PAI pre- and post-reform

	Pre 2002 reform	Post 2002 reform	Overall
Average total PAI (%)	27 (± 12)	38 (± 14)	33 (± 8)
Average PAI (%) for each zone*			
1. Spain, Portugal & Bay of Biscay	121(± 24)	92 (± 30)	107 (± 19)
2. Baltic	53 (± 8)	104 (± 33)	69 (± 12)
3. Irish Sea	23 (± 3)	44 (± 10)	30 (± 4)
4. Skagerrak & Kattegat	24 (± 4)	25 (± 6)	25 (± 3)
5. West of the British Isles & English Channel	10 (± 2)	61 (± 11)	25 (± 4)
6. North Sea	19 (± 3)	28 (± 6)	22 (± 3)
7. Clyde & Rockall	14 (± 3)	35 (± 7)	19 (± 3)
8. N.E. Arctic	24 (± 6)	3 (± 3)	17 (± 5)
9. Iceland	8 (± 2)	11 (± 7)	9 (± 2)
Average PAI (%) for each species*			
1. Hake	152 (± 80)	99 (± 1)	134 (± 53)
2. Megrim	170 (± 31)	22 (± 59)	90 (± 7)
3. Nephrops	76 (± 20)	113 (± 48)	88 (± 21)
4. Cod	28 (± 5)	70 (± 11)	43 (± 5)
Whiting	42 (± 8)	45 (± 11)	43 (± 7)
5. Sprat	55 (± 11)	5 (± 10)	42 (± 9)
6. Sole	12 (± 2)	39 (± 10)	20 (± 4)
7. Herring	13 (± 3)	25 (± 7)	17 (± 3)
8. Saithe	13 (± 3)	8 (± 11)	12 (± 4)
Plaice	8 (± 2)	19 (± 11)	12 (± 4)
9. Haddock	13 (± 3)	2 (± 3)	10 (± 2)
% of decisions TACs set higher than recommended	71 (± 3)	64 (± 6)	68 (± 3)
% of decisions TACs set lower than recommended	10 (± 2)	18 (± 5)	14 (± 3)
% of decisions TACs set according to scientific advice	19 (± 4)	18 (± 4)	18 (± 3)
Number of moratoria implemented/total number recommended	0/13	0/59	0/72

** Listed in descending order of average overall PAI, ± standard error*

2.7. Discussion

In the majority (68%) of TAC setting decisions taken by the Council of Ministers from 1987 to 2011, TACs were implemented higher than those recommended by ICES. On average the adjusted TACs were set 33-37% (mean - summation index) above catch levels recommended as safe by scientists. This disregard for the scientific advice undermines the scientific basis for management, as well as endangering fish stocks and the fisheries that depend on them. In turn, this has led to a lack of confidence in the governance systems in place by both the public and industry (Pita *et al.* 2010; Mackinson *et al.* 2011). In 1983 Leigh (p.90) stated that "the sum of member states' demands added up to more than the total amount of fish available. In the bad old days [...] (this) led to the inflating of TACs, followed by overfishing". From this analysis it is clear that this situation has still not been resolved. However, in 14% of decisions the agreed TACs were actually set lower than those advised. Those species for which decisions were most often set lower than advised were haddock (26% of all decisions), saithe (24%) and sprat (29%). These were set lower by 8%, 9% and 18% respectively when averaged across all areas and the whole timeframe (1987-2011). Setting TACs lower than advised may be an attempt to compensate for exceeding the scientific advice elsewhere or it may be driven by industry demand for certain species over others. For example, haddock, saithe and sprat all have a lower market value than other species such as hake, megrim and nephrops and therefore they may be subject to less industry demand and reductions may be more acceptable.

This evaluation of the integration of scientific advice into decision-making was based on a comparison of the official TAC set with the original scientific advice. Investigating the level of political adjustment relative to the scientific advice (Figure 10) shows no obvious convergence over time, and no reduction in political adjustment after the CFP reform in 2002. The increase in the number of stocks for which moratoria were advised (Figure 11) from 2000 provides a complementary measure to that of the PAI. Advocating a moratorium is a serious action and often the last resort for scientific advisors. However the gravity of this action appears not to be appreciated by the political community as, for the stocks assessed here, no zero-catch advice was accepted; a TAC was always implemented. This shows either naivety in managing natural resources, a lack of trust in scientific assessments or reflects the short-term goals of politicians and the industry they oversee. Whilst the fishing seasons of 2010 and 2011 indicate a greater integration of scientific advice, this may be the result of advice being based on scenarios rather than explicit TACs, as well as recovering fisheries. Overall however, the increasing number of advised moratoria paints a worrying picture for the deterioration of stocks.

Scientific advice is least often followed when reduced catches are advised. One of the reasons often identified to explain this is that reducing landings has a short-term economic cost on the fisheries sector (Roberts *et al.* 2005). Managers may deviate from scientific advice to limit inter-annual variations in landings as they disrupt market chains and eventually result in less profitability (Patterson and Résimont 2007). However, while setting landings higher than recommended may reduce impacts in the short-term, this is likely to increase the cost to fishers in the future when more drastic measures may be needed to rebuild the stock (Shertzer and Prager 2007). Whilst ensuring the long-term sustainability of the resource is in the interest of fishers and the fisheries sector economic perspectives (e.g. discounting (Clark 1973a, b, 1990; Sanchirico *et al.* 2006) and markets (Patterson and Résimont 2007) provide good reasons why long-term objectives may fail to take precedence when setting TACs (Daw and Gray 2005).

A second problem that has been identified with the current TAC system is related to the electoral politics of fisheries ministers (Daw and Gray 2005; Cardinale and Svedäng 2008) and the competitive bargaining forum within which ministers operate. Being democratically elected national politicians, their careers are maintained through popularity amongst their electorate. This may influence their decisions so as to avoid contentious issues that may create unemployment or short-term economic losses and reject long-term policies, the benefits of which will not be felt during their period of office (Daw and Gray 2005). Indeed, the fisheries ministers may consider raising the TACs above those recommended as a service to their respective national industry (Corten 1996). The current decision-making process leads to the paradox of ministers' acting to protect both their national and personal interests while at the same time needing to allocate quotas among member states for mutual benefit and to achieve conservation goals. As the Rt. Hon. John

Gummer, a former UK Minister of Agriculture, Fisheries and Food and the Secretary of State for the Environment, put it; "If you are a fisheries minister you sit around the table arguing about fishermen - not about fish. You're there to represent your fishermen. You're there to ensure that if there are ten fish you get your share and if possible a bit more. The arguments aren't about conservation, unless of course you are arguing about another country" (NIA 2001). With ministers taking this stance it is unsurprising that agreements are often difficult to reach and TACs are adjusted.

In 2003 the EU is estimated to have spent approximately 1.2 billion US$ of public money on beneficial (and necessary) subsidies that promote fishery resource conservation and management; such as scientific research, stock assessments, establishing marine protected areas and ensuring compliance (Sumaila *et al.* 2010). Over the last 2 decades, ICES has produced numerous scientific documents for the EU on the status of fish stocks, advising management strategies and recommending TACs. However, for the species studied within this paper, only 32% of the relevant recommendations have been accepted and implemented. Sustainability of fisheries can only be achieved by effective application of scientific advice.

Often the justification by fisheries ministers to adjust scientifically recommended TACs is that uncertainty is inherent within stock assessment (Sovacool 2009; Khalilian *et al.* 2010), and that scientific advice is based solely on the biological aspects of the fishery neglecting to take socio-economic aspects into account (Aps *et al.* 2007). While the uncertainty inherent in stock assessment is not denied (e.g. Hauge 2011) and it is true that socio-economic aspects are not taken into account when calculating TACs, in the long-term the industry would be best served by following scientific advice to create stable sustainable yields.

In general, the higher the level of political adjustment the greater the proportion of stocks which fall outside safe biological limits. However, with the high level of data deficiency it is difficult to reach robust conclusions, particularly as some of the most valuable stocks remain unassessed by ICES. This is a problem that has often been commented on, and with 61% of stocks within the Northeast Atlantic unassessed it is no wonder that it is difficult to determine trends within many fisheries (e.g. Beddington *et al.* 2007).

The two management zones most prone to political adjustment are the Spanish, Portuguese and Bay of Biscay zone (division VIII/IX) and the Baltic Sea (sub-areas 22-32) (Figure 13). The waters around Iceland (division Va) are the least prone to political adjustment. These results indicate that regional differences are driving the average PAIs for the management zones.

Within the process of setting TACs the notion that preference should be given to countries or regions dependent on fishing is prominent ('relative stability'), although definitions or quantifications of dependency are limited (Hoel and Kvalvik 2006; Anderson *et al.* 2009). Despite this, in an attempt to investigate trends in dependency, I analysed statistics for regional employment and consumption (Figure 15 and Table 4). These data clearly show that the management zones at greater risk of political adjustment are for the most part those with the highest rates of fishery employment and seafood consumption. In particular, the high dependence on fishing in the southern Atlantic area and Baltic areas provides further evidence for the hypothesis that regions with higher dependence on fishing for employment are likely to exert more pressure on politicians, resulting in high political adjustment rates. However, Iceland presents an anomaly in this trend. Whilst following scientific advice most closely, Iceland appears able to support the highest employment and consumption rates (Figure 15), at the same time maintaining stocks above safe biological limits (Figure 12).

Icelandic waters often represent an anomaly within the datasets presented, having the lowest PAI and some of the healthiest stocks whilst being able to support a high regional dependency on fishing (Figure 13 and Figure 15). Iceland, together with Norway, are only members of the European Economic Area rather than the European Union. This enables these countries to gain free access to European markets, except for fish and agricultural products as these were excluded from the agreement. Consequently, the fisheries within Icelandic and Norwegian waters are managed exclusively by these national governments and bilateral agreements between themselves and the EU. Thus there are many differences between the management of the fisheries in Icelandic and Norwegian waters and the rest of European waters including discard bans and a mix of effort and quota regulations (Eliasen *et al.* 2009; Johnsen and Eliasen 2011). In the aftermath of the collapse of the herring fishery and declining pelagic stocks in Nordic waters, Iceland and Norway introduced rights-based management schemes in the form of licenses and individual transferable quotas (ITQs) during the 1970s (Jakobsson and Steffánsson 1999, Eliassen *et al.* 2009). These schemes have been constantly updated in the years since; by 1991 all Icelandic fisheries were managed using ITQs and as of 2009 most Norwegian stocks of economic importance had adopted this system (Eliasen *et al.* 2009). Particularly in Iceland the economic performance of the fishing fleet has improved significantly with the application of the ITQ system, and fisheries are among the most profitable economic sectors in Iceland. In addition to implementing rights-based management however, Iceland also has a history of implementing the precautionary approach to fisheries management and following scientific advice, even when this will cause short-term economic losses (e.g. Rosenberg 2003). For example, in 2011 Iceland banned virtually all halibut fishing because of fears regarding the state of the stock[21].

Whilst some inferences regarding the effect of regional dependency on political adjustment rates can be made, the analysis of employment and consumption statistics only provides an indication of regional dependency on the fishery sector. For example, these data fail to consider the value of the nations' fisheries or the availability of alternative employment.

The EU committed at the World Summit on Sustainable Development in 2002 to develop sustainable fisheries and restore fish stocks by 2015. It has been estimated that even if fishing had been completely halted in 2010 we would still overshoot this target by 30 years (Froese and Proelß 2010). Since 1983, when the CFP came into force, sustainability has been the core goal of fisheries management; however, management under the CFP has only driven many stocks further away from this objective. These results corroborate previous analyses showing that the CFP has failed to have any clear positive effect on marine fish stock development (Sparholt *et al.* 2007).

The decline of European fish stocks is well-documented, with 88% now overexploited relative to maximum sustainable yield targets and 46% fished outside safe biological limits (Condé *et al.* 2010). It is clear from this analysis that the political mismanagement of fisheries must bear considerable responsibility for this

[21] http://www.fishupdate.com/news/fullstory.php/aid/16812

decline. Further responsibility must lie with other management failures such as poor quota management, heavy subsidies, discards policy, high grading and non-compliance with fishery regulations.

Evidence for the unsustainable outcome of the process used within the EU to determine fisheries TACs can be found elsewhere. Competitive bargaining is also endemic in many regional fisheries management organisations, such as the International Commission for the Conservation of Atlantic Tunas (ICCAT) (e.g. Sumaila and Huang 2012). The performance of ICCAT in managing those stocks for which they are responsible, such as the Atlantic bluefin tuna, has been unsatisfactory to say the least (Cullis-Suzuki and Pauly 2010). In fact, competitive bargaining has led to TACs becoming so over-inflated that they are now delivering what many see as the institutionalised extinction of a species (Safina and Klinger 2008; Korman 2011). In contrast to those management zones within European waters, Icelandic and Norwegian fisheries were found to have been subject to lower levels of political adjustment than those managed by the EU (Table 1 and Figure 13). Aside from the key differences in their policies on discarding and their use of individual transferable quotas, competitive bargaining is not present in the same force as that which the EU must deal with. Iceland and Norway are members of the European Economic Area rather than the EU and consequently their fisheries are subject to singular rather than multi-jurisdictional management (Johnsen and Eliasen 2011). There are therefore less 'players' with their own interests around the negotiating table competing for quotas and this seems to have delivered greater cohesion between science and politics in these areas, contributing to largely healthier stocks.

The effect of the CFP reform and EU expansion

I found no significant evidence that the CFP reform in 2002 improved decision-making (see Figure 10, Figure 11 and Table 5). In addition, the increased number of stocks for which moratoria were advised (and ignored) indicates declines in stock health and a continuing disregard for a tool of last resort by managers. I argue that this provides ample evidence that the previous reform of the CFP was a failure in sustainability terms.

It is possible that the potential benefits of the CFP reform have not been realised as a result of the expansion of the EU to include ten new member states in 2004. New members will contribute to the TAC decision-making process and may therefore worsen political adjustment by incorporating more delegates, each bringing their own interests to the table (del Valle and Astorkiza 2007). With a further expansion of the EU in 2007 to include Romania and Bulgaria this issue is likely to have worsened. The close timing of the reform and expansion events meant that whilst this is possible it could not be tested for.

Reviews of the 2002 CFP reform have been critical (e.g. Gray and Hatchard 2003; Penas 2007; Symes 2007). One of the main aims of the reform was to tackle the lack of stakeholder participation, highlighted as being an "internal systemic weakness" in the reform Green Paper on the CFP (CEC 2001). In addition, the need to reduce the power that member states exert in the Council of Ministers has long been recognised (Holden 1994; Symes 1997). While the 2002 reform recognised the need to increase stakeholder participation and decentralise authority, it has been argued that instead of accomplishing this, the reform actually reinforced the top-down authority of the European Commission, failing both to devolve power to the lowest competent authorities and to involve stakeholders within decision-making (Gray and Hatchard 2003; Symes 2007). Aims to reduce the powers of the Commission and the Council of Ministers therefore fell by the wayside and policy formation remained entrenched within them (Penas 2007).

A separate issue within fisheries management is the inherent problem of enforcement (Beddington *et al.* 2007). In order for fishery regulations to be successful management needs to encourage compliance. Involving stakeholders in a co-management system has been suggested as a way to achieve this (Browman and Stergiou 2004) although this method has also been heavily criticised (Gray 2005). Assigning rights to shares in the fishery through ITQs aims to tackle the tragedy of the commons problem (Hilborn 2004), as well as removing competitive TACs (Sovacool 2009), and has been successful in promoting resource ownership elsewhere (Sissenwine and Mace 2001; Beddington *et al.* 2007; Eliasen *et al.* 2009). The theory is that by assigning property rights to individual fishers, corporations or communities, the incentives are redirected from increasing fishing intensity to efficient use and conservation of the resource (Ostrom 2008). The use of resource rights and ITQs may therefore help to devolve power and alter the economic rules that currently govern the fishing industry.

2.8. Conclusions and policy recommendations

Political decisions to overrule scientific advice are endemic within the fisheries decision-making process in Europe (and many other management bodies, such as RFMOs). With increasing numbers of moratoria being advised this implies that current decision-making practices and policies are failing to impart sustainability into fish stocks. However, not all species, zones or areas were prone to the same degree of political adjustment. This may be at least partly explained by market prices and regional dependency. The highest levels of political adjustment were generally implemented for those species with higher market prices, while zones most susceptible to adjustment were those with the highest regional dependence on fishing for employment.

The annual negotiation and setting of TACs needs to be changed, moving away from political debates between the Commission and national fisheries ministers. Instead, I advocate that long-term sustainability should be prioritised. When setting TACs scientific advice should be followed, and in setting quotas socio-economic interests should be considered. The use of rights-based management should be considered and adherence to scientific recommendations should be legally binding to remove the temptation to set TACs higher than the ecosystem can support. I also recommend that TACs should be set slightly lower than scientifically advised in order to allow for environmental variability, perturbations in the system and fluctuations in stock level.

Chapter 3.

Is Political Adjustment of Total Allowable Catches a Cause of Fishery Collapse in Europe?

O'Leary, B.C., Smart, J.C.R., Neale, F.C. Hawkins, J.P., Newman, S. Milman, A.C., Roberts, C.M., 2011. Fisheries Mismanagement. Mar. Pollut. Bull.. 62, 2642-2648.

3.1. Abstract

In this chapter I develop a stochastic single species biomass model, extended to include a total allowable catch (TAC) management system to investigate the impact of systematically setting TACs higher than recommended as safe by scientists. I model the effects of such politically-driven decision-making on stock sustainability for two stocks with differing life history characteristics; an early maturing species with a high fecundity and a late maturing species with a low fecundity. My results suggest that political adjustment of scientific recommendations dramatically increases the probability of a stock collapsing within 40 years in both an environment that is highly variable and in one that is more stable. When additional mortality is present in the form of juvenile bycatch the risk of collapse within 40 years increases in some cases to almost 100%. At both levels of environmental variability the risk of collapse is reduced considerably by following scientific advice. Consequently, I propose that historical political adjustment of scientific recommendations has contributed to the overexploitation of European fisheries since the beginning of the Common Fisheries Policy 24 years ago. The importance of basing management targets on precautionary limits is also highlighted.

With 88% of European fish stocks overexploited relative to maximum sustainable yield targets, I conclude that political mismanagement regarding the integration of scientific advice must bear a considerable share of the responsibility for this decline.

3.2. Introduction

In the previous chapter I have shown that political adjustment to scientifically recommended total allowable catches (TACs) within European fisheries has been rife. In the majority of TAC setting decisions taken by the Council of Ministers between 1987 and 2011 (68%), TACs were implemented higher than those recommended by the International Council for the Exploration of the Sea (ICES) and the Scientific, Technical and Economic Committee on Fisheries (STECF). The degree to which ministers adjusted scientifically recommended TACs was between 33-37% above safe recommendations. Supposing scientific advice constitutes reliable and accurate guidance to the number of fish it is safe to remove from a stock, this suggests that the implementation of TACs within the EU could be contributing to the lack of sustainability of fish stocks within EU waters.

This chapter continues the analysis of the impact of political adjustment on TACs through the development of a stochastic lagged recruitment, survival, growth model. I aim to separate the issue of political adjustment from any potential external factors, such as non-compliance (by fishers to TACs), discarding and high-grading, to more readily examine the extent of its impact.

3.3. Existing modelling analyses of the TAC management system and the impact of TACs on fisheries

Whilst there are other studies in the fisheries management literature investigating the issue of political adjustment (Karagiannakos 1996; Daan 1997; Maguire 2001; del Valle and Astorkiza 2007; Reiss *et al.* 2010) few have taken a modelling approach. Cardinale and Svedäng (2008) provide evidence that the short-term perspective of fisheries policy holds a large part of the blame for the failure of the TAC system. Analysing data on stock status, scientific advice and implemented TACs for 18 stocks of gadoids in the North East Atlantic between 1987 and 2005 the deviation between advised and implemented TACs was determined. Using a deterministic single-species model they found that if scientific advice is applied, the stock is maintained at a relatively healthy state. Consequently, they conclude that even in the face of uncertainty within stock assessment, it is the practice of ignoring the advice that has led to the decline in marine resources.

Hamon *et al.* (2007) presented a management strategy evaluation case study of the North Sea roundfish fisheries investigating TACs in mixed fisheries under alternative assumptions of fisherman behaviour. They found that single-stock management objectives cannot be accomplished by TACs because of conflicting incentives to fishing fleets. Using a dynamic state variable model, Poos *et al.* (2010) examined the effect of restrictive quotas on the spatio-temporal allocation and discarding of marketable fish in the beam-trawl fishery for sole and plaice. They found that in a multispecies fishery, raising quota restrictions for one species may result in increased discarding of marketable fish as well as encouraging the practise of high-grading. Constraining the quota of one species was also found to shift fishing effort away from areas with high catches of that species towards areas with profitable catches of other species not constrained by quotas. Restrictive quotas were therefore found to influence the spatial distribution of fishing effort and discarding behaviour, particularly as they allow fishers to continue fishing in a multispecies fishery even if one of the quotas is exhausted. Not only is this detrimental to the fish stocks and the fishers, over-quota discarding also disrupts the link between catches and landings in mixed fisheries and therefore may corrupt the basis of scientific advice and increase the risk of stock collapse (Rijnsdorp *et al.* 2007).

Whilst limited, these studies corroborate statistical evidence (outlined in Chapter 2) that the TAC management system has failed to impart sustainability into European fisheries, at least partly as a result of the process of political adjustment. This research investigates this subject further using stochastic modelling processes with political adjustment based on the average level of political adjustment in decisions since 1987. Since random fluctuations are inherent in populations and their environment, stochastic modelling was applied so as to incorporate chance variation as an integral component of the model, particularly when examining risk. It is hoped that this research will provide further evidence helpful to improving the TAC system during the next reform of the CFP in 2012 (CEC 2009).

3.4. Aims and objectives

The aim of this research was to explore the impact of political adjustment in a fishery over a management timeframe. By analysing the results of the fishery model the following questions may be addressed:

1. What is the risk of collapse for species with different life history characteristics (i.e. early maturing and fecund vs. late maturing and less fecund) over a 40 year timeframe if the scientifically recommended TACs are consistently raised at a number of political adjustment levels?

2. How does environmental variability affect the resilience of each species to overfishing?

3.5. Methodology

A stochastic, single-species biomass model was developed to investigate the impact that historical political adjustment to recommended TACs may have had on the status of European stocks over the period of the CFP. The model was based on a lagged recruitment, survival and growth model (Hilborn and Mangel 1997) and extended to incorporate a TAC management system. Within this model the following assumptions were made:

1. Fishing takes place as a pulse event each year, as the model runs in discrete time.

2. Mature individuals have the same weight, fecundity and survival rate.

3. Recruitment follows the form of the Beverton-Holt stock recruitment curve.

4. There is knife-edge selectivity, i.e. fish reach sexual maturity and become vulnerable to the fishing gear at the same age.

5. Vulnerability to harvest is independent of age above the age-at-maturity.

6. Natural mortality is independent of age above the age-at-maturity.

7. The fishing quota is always filled unless the stock collapses; there is no factor relating ease of capture with density dependence included in the model.

The model used is an approximation of the delay-difference model of Deriso (1980). Delay-difference models provide an intermediate option between age-aggregated and more complicated (age- and size-structured) models, being based on assumptions that allow age-structured dynamics to be simplified to a single equation (equation 3.1). These models are useful when a 'realistic' model is needed that allows for generalised species characteristics but does not need to be species-specific or rely on large amounts of data.

Scenarios considered

The model was run to determine the impact of political adjustment on two stocks with differing life history characteristics. Two types of life history characteristics were examined; an early maturing species with a high fecundity and a late maturing species with a low fecundity. These scenarios were used to investigate the impact of continuously adjusting scientifically recommended TACs by fixed percentages over 40 years, a realistic management timeframe.

The additional impact of juvenile mortality from bycatch and discarding was also considered. In general, demersal fisheries typically experience higher bycatch and discard rates than pelagic fisheries (Alverson *et al.,* 1994), with estimates reaching up to 94% for cod (ICES, 2009) compared to 11% for herring (Pierce *et al.,* 2002). Our modelling used an intermediate bycatch value of 50%. Juveniles were considered to be immature fish which are not subject to the TAC. Juvenile bycatch was introduced into the system through recruitment adjustments during each year of a simulation run.

Model structure:

A lagged recruitment, survival and growth model (Hilborn and Mangel 1997) was used to model the dynamics of two hypothetical fish stocks with different life history characteristics. Fish population dynamics are described in terms of biomass. In its discrete form this model can be expressed as:

$$B_{t+1} = sB_t + R_t - C_t \tag{3.1}$$

Where s describes the change in biomass (B) of the stock from one year (t) to the next ($t+1$) (i.e. survivorship from natural mortality only); R represents the recruitment to the population in year t and C is the catch taken from the stock in year t.

Recruitment is a main driving force of fisheries stock assessment models and is typically modelled by including a random component around the stock-recruitment relationship that describes the deviation of recruitment from the curve (Iles 1994). The levels of recruitment usually drive absolute biomass and consequently are important for forward projections and for calculating management quantities, such as maximum sustainable yield (MSY) (Haltuch *et al.* 2008). The variability within recruitment often contributes substantially to the difficulty, and the uncertainty, in predicting future stock biomass and catch (Iles 1994; Haltuch *et al.* 2008).

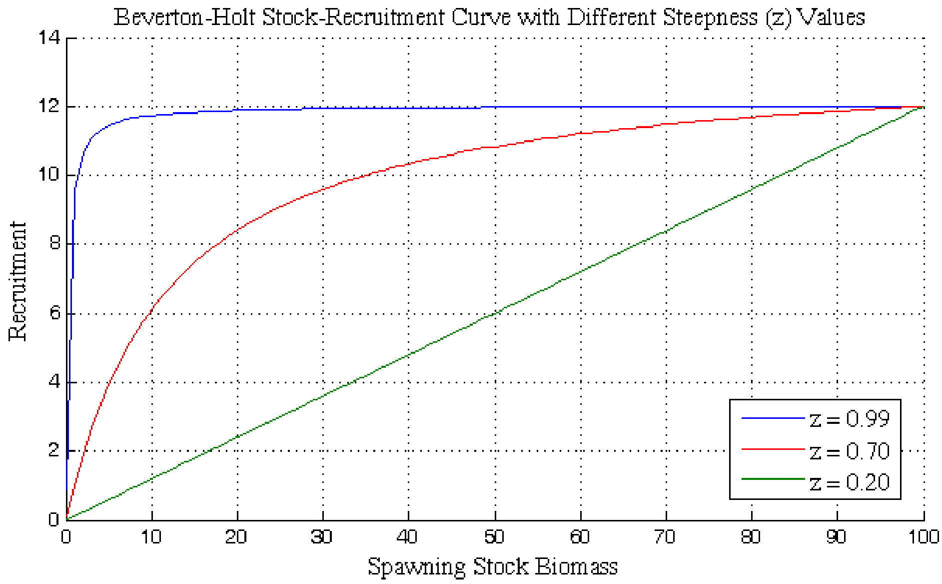
Recruitment is related to spawning stock biomass (SSB), however the relationship is often not clear. At a very low SSB it is likely that there is a strong relationship between recruitment and stock size however at high SSB and for the level of biomass corresponding to management targets, data often suggests no relationship between SSB and recruitment (Needle 2002; Megrey *et al.* 2005). This difficulty in detecting relationships is the result of large variability in recruitment levels caused by other factors than SSB, such as environmental conditions and density dependent factors (reduced resources, increased disease, cannibalism) (Needle 2002; Megrey *et al.* 2005; Cury *et al.* 2008).

There are a variety of stock-recruitment relationships defined within the literature (Beverton and Holt 1957; Cushing 1971; Ricker 1975; Deriso 1980; Schnute 1985; Barrowman and Myers 2000). However the Beverton and Holt (1957) and Ricker (1975) models are most commonly used in fisheries stock assessment models (Cadigan 2009). The Beverton-Holt model represents a near constant recruitment at high SSB and recruitment declines at low SSB and the basic property is that average recruitment constantly increases toward an asymptote as SSB increases. The relationship is based on the assumption that juvenile competition results in a mortality rate that is linearly dependent upon the number of fish alive at any time in the cohort. Consequently, as the spawning stock increases, the individuals disappear faster, e.g. competition for food and space results in fewer recruits (Beverton and Holt 1957).

One drawback to the original Beverton-Holt recruitment curve formula ($R_t = (\alpha B_t)/(\beta + B_t)$) is that any change in the parameters α and β results in a different unfished equilibrium (i.e. carrying capacity). Consequently, the slope of the stock recruitment curve cannot be adjusted to see whether making it steeper affects the behaviour, without also changing the virgin biomass. Following Mace and Doonan (1988) and Hilborn and Mangel (1997) the parameters α and β were therefore redefined to include a steepness parameter (z) to represent how steeply the stock-recruitment curve ascends (see equation 3.4 and 3.5). The term 'steepness' was first defined by Mace and Doonan (1988) and represents the recruitment, relative to recruitment at equilibrium in the absence of fishing, that occurs when spawner abundance has been reduced to 20% of its virgin level (B_0). A high steepness value (z

= 0.99) indicates almost constant recruitment that is essentially independent of the spawning stock biomass, while a low steepness value of (z = 0.20) produces a proportional relationship between recruitment and spawning stock. This method of re-parameterising the stock-recruitment relationship allows the steepness of the curve to be altered while maintaining the same carrying capacity for the environment (Figure 16). Therefore, an investigation can be made into the impact of different z values on the sustainable and unsustainable yields.

Figure 16. The characterisation of the Beverton–Holt stock–recruit relationship using the steepness parameter, z, for z = 0.2, 0.7 and 0.99



Recruitment (R_t) can be expressed as:

$$R_t = \frac{B_{t-L}}{\alpha + \beta B_{t-L}} \qquad (3.2)$$

Here, L refers to the time lag in years between birth and recruitment to the fishery. Recruitment in year t therefore depends on the stock biomass L years earlier. Consequently, two distinct age groups are modelled; recruits, i.e. fish aged less than L, and fish older than L years which are fecund, i.e. capable of producing new biomass.

Fisheries are highly dynamic systems subject to a variety of biological and environmental controls and managed through imperfect science and management errors. Consequently, these systems should be evaluated using stochastic models. Stochasticity (σ) is introduced into the system through recruitment during each year of a simulation run. Recruitment is calculated using a deterministic equation (equation 3.2) which is then multiplied by a value drawn randomly from a uniform distribution spanning –1 to 1 and scaled by a standard deviation value, of which variance is manually adjusted to reach the required stochasticity levels, becoming equation 3.3.

$$R_t = \left(\frac{B_{t-L}}{\alpha+\beta B_{t-L}}\right).(1+\sigma) \tag{3.3}$$

Two levels of stochasticity were investigated, high and low. High stochasticity was set according to the level of variation that led to a 10% chance of stock collapse from B_{MSY} within 20 years. This scenario therefore produces highly variable recruitment which is realistic for many species (Needle 2002). The low level of stochasticity was defined by the variation that led to a 0.5% probability of collapse from B_{MSY} within 20 years, producing a correspondingly low variation in recruitment.

The re-parameterisation of Beverton-Holt to include a z value defines the parameters α and β as (Mace and Doonan 1988):

$$\alpha = \frac{B_0}{R_0}\left(1-\left(\frac{z-0.2}{0.8z}\right)\right) \tag{3.4}$$

$$\beta = \frac{z-0.2}{0.8zR_0} \tag{3.5}$$

where $R_0 = B_0(1-s)$. (3.6)

Within this model the biomass at MSY (B_{MSY}), and the MSY itself, are defined as (Hilborn and Mangel 1997):

$$B_{MSY} = \frac{1}{\beta}\left(\sqrt{\frac{\alpha}{(1-s)}}-\alpha\right) \tag{3.7}$$

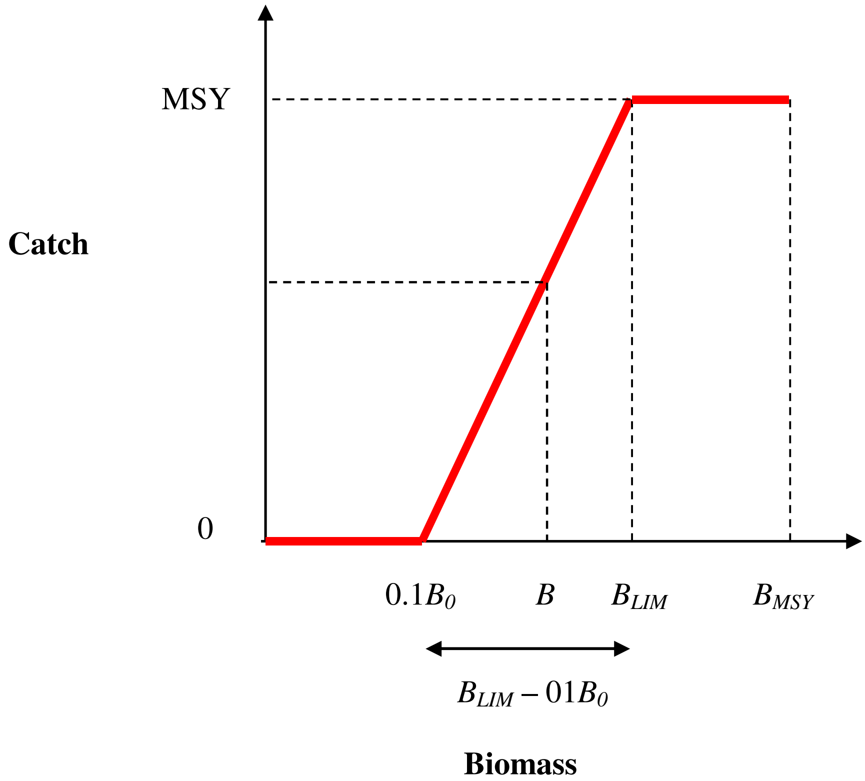
$$MSY = B_{MSY}\left(s-1+\frac{1}{\alpha+\beta B_{MSY}}\right) \tag{3.8}$$

A second dynamic element was integrated into this model by incorporating fluctuating catch quotas in direct response to the stock's biomass in each year to represent the TAC system. Scientific recommendations were taken to be the amount that the catch level needed to be altered (reduced, increased or maintained) in order to maintain stocks at their MSY biomass (B_{MSY}). Should the biomass of the stock fall below a set limit (B_{LIM}) the recommended quotas assumed a proportional decrease (Figure 17). Scientifically recommended TACs were set to MSY above B_{LIM} and then adjusted once the biomass (B) reaches or falls below this level, aiming to rebuild the stock to B_{MSY}. The TAC was calculated by equation 3.9 where B_{LIM} is defined as B_{MSY} and $0.1B_0$ represents the 'collapse' threshold.

$$TAC = \left(\frac{B_t - 0.1B_0 \times MSY}{B_{LIM} - 0.1B_0}\right) \tag{3.9}$$

B_{LIM} was set at B_{MSY} because in a well-managed fishery the target is to maintain the stock at B_{MSY} and therefore once the stock biomass falls below this level the TACs should be adjusted to allow the stock to rebuild to this level. Setting B_{LIM} to B_{MSY} is a conservative estimate of the status of stocks today as in many cases stocks have been driven below this level (Beddington *et al.* 2007). Political adjustment was modelled by multiplying the recommended TAC (equation 3.9) by the desired level, e.g. recommended TAC * 10%. Throughout each simulation the level of political adjustment was held constant.

Figure 17. Schematic diagram representing the TAC system



B represents spawning stock biomass; B_{LIM} *the biomass limit;* B_{MSY} *the biomass level corresponding to MSY; and* $0.1B_0$ *the level at which the population is considered to have collapsed (10% virgin biomass).*

The additional mortality associated with discarding juveniles caught as bycatch within the fishery was also considered as in many fisheries juveniles are taken as bycatch and discarded[22]. Juveniles were considered to be those fish under L years which are not subject to the TAC. Juvenile bycatch is introduced into the model by adjusting recruitment during each year of a simulation run. Recruitment is calculated initially via equation 3.3 and then scaled by the juvenile bycatch rate (j) (equation 3.10).

$$R_t = \left(\left(\frac{B_{t-L}}{\alpha + \beta B_{t-L}} \right) . (1 + \sigma) \right) j \qquad (3.10)$$

Table 6 shows the parameter values used for these simulations.

[22] e.g. In 2008, 94% of 1 year old cod, 73% of 2 year old, 64% of 3 year old and 12% of 4 year old cod, were caught and discarded in the North Sea, Eastern Channel and Skagerrak (ICES advice 2009)

Table 6. Parameter values for simulations

Parameter	Value	Description
Virgin biomass, B_0	100	Also referred to as environmental carrying capacity. Biomass referred to throughout as a percentage of carrying capacity.
Survivorship, s	0.88	Survivorship was taken to be a function of natural mortality, taken as 0.2 based on estimates by Pauly (1980), and growth in mass of surviving individuals each year, taken to be 10%. Units: survival probability per year per individual
Steepness of stock-recruitment curve, z	0.4, 0.7	Represents how steeply the Beverton-Holt stock recruitment curve ascends. No units -dimensionless.
α	0.8929, 3.125	Recruit production parameter
β	0.0744, 0.0521	Recruit production parameter
Lag time, L	2, 5	Lag time (years) between birth and recruitment.
B_{LIM}	B_{MSY}	The biomass level at which the TAC system is implemented.
Population collapse	$0.1B_0$	10% of virgin biomass (Worm *et al.* 2006)
Juvenile bycatch rate, j	0.5	Proportion of juveniles caught as bycatch.

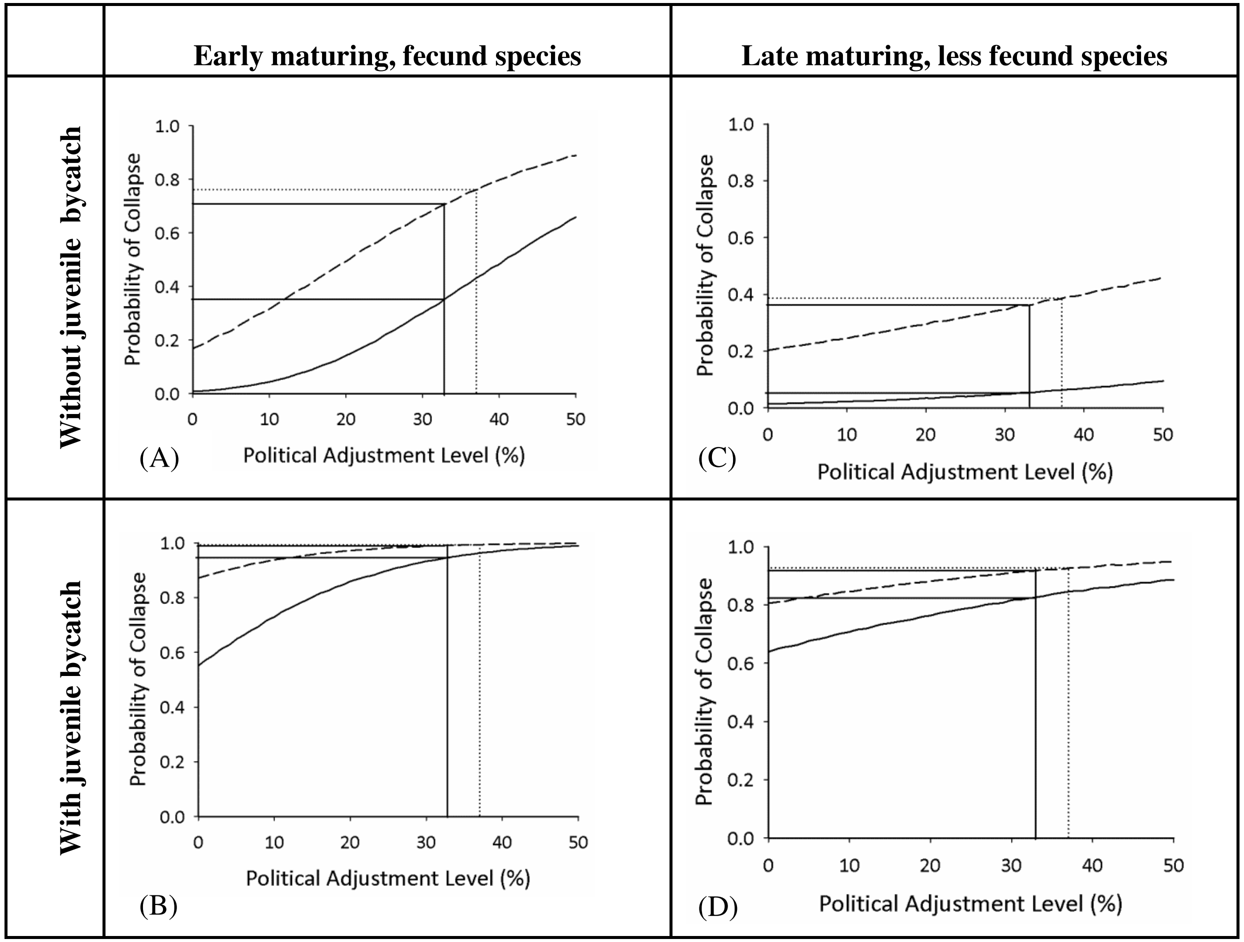
The model was run until the stock collapsed (defined as ≤ $0.1B_0$ (Worm *et al.* 2006)) or for a maximum of 100 years. For each scenario 10,000 simulations were completed. All model simulations were run from an initial biomass of B_{MSY}. This model was implemented in MATLAB (Mathworks 2008).

3.6. Results

Simulations were run for political adjustment levels of 0-50% in 1% increments. To examine the effect of current decision-making practices, results for adjustment levels of 33% and 37% were extracted for two TAC setting scenarios. These two values were chosen as a result of earlier statistical analyses which concluded that on average the adjusted TACs were set 33-37% above catch levels recommended as safe by scientists. In addition, the model was run to examine the impact of including a juvenile bycatch rate of 0.5. It is recognised that in some fisheries, juvenile bycatch may reach levels of 94%[23] and so the value of 50% was chosen in order to act as a conservative estimate.

[23] E.g. In 2008, 94% of 1 year old cod were caught and discarded in the North Sea, Eastern Channel and Skagerrak (ICES advice 2009).

Figure 18. The effect of political adjustment on the probability of stock collapse within 40 years in modelled scenarios.



Probability of collapse for (A) an early maturing species. (B) an early maturing species when 50% of juveniles are caught as bycatch. (C) a late maturing species. (D) a late maturing species when 50% of juveniles are caught as bycatch. All scenarios show lower resilience to political adjustment when subject to high environmental variability (dashed lines) compared to low (solid lines). The probability of collapse at average levels of political adjustment is indicated by the solid (33%) and dotted (37%) vertical and horizontal lines.

Figure 18(A) shows that an early-maturing, highly fecund species has a 36-44% chance of collapse within 40 years in a low stochasticity environment when adjusting the TACs by 33-37%. This risk increases to 71-77% at the higher level of environmental variability. At both levels of environmental variability, the risk of collapse is reduced considerably by following the scientific advice. Within a less variable environment the risk of collapse within 40 years remains under 2% when fishing at the scientifically advised level. At the higher level of stochasticity the risk remains high being 17% even at MSY. However, this is still a considerable reduction from 77%.

A late-maturing, low fecundity species appears to show a higher resilience to political adjustment, with the risk of collapse within 40 years at low stochasticity being only 5-6% and at high being 36-38% (Figure 18(C)). These somewhat counter-intuitive results are discussed later in this chapter. In general however, politically adjusting scientifically recommended TACs is shown to increase the risk of stock collapse whatever the species, sometimes to precariously high levels where only a succession of good recruitment years will maintain the stock.

When juvenile bycatch mortality is included within the simulations, the risk of collapse within 40 years increases as expected, in some cases to almost 100% (Figure 18(B,D)). The early maturing, more fecund species again appears to be more susceptible to political adjustment to TACs. However, both species show between an 83-99% risk of collapse at both levels of stochasticity. Following the scientifically advised TACs leads to a reduction in the risk of collapse to between 55-87%.

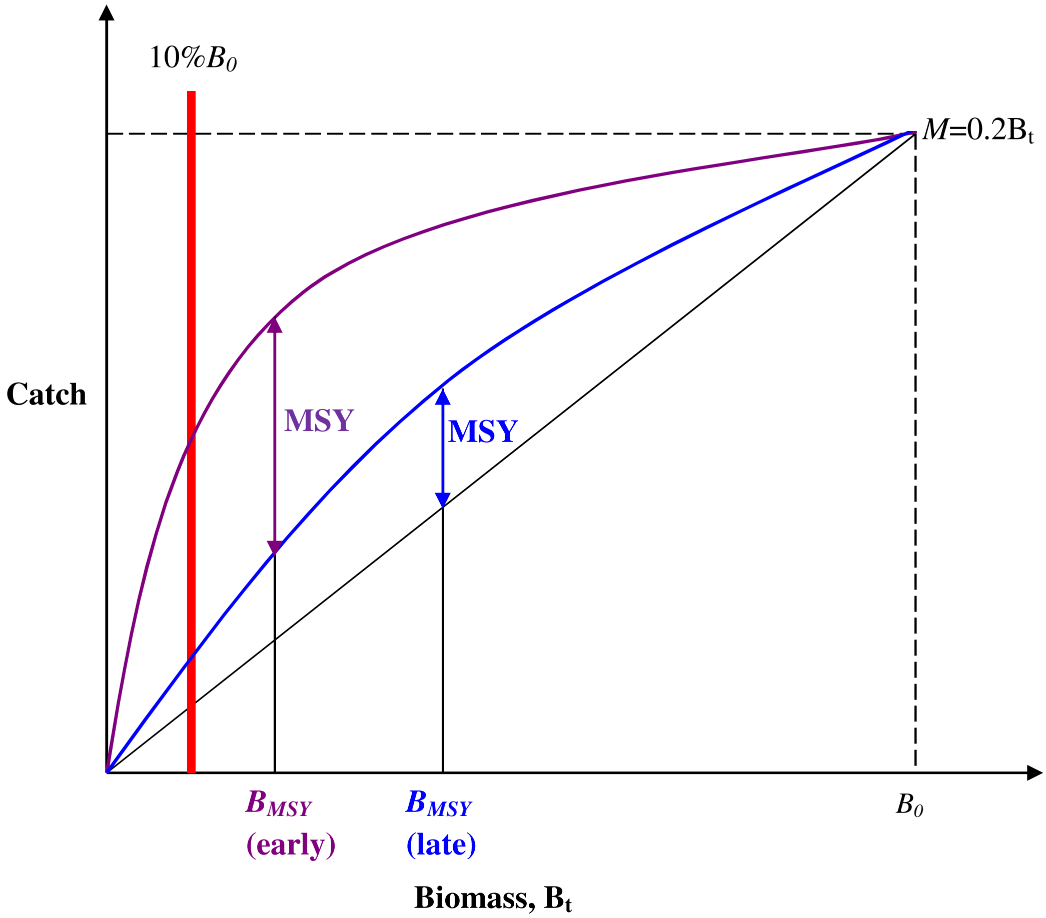
3.7. Discussion

Increasing environmental variability increases the risk of fishery stock collapse even when following scientific advice, no matter the characteristic of the stock. Anthropogenic climate change is likely to increase environmental variability in the future and place increasing pressure on marine species (Hoegh-Guldberg and Bruno 2011). Consequently, the balance between exploiting at maximum sustainable yield and overexploitation will become harder to manage. However, at any level of stochasticity following the scientific advice offers a much more sustainable solution for fishers than when TACs are adjusted upwards. In addition, politically adjusting scientifically recommended TACs is shown to increase the risk of stock collapse whatever the species, sometimes to extreme levels where only a succession of good recruitment years will maintain the stock. Including the bycatch of juveniles into the model simulations increases the risk of species collapse, in some cases to almost 100% within 40 years. This indicates the importance of selective fishing techniques and the need to limit juvenile bycatch within fisheries.

The model shows that political decisions to adjust advised TACs undermines their use and their scientific basis. Results also indicate that basing TACs on MSY targets is insufficient to maintain sustainable stocks under environmental uncertainty and bycatch scenarios. Consequently, failing to follow these advised TACs compounds the problem and moves the risk of stock collapse over 40-year timeframes from likely, when scientific advice based on MSY is followed, to a near certainty, under political adjustment scenarios.

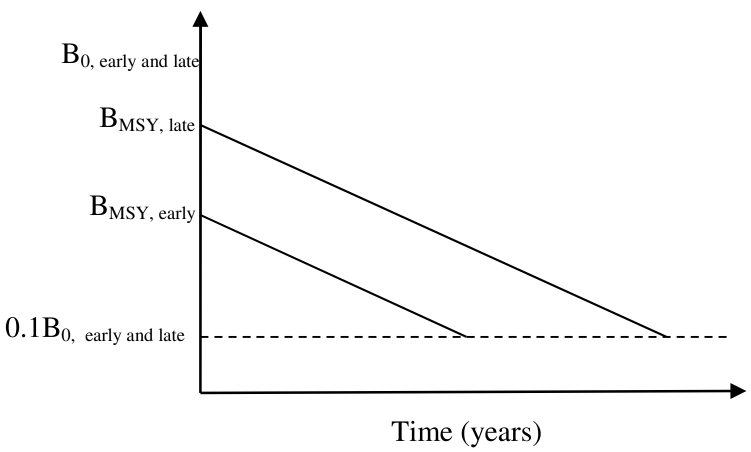
While the general trends are as expected, at first glance the model results appear to be counter-intuitive, with the early maturing more fecund species appearing to be less resilient to political adjustment than the later maturing less fecund species (Figure 18). In fact, *per extra tonnage of catch*, the later maturing, less fecund species *is* less resistant to political adjustment. The model is designed to take into account specific fecundity parameters through the Beverton-Holt stock-recruitment relationship. Consequently the B_{MSY} and MSY targets, together with the subsequent recommended TACs are determined in relation to the stock characteristics. The later maturing less fecund species therefore supports a smaller MSY than a faster maturing more fecund species (Figure 19). Consequently, a percentage increase on the recommended TAC each year to represent the political adjustment level results in a much smaller extra tonnage of catch taken for the later maturing species than the same percentage increase on the TAC for the faster maturing species. In addition, B_{MSY} for the late maturing species is a higher proportion of unexploited biomass than for the early maturing species. Because of this, management of early maturing species is more precarious since the gap between B_{MSY} and a state of collapse is less (Figure 20). The higher starting biomass (B_{MSY}) and the lower absolute MSY for the later maturing species therefore makes it appear as if the stock is more resilient in the time to collapse curves.

Figure 19. Catch biomass curve explaining the model results



Here the line M represents natural mortality at 20% of the stock per year. B_0 is the carrying capacity population. The red line corresponds to 10%B_0, or the level at which the population collapses. The blue line represents the later maturing less fecund species, the purple line the early maturing more fecund species. MSY is calculated to be the 'surplus' population above mortality. The double-headed arrows represent the MSY for each species. Note the difference in the B_{MSY} values, the distance between each B_{MSY} and 10%B_0 and in the size (tonnage) of the MSY for each species.

Figure 20. Relationship between MSY management targets and stock biomass



Any fisheries policy that aims to promote long-term industry viability must promote the sustainable use of fish stocks. Regulations should therefore be based on transparent science-based decision-making rather than a discretionary political decision-making process. Coherent laws and regulations should be established that offer economic incentives to ensure compliance by fishers, and enforcement of the legal framework should be guaranteed. Currently, the CFP fails to achieve this. The lack of transparency of the decision-making process regarding TACs is evident. Furthermore the lack of effective control and enforcement has been highlighted by the Commission itself in its Green Paper; "[f]isheries control has generally been weak, penalties are not dissuasive and inspections not frequent enough to encourage compliance" (CEC 2009).

The mechanisms that enable the joint participation of the EC/EU and its member states in multilateral agreements are complex. Within fisheries, competences for negotiation of total allowable catches and other management strategies are mixed (i.e. authority is shared between member states and the European Commission, known as "mixity" (McGoldrick 1997; Leal-Arcas 2004)). In the case of fisheries member states have determined the extent to which the EC may enter the bargaining process. To date, this is limited to the formulation of proposals for TACs based on scientific advice from ICES and STECF. Mixity is a common and widespread practice in EU policy, especially in areas of outstanding transdisciplinary character (Frank 2007). However, this may result in conflicting relationships between EC and domestic law. Presumably, it is because of short-term and domestic political considerations (e.g. re-election probability, promotion probability, influence of interest groups) that TACs are consistently set higher than those recommended by ICES (Khalilian *et al.* 2010). Even after the 2002 reform of the CFP, TACs systematically exceeded those suggested by scientists (Daw and Gray 2005, and see Chapter 2). This unwillingness to integrate scientific advice into TAC decision-making practices is at least partly driven by the threat of short-term economic hardship and increased unemployment in the fisheries sector overriding the concern about collapsing fish stocks in the long run (Roberts *et al.* 2005; Patterson and Résimont 2007). However, even political economists emphasise that long-term rules generally outperform discretionary decision-making (Kydland and Prescott 1977; Franchino and Rahming 2003). The Commission has itself concluded that the decision-making process of the CFP needs to be brought in line with other EU policies with a "clear hierarchy between fundamental principles and technical implementation" (CEC 2009). In addition, relative stability contributes to political pressures to raise TACs. The principle of relative stability (established in 1983) provides a distribution guideline for quotas between the Member States (Princen 2010). The Commission states that this "creates inflationary pressure on TACs because a Member State that wants a higher quota has no other option but to seek an increase of the whole Community TAC" (CEC 2009).

The status currently afforded environmental issues in the European agenda is significant[24]. As fishing is a factor of ecological disturbance within marine ecosystems the incorporation of an increasing number of environmental provisions and conservation policies in fisheries agreements and policy programmes has been seen (Princen 2010). TACs have been identified as the cornerstone of conservation policy within the Common Fisheries Policy (CFP) (Karagiannakos 1996). However, mixity within TAC negotiations has resulted in individual actors trying to maximise their allocation, often (in the case of TACs) to the detriment of the purpose of the community (i.e. the inflation of scientifically recommended TACs).

In order to overcome this, effort regulation has been suggested as a viable alternative to the quota system (Rossiter and Stead 2003; Cotter 2010). The major advantage that effort limitations (e.g. days-at-sea, fleet capacity) have over TACs and quotas are the ease by which they may be monitored. However, effort regulation often triggers technological progress to move in a detrimental direction ('technological creep'), i.e. fishers may develop methods to catch more fish in shorter time periods (Baudron *et al.* 2010). Consequently, effort regulations have to be constantly adjusted to account for fishing power. As a TAC system is independent of technology level they only need to be adjusted to recruitment and stock size. Under the CFP, fishers are currently forced to discard fish caught over-quota or under-sized resulting in additional, and unaccounted for, fishing mortality (Johnsen and Eliasen 2011). As discards are often not recorded this unaccounted mortality undermines the effectiveness of the TAC system and the credibility of the CFP's sustainability goal. An individual transferable quota (ITQ) system, combined with the prohibition of

[24] The EU currently has over 600 pieces of legislation that encompass the environment. Summaries of all legislation may be found at http://europa.eu/legislation_summaries/environment/index_en.htm

discards may therefore be suitable to solve the problem of relative stability. This would reinstate the link between landings and mortality improving the accuracy of stock assessment science. While TACs should be set according to scientific advice (taking into account mixed species and ecosystem impacts) and the initial allocation of fishing quotas according to the principle of relative stability, ITQs would allow quota's to be redistributed according to market values.

The results of this research serve to enhance that of the previous chapter and expand the modelling literature of the TAC system and the impact of political adjustment. They show that even when all other factors are optimal (i.e. compliance by fishers to TACs, no discards or bycatch) political adjustment leads to declining stocks and increases the risk of fishery collapse.

Model limitations

This model assumes that scientists have perfect information about the status of the stock biomass every year. The knowledge of scientists regarding stock status depends on the predictability of the stock-recruitment relationship as well as on the accuracy of catch reporting and the available information on discard rates. Consequently, the accuracy of recommended TACs is fraught with uncertainty and subject to potentially significant errors (Kraak *et al.* 2010; Hauge 2011). This model fails to take this uncertainty into account. A stochastic element could have been added to the recommended TACs every year in order to account for this, however this was omitted in order to present a 'best-case' scenario.

A discrete time model was used to describe fishery dynamics, however natural mortality and exploitation occur continuously throughout the year. Consequently, these processes would be more appropriately modelled in continuous time. Continuous time was not chosen as it was deemed unnecessary for the purposes of this model and because of the increased computational complexity it adds. As few differences between discrete and continuous time models of fishery dynamics have been found in long run equilibrium results (e.g. Hannesson 1998) I felt that the use of continuous time was unwarranted.

This model examines the general state of spawning stock biomass of a single species by considering the whole population to have generalised life characteristics of recruit production and survival. Large (and therefore the more fecund) fish are more vulnerable to fishing and consequently one impact of exploitation is to reduce the reproductive capacity of a stock (Pauly *et al.* 2002; Kaiser *et al.* 2005; Conover *et al.* 2009). In reality this may help to drive stocks down faster as a result of fewer recruits. However, this model cannot be used to investigate the change in reproductive capacity of a population as a result of selective fishing.

Political adjustment levels were set deterministically at a fixed value every year. In reality, the amount by which TACs are adjusted from those recommended each year varies considerably. A stochastic element could have been added to simulate this, however, in order to facilitate direct comparison between political adjustment levels this approach was not taken. In addition, fishers were only allowed to take the agreed TAC. Therefore, no considerations of the practice of discarding, high-grading or illegal landings were made. When juvenile bycatch was considered this only served to drive the stock down faster and a reasonable assumption would be that other wasteful practices encouraged by the TAC system would lead to similar results. This could be investigated in the future.

Within the model stochasticity is set to increase linearly with biomass. Within the literature stochasticity is often normally distributed around the deterministic behaviour and scaled by the square root of the size of the deterministic (in this case recruitment) equation (McKane and Newman 2005; Datta *et al.* 2010). Using this scaling means that as population size increases, the absolute magnitude of the variability increases although relative magnitude decreases (i.e. there is more fluctuation in numbers with increasing population size although the effect is less noticeable due to the larger population size). Upon reflection, this may be a more appropriate way to add stochasticity into this model. Appendix 4 details my response to a comment by Cook *et al.* (in review) on the published article which presented these results. In this, cod and herring are taken as case studies and stochasticity is applied using a lognormal distribution.

3.8. Conclusions and policy recommendations

Fishers are faced with multiple risks to their business; the unpredictability of future catch rates, prices and costs as well as dependence on management decisions, many of which are highly erratic. While the former are largely beyond the control of fisheries managers, effective fisheries management should reduce uncertainty about future catches and increase the sustainability of stocks. Since the introduction of the CFP and TAC system, the integration of scientific knowledge into fisheries management decisions has been shown to be haphazard and often disregarded completely (Chapter 2). Without the integration of scientific advice in decision-making there is little chance that new approaches to management such as the ecosystem approach to fisheries management will achieve their objectives. The results obtained using this model show that political decisions not to follow scientists' recommendations are a leading cause of increased risk of stock collapse. This implies that political adjustment of TACs contributes to the CFP's failure to secure sustainable fish stocks. Consequently, I propose that historical political adjustment of scientific recommendations has contributed to the overexploitation of European fisheries since the beginning of the Common Fisheries Policy 40 years ago.

The key lesson to take from this and previous research, is that ignoring scientific advice translates into stocks being largely overexploited while the reverse does not. The lack of an adequate governance system to implement TACs is one of the main obstacles to sustainable fisheries policy. While the 2002 reform failed to tackle these issues it is hoped that the upcoming reform in 2012 will address its governance failures. However, despite the many criticisms of the TAC system, it is likely to remain in European fisheries management within the reform of the CFP in 2012 for two reasons; firstly, the TAC system represents one of the key components of fisheries management throughout the world and secondly, it forms the basis of the ITQ system, seen as one of the shining hopes for incorporating sustainability into fisheries (del Valle and Astorkiza 2007). There is consequently a need to find mechanisms to minimise the effect of perverse external pressures in taking the final TAC decision. Unless this issue is resolved, the second reform of the CFP is likely to be as ineffective as the first, and how embarrassing would it be if the 2020 review of the CFP came out with the same conclusions as the 1994 review? That, "on the basis of whether the conservation policy has achieved its political objectives, the conservation policy can only be adjudged a total success. [...] In contrast, it has been an almost total practical failure" (Holden 1994p. 167). Political success being the status quo approach to managing fish stocks and the pacification of conflicts between European states regarding access to fishing grounds. Practical failure with regards to the protection and conservation of endangered fish stocks and manage other stocks sustainably.

Chapter 4.

The First Network of Marine Protected Areas (MPAs) in the High Seas: The Process, the Challenges, and Where Next

O’Leary, B.C., Brown, R.L., Johnson, D.E., von Nordheim, H., Ardron, J., Packeiser, T., Roberts, C.M., 2012. The first network of marine protected areas (MPAs) in the high seas: the process, the challenges, and where next. Mar. Policy. 36, 598-605.

4.1. Preface

The previous two chapters have highlighted one difficulty faced when managing fisheries in a political system. This chapter presents a success story of the harmonisation of a political and scientific process to achieve a common goal in order to show how the systems may complement each other.

In the instance presented here, the combination of a strong political mandate and a scientific basis led to a significant outcome - the establishment of the world's first network of high seas marine protected areas. Without one or the other this outcome is unlikely to have been achieved.

The remainder of this chapter takes the form of a paper published in Marine Policy (O'Leary *et al.* 2012).

4.2. Abstract

Marine protected areas (MPAs) are increasingly being established to protect and rebuild coastal and marine ecosystems. However, while the high seas are increasingly subject to exploitation, globally few MPAs exist in areas beyond national jurisdiction. In 2010 a substantial step forward was made in the protection of high seas ecosystems with 286,200 km^2 of the North-East Atlantic established as six MPAs. Here a summary is presented of how the world's first network of high seas marine protected areas was created under the OSPAR Convention, the main challenges, and a series of key lessons learned, aiming to highlight approaches that also may be effective for similar efforts in the future. It is concluded that the designation of these six MPAs is just the start of the process and to achieve ecological coherence and representativity in the North-East Atlantic, the network will have to be complemented over time by additional MPA sites.

4.3. Introduction

In September 2010 OSPAR ministers from 15 European nations took an unprecedented step and established the world's first network of marine protected areas on the high seas. They declared six protected areas[25] that together cover 286,200 km^2 of the North-East Atlantic, larger than the combined land area of OSPAR's six smallest Contracting Parties[26]. In this paper, how this achievement was accomplished is described.

Spatial planning has become one essential tool to manage human activities and conserve the marine environment (Ardron *et al.* 2008; Gaines *et al.* 2010). As part of a suite of management measures many national governments are committed to establishing networks of MPAs under both national and international law and agreements[27]. This commitment to establish MPAs is driven by international concern. Multilateral agreements promote MPAs as a measure to conserve and protect marine biodiversity, help reduce the decline of biomass of the oceans and the risk of fisheries collapse and to ameliorate the negative impacts of human activities such as cable laying (CBD 2008; Kar and Chakraborty 2009; Gaines *et al.* 2010). Legally, areas beyond national jurisdiction (ABNJ) are composed of the High Seas (waters beyond the zones of national jurisdiction[28]) and the Area (the seabed, ocean floor and subsoil thereof beyond the limits of national jurisdiction[29]). For those ABNJ, the United Nations General Assembly (UNGA) Resolutions of 2004 (59/25), 2006 (61/105) and 2009 (64/72) in particular have driven the urgency for the protection of vulnerable marine ecosystems (VMEs) from destructive bottom fishing. In response to these resolutions some regional fisheries management organisations (RFMOs), such as the North-East Atlantic Fisheries Commission (NEAFC), have

[25] The Charlie-Gibbs South MPA, Josephine Seamount High Seas MPA, the Mid-Atlantic Ridge North of the Azores High Seas MPA, Altair Seamount High Seas MPA, Anitaltair Seamount High Seas MPA and the Milne Seamount Complex MPA.

[26] Listed in ascending size order: Luxembourg, Belgium, Netherlands, Switzerland, Ireland, Portugal.

[27] These include commitments to establish representative networks of MPAs by 2012 at the World Summit on Sustainable Development (WSSD) in 2002, and subsequent United Nations General Assembly (UNGA) resolutions and Convention on Biological Diversity (CBD) decisions. In particular the latest target by the CBD is that "By 2020, at least 17 per cent of terrestrial and inland water, and 10 per cent of coastal and marine areas, especially areas of particular importance for biodiversity and ecosystem services, are conserved through effectively and equitably managed, ecologically representative and well-connected systems of protected areas and other effective area-based conservation measures, and integrated into the wider landscapes and seascapes" (target 11, http://www.cbd.int/sp/targets/).

[28] United Nations Convention of the Law of the Sea (UNCLOS) Article 1(1)(1)

[29] UNCLOS Article 86

adopted spatial conservation measures to protect VMEs from bottom trawling[30] (Benn *et al.* 2010; NEAFC 2010).

Effective spatial planning of marine areas relies on spatially explicit data and knowledge in order to define boundaries and designate areas that are biologically important and socially and economically acceptable (Costello *et al.* 2010). Systematic science-based approaches for MPA selection are encouraged, with varying degrees of implementation, in an effort to achieve conservation objectives at a low cost (Margules and Pressey 2000; Ban *et al.* 2009). However, often the necessary data are few, particularly in ABNJ as these regions have been much less studied compared to habitats closer to coastlines (Harris *et al.* 2007; Howell 2010; Auster *et al.* 2011). Nonetheless, there is mounting evidence in ABNJ, as well as from Exclusive Economic Zones (EEZ) or near-shore waters to indicate that present human activities are causing serious damage to a wide variety of habitats in ABNJ (e.g. Schlacher *et al.* 2010). In addition, many deep-sea species possess characteristics such as slow growth and late maturity, which make them extremely vulnerable to fishing (Cheung *et al.* 2007). Deep-sea biogenic habitats also show high vulnerability to extractive human activities due to the slow growth rates and extreme longevity of their constituent species (Roberts 2002). As resources from shallower marine environments become further depleted, deep-sea exploitation is increasingly more attractive and feasible due to technological advances (UNEP 2007) strengthening the imperative to conserve biota at risk from fishing and emerging anthropogenic activities such as deep-sea mining.

The North-East Atlantic is considered to be heavily impacted by human activities (Halpern *et al.* 2008; Benn *et al.* 2010). Within this area the OSPAR Commission (OSPAR)[31] has an obligation and a mandate to protect marine biodiversity. Acting under the overarching legal framework of the United Nations Convention on the Law of the Sea (UNCLOS)[32] OSPAR is an example of regional seas cooperation, whereby States can collectively decide to adopt measures to protect the marine

---

[30] See http://www.neafc.org/managing_fisheries for more information regarding current closed areas to bottom fishing.

[31] The 1992 OSPAR Convention (entry into force 1998) consolidated the 1972 Oslo Convention (to control pollution from dumping) and 1974 Paris Convention (to control pollution from land-based sources and has a mandate to protect and conserve the marine environment of the North-east Atlantic.

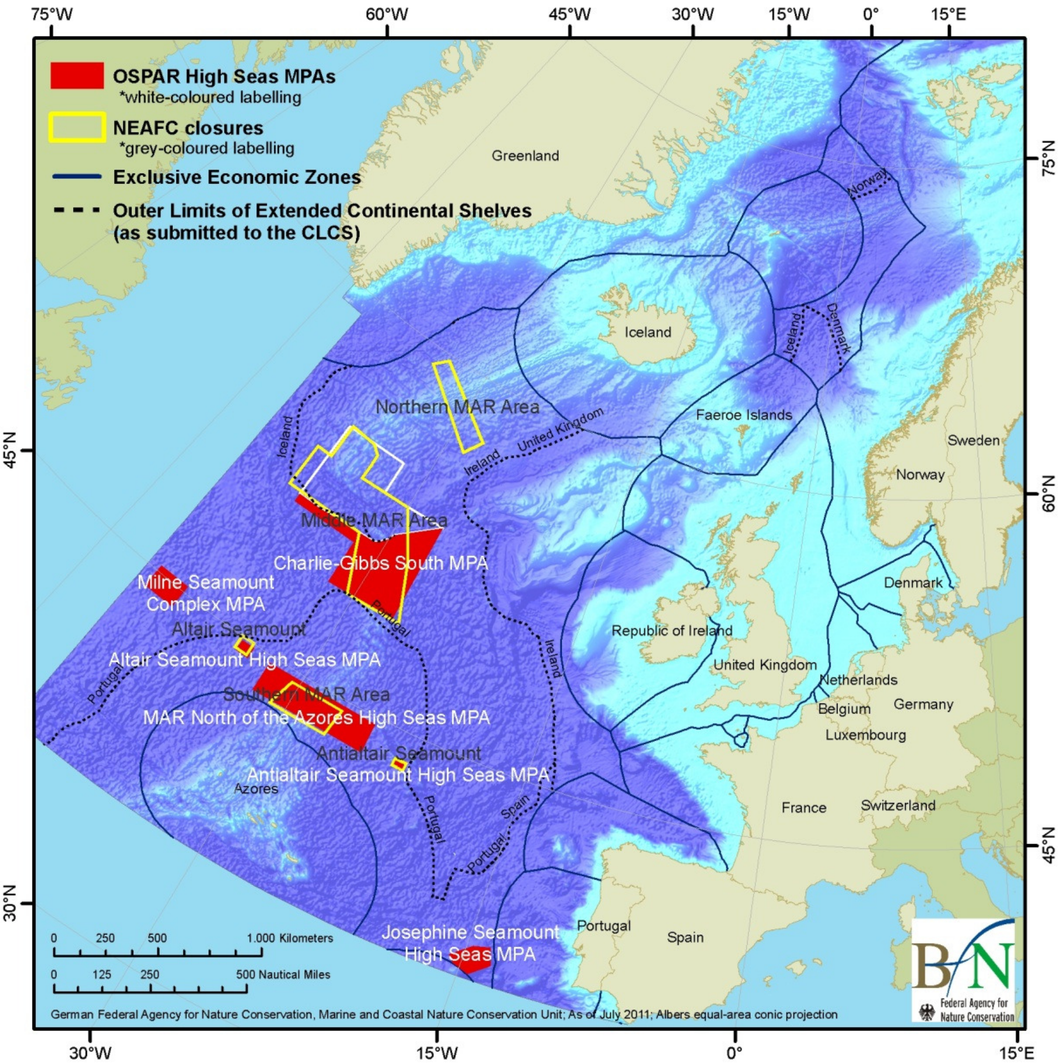
[32] United Nations Convention on the Law of the Sea, Montego Bay, 10 December 1982, in force 16 November 1994 , 1833 *United Nations Treaty Series* 396; www.un.org/Depts/los

environment. In 1995 the Convention on Biological Diversity (CBD) Jakarta Mandate on Marine and Coastal Biodiversity obliged Parties to establish a global network of MPAs (CBD 1995). This commitment was further elaborated to include the deadline of 2012 for the establishment of representative networks as agreed by the World Summit on Sustainable Development (WSSD) in 2002. In order to contribute to this target a joint OSPAR and Helsinki Commission (HELCOM) agreement was adopted in 2003 to create an "ecologically coherent[33] network of well-managed MPAs" by 2010[34]. Approximately 40% of the OSPAR maritime area falls within ABNJ and as such, this commitment included a clear remit to identify and designate MPAs in ABNJ. However, the pursuit of MPAs within the national waters of each Contracting Party preceded those in ABNJ, mainly for pragmatic reasons; i.e. there is clear national ownership and therefore responsibility for conservation. How to go about designating protected areas in ABNJ was less clear, though it was recognised as necessary due to the huge offshore gap in representativity and ecological coherence that resulted from the approach adopted (OSPAR 2007).

[33] A definition of ecological coherence of MPA network has been agreed by the meeting of the working group on MPA, Species and Habitats (MASH) in Norway, 5-8 October 2004. The network should therefore be based on 14 recommendations including key criteria (e.g. connectivity of MPAs, representativity of critical habitats and species, etc.)

[34] OSPAR Recommendation 2003/3; Record of the Joint Ministerial Meeting of Helsinki & OSPAR Commissions 2003.

Figure 21. The current network of OSPAR MPAs and fishery closures implemented by the North East Atlantic Fisheries Commission



Map courtesy of Mirko Hauswirth of the German Federal Agency for Nature Conservation (BfN)

4.4. Establishing the scientific case for protection and gaining political support

As early as 2000, the Worldwide Fund for Nature (WWF), acting as an Observer Organisation within OSPAR, campaigned to protect sites in ABNJ within the OSPAR maritime area (Christiansen 2006). WWF conceived and presented a proposal for the Charlie-Gibbs Fracture Zone (CGFZ), a large area of the Mid-Atlantic Ridge, to be protected on account of its vulnerability to human activities. To be taken forward, OSPAR Rules of Procedure (OSPAR 2005) require any Observer proposal to be supported by a Contracting Party. In 2007, the Netherlands co-supported the proposal to consider the CGFZ as a 'pilot' in order to provide the impetus to build up the scientific case according to the criteria and conservation principles established by OSPAR and other international fora (e.g. the Food and Agriculture Organisation (FAO) and CBD)[35].

Under the auspices of OSPAR's expert group on MPAs, the latest scientific findings were used to strengthen the original WWF proposal, and scientific experts advised on the application of the agreed set of OSPAR ecological selection criteria (Czybulka and Kersandt 2000). The development of a comprehensive background document for a CGFZ MPA (Christiansen 2006) convinced more Contracting Parties to support and co-sponsor the proposal in 2008. At this time the OSPAR Commission also agreed on a roadmap setting out considerations and steps leading up to the possible adoption of MPAs in ABNJ in 2010[36]. Specifically, this roadmap was drawn up in connection with the CGFZ MPA; however, by analogy it provided a useful framework for any future proposal.

OSPAR's competencies for management of human activities include scientific research, cable-laying, dumping of waste, construction of installations and artificial islands and deep-sea tourism but do not extend to fishing, mining or shipping (Czybulka and Kersandt 2000; Owen 2006). Consequently, in order to create a network of MPAs in ABNJ it is essential for OSPAR to work with other international organisations that have a legal competence over activities within their Regulatory Area. The roadmap set out timeframes for work to be carried by the relevant OSPAR bodies as well as the premise for involving other Competent Authorities[37].

With the deadline of 2010 fast approaching the requirement to select sites for MPAs in ABNJ had become more obvious and urgent. In 2007, Germany, as convenor of the OSPAR MPA group, commissioned a scoping report from the University of York in the UK to investigate potential sites for high seas MPAs in the wider Atlantic Region. The aim of this report was to identify sites representative of the various biogeographic areas within the Wider Atlantic Region. This report identified

[35] Subsequently, in 2008 France, Portugal and Germany also agreed to support the proposal.
[36] Summary Record OSPAR 2008; (OSPAR 08/24/1-E); www.ospar.org
[37] See: *General outline of roadmap for further work on the Charlie-Gibbs Fracture Zone (CGFZ)/Mid Atlantic Ridge proposal 2008/09* (*ibid.* Annex 10)

eight further potential MPA sites[38] by reviewing scientific literature, mapping significant and vulnerable marine habitats, consulting with scientists familiar with the region, and prioritising areas currently within reach of serious impact (mainly fishable depth zones; 2000m delimits the maximum fishing depth of the predominant past and present fishing activities (Bailey *et al.* 2009). On the basis of this work, in 2009 OSPAR accepted 'in principle' the scientific case and conservation objectives for seven potential MPAs (the Rockall and Hatton Banks proposal having been set aside in 2008 due to unsettled ownership disputes), which included the CGFZ whose boundaries were enlarged from the WWF proposal to capture a wider range of habitats off the Mid-Atlantic Ridge[39]. Essential to maintaining the momentum throughout this process was the role of a lead agency and political 'champion' to move MPAs in ABNJ higher up the political agenda, to facilitate collaboration and to overcome difficulties or delays in the implementation of the MPAs. The CGFZ proposal owes a lot to the early and lasting support of the Netherlands, and subsequently other OSPAR Contacting Parties. Consequently Germany, as the convenor of the OSPAR MPA group since 1998, ensured that the momentum of the process was maintained as well as providing funding for the external scoping study leading to the additional area proposals.

Meanwhile, in accordance with the roadmap, OSPAR sought to formalise working relationships with other key competent authorities including the International Maritime Organisation (IMO), the International Seabed Authority (ISA) and the North-East Atlantic Fisheries Commission (NEAFC). The adoption of formal Memoranda of Understanding[40] between these organisations strengthened attempts to broker a prospective ‘collective arrangement’ for the potential management of selected areas in ABNJ.

This process ultimately led to a significant political outcome with the first network of MPAs in the high seas being designated in the course of the OSPAR Ministerial Meeting (20-24 September 2010; Bergen, Norway). Throughout the process a number of scientific and political challenges were encountered. Below these

[38] Reykjanes Ridge, Southern Mid-Atlantic Ridge, Charlie-Gibbs Fracture Zone, Rockall and Hatton Banks, Altair seamount, Antialtair seamount, Milne seamount and Josephine seamount.
[39] The CGFZ was approved "in principle" (without conservation objectives) a year earlier.
[40] See www.ospar.org “International cooperation”.

challenges are discussed and lessons learned from this pioneering process are highlighted in order to inform future MPA designations in ABNJ.

4.5. Scientific challenges to site selection and nomination proforma

4.5.1. Data deficiency

Scientific knowledge of biodiversity is limited. Only a fraction of the planet's species have been formally described and only fragmentary information about the geographical distributions of most species exists (Brito 2010). Within the high seas available data are limited to a few areas and species of known interest, to nautical maps (i.e. depth contours), to satellite measurements (i.e. sea surface height anomalies, oceanographic and primary productivity data) and to broad biogeographic regions (Harris *et al.* 2007; Howell 2010).

With the availability of ecological data so limited, the utility of a science-based approach to MPA selection lies in the proposal of potential sites that warrant protection either based upon specific, sometimes fragmentary knowledge, if available, or through reasonable inference from similar sites. In the extensive review of the North-East Atlantic carried out by the University of York and collaborators, the available ecological data were insufficient in most areas to conclusively support the nomination of specific sites of conservation importance. For example, few scientific studies mention the Milne seamount complex by name and little biological information is available (OSPAR 2010b). It was nominated and protected largely on the basis of inferred importance to biodiversity from similar habitats and places around the world, from its isolated position relative to the Mid-Atlantic Ridge, and from the known vulnerability to fishing of similar habitats in other places, some of them nearby (e.g. Corner Rise seamount (Waller *et al.* 2007)). However, although there was a significant lack of biodiversity knowledge for the majority of areas in the North-East Atlantic, the project MAR-ECO[41], an element of the Census of Marine Life, was completed during the timeframe of these proposals (2004-2010). This project aimed to enhance understanding of the occurrence, distribution and ecology of animals and animal communities along the Mid-Atlantic Ridge between Iceland and the Azores. The knowledge gained from this research was invaluable in making

[41] http://www.mar-eco.no/

the scientific case for the proposed MPAs[42]. Nevertheless, within the remainder of the North-East Atlantic much flexibility existed as to where to place the MPAs to protect both known and unknown biodiversity and features effectively.

When data are poor, scoping can be informed through stakeholder engagement, synthesis from disparate or informal sources, and understanding from similar ecosystems. If scientific data alone are relied on to make the case for protection, then the MPA network is unlikely to take into account known, but poorly documented areas of ecological importance, as well as a full range of representative sites. In part, this is the result of research becoming focused on certain areas of specific interest, often with the advantage of existent time-series data, rather than new areas or those that are more common (Sastre and Lobo 2009). Consequently, sites would be likely to be protected because they are well-studied, rather than because they are representative of the habitat or biodiversity or threatened by potential adverse effects of human activities. In the case of the OSPAR MPA network available information was often greater for sites subject to fishing, so sites prioritised according to these data were mostly those at greatest risk of impact.

This flexibility also applied to the determination of boundaries for sites proposed, particularly as conditions in the marine environment are dynamic. When selecting MPA dimensions and boundaries consideration was given to the size of the feature being protected, the vulnerability of the feature, the potential acceptability of the boundaries and their practicality and enforceability. Vulnerability of the area was determined according to the depths affected by current and potential future fishing practices. To increase international acceptability for proposed MPAs sites were also chosen taking into account the closures to bottom fishing by the North East Atlantic Fisheries Commission[43] (Figure 21). In parallel to OSPAR, NEAFC was also developing site proposals for additional closures in the North-East Atlantic in recognition of the UN Resolution 61/105. During the selection process a member of NEAFC was also a member of the OSPAR MPA group. This allowed for the delivery of a more coherent high seas network, with the final set of MPA sites

[42] Results from the MAR-ECO project have been presented in 22 papers published in two special journal editions: Deep-Sea Research II (2008), 55(1-2):1-268 and Marine Biology Research (2008), 4(1-2):1-163.
[43] At the time these proposals were developed the closures were located on the Hatton Bank, Logachev Mounds, North-west Rockall and the West Rockall Mounds, the Altair Seamount, Antialtair Seamount, Fraday Seamount, Hekate Seamount and a section of the Reykjanes Ridge.

established from NEAFC and OSPAR being closely aligned geographically (Figure 21). In order to fulfil practicality and enforceability criteria, boundaries were kept as straight as possible to simplify management and compliance within proposed protected areas.

4.5.2. Criteria for selection

In 2003 OSPAR adopted scientific criteria to guide the identification and selection of MPAs in the OSPAR maritime area as well as scientific guidance for designing representative networks of MPAs (OSPAR 2003a, b). Following this, in 2009 the Conference of the Parties to the CBD adopted their own set of scientific criteria to identify ecologically or biologically significant marine areas (EBSAs) in need of protection[44]. These criteria include naturalness, biogeographic importance, ecological importance, scientific importance and practicality/feasibility. Based on these evidence was gathered in order to identify important sites and to develop detailed proposals for each candidate MPA.

The prior development and adoption of scientific guidelines and criteria for selecting MPAs in ABNJ was essential to the successful designation of the areas. Without them, it would not have been possible to produce acceptable proposals and gain political agreement. However, when researching and writing the proposals, the question of weighting among criteria became apparent. OSPAR states that "an area qualifies for selection as an MPA if it meets several but not necessarily all of the [...] criteria" (OSPAR 2003b). Whilst this implies that not all criteria must be met for an area to be considered ecologically or biologically important, and maintaining flexibility in this has its own advantages, in practice it turned out that it was better to provide evidence that a site could meet many of the criteria.

4.5.3. How much evidence is enough?

Whilst establishing the scientific case for each MPA the question of how much evidence was sufficient to justify protection of a site was raised. No guidelines exist regarding the quantity or quality of supporting evidence so the proposals written were test cases for OSPAR Contracting Parties with the aim to make them as comprehensive as possible (based on the best available expert knowledge). The sites

[44] Decision IX/20, annexes I & II. Marine and Coastal Biodiversity (2008)

chosen for nomination were selected largely as a result of the greater volume of scientific knowledge available for these areas, as well as assessment of the need for protection based on vulnerability to existing and potential future impacts. Future proposals for high seas MPAs in the North-East Atlantic or elsewhere, particularly those addressing gaps in representativity, are likely to be subject to greater levels of inference if there is less specific knowledge of the areas. It is therefore important that this issue be recognised when future proposals are received.

4.6. Legal and political challenges

4.6.1. Scientific uncertainty

Gaining political support for data-poor management in the high seas is imperative. The paucity of data in the high seas will not be solved in the near future (Harris *et al.* 2007; Howell 2010) and scientific research usually lags behind exploitation (e.g. Brewin *et al.* 2007). With concern over human impacts increasing (Benn *et al.* 2010), conservation actions cannot wait for the availability of completely reliable and comprehensive datasets on biodiversity and the impacts from human activities, especially since such impacts may be irreversible in the deep sea on meaningful human timescales (Roberts 2002; Waller *et al.* 2007). Under the umbrella of the precautionary principle, which is explicitly set out in international conventions and agreements including the law of the sea (UNCLOS), CBD and OSPAR, this lack of specific knowledge coupled with the more certain costs of inaction could be considered the main justification for large protected areas in the high seas. However, such characteristics make political action challenging, and bold action from politicians will be necessary. Even if the inferences are found to be incorrect at a later date, precautionary data-poor management will likely safeguard biodiversity better than the alternative of no management at all.

4.6.2. Complexities introduced by Outer Continental Shelf submissions

UNCLOS grants sovereign rights to all coastal States to explore and exploit the natural resources of their continental shelves which may extend to 200nm from the baseline. However, Article 76 of UNCLOS provided a mechanism for coastal States to seek to establish an extended outer limit to their continental shelf beyond

200nm[45]. The deadline for submissions to the UN Commission on the Limits of the Continental Shelf (UNCLCS) was set to be ten years after ascending to the Convention, which for the Member States of OSPAR fell in May 2009.

Areas beyond national jurisdiction (ABNJ) comprise the pelagic realm that falls beyond exclusive economic zones (200nm) and, as of the decisions taken by UNCLCS, the seafloor beyond the outer continental shelf boundaries (also commonly referred to as the extended continental shelf) (ISA 2010; Salpin and Germani 2010). Renewed legal uncertainty was introduced to the OSPAR MPA selection process in 2009 as a consequence of submissions of several OSPAR Contracting Parties to the UNCLCS to extend continental shelves within the OSPAR maritime area. These submissions created some confusion as to how MPAs with dual legislation (the water column under international legislation and national legislation being applied for the seafloor) could function.

In the North-East Atlantic all coastal states whose Exclusive Economic Zone borders the high seas have made individual, partial, joint and/or sometimes overlapping submissions to UNCLCS[46]. Following these submissions all but one of the OSPAR sites proposed for protection based on the available science were found to have a seabed falling partly or fully under national jurisdiction if submissions were successful (see Figure 21). The complexities arising as a result of these submissions include: complex jurisdictional issues until the UNCLCS has completed its work and issued recommendations, dual legislation and unsettled disputes as to the delimitation of the outer limits of extended continental shelves (e.g. the Rockall and Hatton Banks).

Although at first glance the dual legal regimes of the seabed and the water column appear to be legally difficult to co-manage, in practice success boils down to a willingness of States to engage cooperatively with the competent authorities. OSPAR and Portugal worked together and agreed to develop common management strategies for the Josephine Seamount, the Mid-Atlantic Ridge North of the Azores,

[45] Up to 350nm from its coastline or 100 miles from where the sea reaches a depth of 2,500m, and where specified geological conditions exist. See:
http://www.un.org/Depts/los/convention_agreements/texts/unclos/part6.htm

[46] For details of claims see: http://www.un.org/Depts/los/clcs_new/commission_submissions.htm

Altair Seamount and Antialtair Seamount MPAs[47]. All of these candidate MPAs fell within the ambit of the Portuguese submission to extend the limits of their continental shelves.

Nonetheless, outstanding outer continental shelf submissions in other parts of the OSPAR maritime area have presented difficulties for MPA designations, where Contracting Parties to OSPAR were not yet willing to enter into such co-management discussions. The options that OSPAR has for designating MPAs in ABNJ where extended continental shelf submissions exist are as follows: firstly, to postpone MPA designation until the UNCLCS has completed its work and the legal uncertainty has been removed, which present estimates suggest could take 23 years (Albuquerque 2010). This approach would fail to apply the precautionary principle as well as prevent global targets for marine protection to be reached in international waters. Secondly, to establish MPAs in areas not subject to outer continental shelf submissions, a pragmatic and realistic option although this would limit the ecological coherence of networks as well as leaving a large proportion of the high seas unprotected until submissions are resolved. Thirdly, to establish MPAs which only protect the water column overlying the seabed that is subject to outer continental shelf submissions, an option that is likely to raise legal issues and unlikely to find the support of the OSPAR Contracting Parties who have submitted seabed submissions under those waters (OSPAR 2010c). Lastly, as in the above example with regard to sites that fell within the Portuguese submissions, OSPAR could with the support of the relevant Contracting Parties, establish MPAs without prejudice to the outcome of the UNCLCS submissions, which would be subject to review, once the UNCLCS decision and outer continental shelf limits were established.

This final option was considered most preferable by OSPAR for the CGFZ MPA (which is subject to a seabed submission by Iceland); however, political consensus was not achieved in respect of the whole area proposed as a candidate MPA. In the end, the Charlie-Gibbs South MPA was designated for those areas lying outside the Icelandic submission, with the provision that the remaining area (Charlie-Gibbs North) would be reconsidered as a MPA subject to a more lengthy political process.

[47] OSPAR Recommendations 2010/14, 2010/15, 2010/16 and 2010/17.

Other OSPAR MPAs such as those protecting Josephine Seamount, the Mid-Atlantic Ridge North of the Azores, Altair Seamount and Antialtair Seamount were designated by both OSPAR (for the waters overlying the seabed) and Portugal (for the seabed)[48]. Without the support of Portugal for these MPAs the current OSPAR network of high seas MPAs is unlikely to have been established.

Until the UNCLCS has made its recommendations and States have established the limits of their continental shelf these issues will continue. Consequently, the establishment and management of MPAs in ABNJ will in such circumstances require cooperation and willingness between coastal States and competent organisations.

4.6.3. Stakeholder engagement

For a MPA to achieve its objectives, specifically that of conservation and protection, the coherent management of the seabed together with the water column is essential. Activities that occur in either realm will most likely impact the flora and fauna of the other (Salpin and Germani 2010). Consequently, it is essential that international organisations work together to develop management objectives and plans. Given that the management of human activities in the high seas of the North-East Atlantic fall under the competencies of a number of international organisations and conventions a broad understanding for MPA designation and management was needed. This also facilitates the move from a sectoral to an integrated ecosystem-based approach and extensive dialogue between OSPAR and other competent authorities[49] was necessary to accomplish this.

MPAs in ABNJ need to be established within the context of the legal status of the area, including specific interests, rights and competencies of organisations, communities and coastal states (Salpin and Germani 2010). Marine conservation efforts are often hindered by the difficulties in addressing multiple, and often conflicting uses (Harris *et al.* 2007). Within the North-East Atlantic competent authorities include the International Maritime Organisation (IMO), International

[48] OSPAR Decisions 2010/3; 2010/4; 2010/5;2010/6

[49] Other Competent Authorities for the high seas of the North-East Atlantic are: the North-east Atlantic Fisheries commission (NEAFC), the International Seabed Authority (ISA), the International Maritime Organisation (IMO), the International Commission for the Conservation of Atlantic Tunas (ICCAT), the North Atlantic Salmon Conservation Organisation (NASCO) and the North Atlantic Marine Mammal Commission (NAMMCO)

Seabed Authority (ISA) and the North-East Atlantic Fisheries Commission (NEAFC). So as to build a framework for the prolonged cooperation on the protection of MPAs in ABNJ and to initiate the first efforts towards multi-sectoral management, OSPAR sought to consult these key stakeholders during the nomination proforma stage[50]. Long-term cooperation between such stakeholders is essential for MPAs in ABNJ to achieve their objectives given the legal weakness of governance at a global level (there is currently no mechanism for MPA creation under UNCLOS). In addition, when data are poor, management can be informed through a stakeholder engagement process in order to increase acceptance and therefore compliance.

4.7. Lessons learned

The conservation and management of the high seas poses a new challenge to policy-makers and requires the application of international cooperation and political will. Despite the novelty of the process of high seas MPA designation OSPAR made swift progress because the Contracting Parties had already well-established cooperative relationships on issues of environmental protection. In particular, the target of establishing an OSPAR MPA network had been thoroughly endorsed and all but one of the coastal Contracting Parties[51] have now designated MPAs within national waters, albeit in varying number, coverage and distribution (OSPAR 2010a). Within the high seas balancing the trade-off between knowledge acquisition and conservation action is particularly acute. With limited data available for marine species and ecosystems within the high seas the precautionary principle has to be embraced in order to protect biodiversity. Throughout the process for the North-East Atlantic OSPAR embraced the precautionary principle designating areas, such as the Milne Seamount Complex MPA, on the basis of very little site-specific information, using instead inferential information from other similar places. Aiding this approach was the lack of significant economic activity in the areas proposed, and the parallel action undertaken by NEAFC as a result of the UNGA Resolution 61/105 and high

[50] See: *General outline of roadmap for further work on the Charlie-Gibbs Fracture Zone (CGFZ)/Mid Atlantic Ridge proposal 2008/09* (Summary Record OSPAR 2008; (OSPAR 08/24/1-E), Annex 10; www.ospar.org

[51] Several OSPAR Contracting Parties are not North-East Atlantic littoral states (Switzerland, Luxembourg, Finland).

level international statements made regarding the need to protect the high seas (e.g. through the CBD and FAO).

As it stands, this network of high seas MPAs falls short of OSPAR's target of ecological coherence. OSPAR has defined ecological coherence as including the necessity for the network to represent all types of habitat and species and for that network to exhibit connectivity (Ardron 2008; OSPAR 2010a). Assessing the ecological coherence of the MPA network is difficult due to the lack of detailed ecological data (OSPAR 2010a). However, the network is not yet spatially well-distributed across the wider Atlantic and key habitats, such as the abyssal plain, are not adequately represented. Indeed, only 3.15% of OSPAR’s maritime area currently falls within MPAs[52] falling short of the global commitment to protect at least 10% of each biogeographic region[53] (although in 2010 the CBD put back this target to 2020 (CBD COP10, Nagoya, Japan)). If the network is not well-distributed across space then it is likely that the network will not exhibit connectivity or representativity of ecoregions and habitats. Furthermore, the UN Resolution of 2006 (61/105) mandated the protection of *all* vulnerable deep sea ecosystems from fishing. This can reasonably be interpreted to mean that all seamount and other vulnerable habitats should be protected which is not the case at present. Justifiably, this network focused on only a subset of the habitats present in the region based on knowledge and the immediate need for protection, further sites will need to move beyond this. Recognising that the original target of achieving an ecologically coherent network of well-managed MPAs by 2010 has been missed, OSPAR has agreed a new deadline of 2012 for the establishment of a coherent network, with effective management of sites to be in place by 2016 at the latest[54]. Over the next two years the network (both within national waters and ABNJ) will therefore need to be extended.

Key lessons learned from this experience of high seas MPA designation include the following:

1. **Targets and deadlines** are essential to motivate action. The 2010 deadline set at an OSPAR ministerial conference helped spur OSPAR Contracting

[52] Report of the Ministerial Meeting 2010 (OSPAR/MMC 2010); www.ospar.org
[53] 2002 World Summit on Sustainable Development
[54] Report of the Ministerial Meeting 2010 (OSPAR/MMC 2010); www.ospar.org

Parties to seek the necessary knowledge and to gather the political drive to designate sites also in ABNJ.

2. **An adopted set of criteria** is needed to focus site nominations as well as a **clearly identified selection process** and agreed conservation objectives for the MPAs. Roadmaps can be a useful tool to help focus actions.

3. A "**Champion**" organisation and/or Contracting Party/Parties can help move the process forward by raising awareness of the gaps in political endeavours and putting forward the necessary options and tools to redeem these. In short, they build confidence that the goal is achievable. The non-governmental organisation (NGO) WWF played a critical role in bringing the CGFZ to the attention of OSPAR and consequently in initiating the process of considering MPAs in ABNJ. After support for the proposals was expressed from other Contacting Parties, Germany, as chair of the OSPAR MPA group, assumed this role driving the process politically.

4. **Independent evaluation**(s) of proposed MPAs in ABNJ are useful to remove questions of bias and to add further scientific credence. In this case, all of the proposals for MPAs in ABNJ underwent ICES reviews as well as reviews from an ad hoc OSPAR advisory panel, and a separate independent study. These evaluations provided further scientific consensus regarding the sites proposed.

5. **Transparency**: Presenting the evidence base clearly in the form of nomination proformas to justify MPA selection is essential. Whilst scientific data in ABNJ are still limited, a scarcity of data on specific features or habitats in particular places should not be construed as grounds to delay implementation of needed protection. Instead, the precautionary principle should be invoked, as noted in UN General Assembly Resolution

A/RES/61/105 of 2006[55], to protect vulnerable sites before the damage done to them increases to a point beyond the possibility of recovery. The acceptance of data limitations is an essential pre-condition to progress as the spatial scale of high seas MPAs in ABNJ is unlikely to be matched by adequate data coverage.

6. **Synergistic policy drivers**: The interplay of parallel processes and momentum can have significant positive effects. The establishment of the earlier North East Atlantic Fisheries Commission fisheries closures in the North-East Atlantic can be considered, at least in part, as a response to requirements arising from UNGA Resolution 61/105 on sustainable fisheries. These closures helped pave the way for the OSPAR MPAs, which in turn may have encouraged the additional NEAFC fisheries closures.

7. **Cooperation amongst competent authorities** is essential for MPA designation in ABNJ as well as for future management of the sites. The process of designating the OSPAR MPAs has led to greater cooperation and discussion between all key competent authorities. Particularly in the case of the Charlie-Gibbs South MPA, where a meeting between NEAFC, International Maritime Organisation, International Seabed Authority and OSPAR during spring 2010 was devoted to the discussion of co-management. Effective management of these areas will be key to achieving conservation objectives as set out by OSPAR and will require continued close cooperation amongst competent authorities and sustained political will.

8. **Allow sufficient time**: Like creating MPAs in national waters, designating MPAs in ABNJ is time consuming. It is a process of building momentum - identifying a lead country or countries, gathering the scientific rationale, convincing co-sponsors, raising awareness and achieving consensus.

---

[55] UN General Assembly Resolution A/RES/61/105 of 2006, paragraph 80. "*Calls upon* States to take action immediately, individually and through regional fisheries management organizations and arrangements, and consistent with the precautionary approach and ecosystem approaches, to sustainably manage fish stocks and protect vulnerable marine ecosystems, including seamounts, hydrothermal vents and cold water corals, from destructive fishing practices, recognizing the immense importance and value of deep sea ecosystems and the biodiversity they contain"

9. **Strong political commitment** and willingness are required to enable Contracting Parties to collaborate and cooperate on work to implement high seas MPA networks. The potential extension of the continental shelves of coastal states currently enhances legal uncertainty over the governance of some high seas MPAs. Without such commitment, legal conflicts such as unregulated boundary issues may be intractable and without such willingness legal complexities may be used as reasons to deter engagement.

10. **Regional Seas Conventions** such as OSPAR provide a valuable platform to facilitate cooperation and communication among Contracting Parties as well as with other competent authorities for the establishment of high seas MPA networks. In the current absence of a global implementing agreement for MPAs under UNCLOS, such bodies may represent a promising approach to achieving protection in ABNJ.

11. **Compliance**: Designating MPAs is just the start of a process. Whether these MPAs can deliver agreed conservation objectives and improve the target of sustainable management of the oceans depends on long-term collaborative arrangements between those institutions and organisations with legal competence over the areas. Multiparty monitoring, control and surveillance plans, combined with the political and social will for compliance, will therefore be essential.

4.8. Conclusions

The OSPAR high seas MPAs represent the first elements of an ecologically coherent network of MPAs in ABNJ setting a global precedent and as such are to be commended. However, this is only a start. It is likely that habitats, both within and beyond fishing depth are not represented and further representative protection of the North-East Atlantic is needed. As such, a gap analysis should be undertaken (and is now underway) such that this initial network can be complemented over time through the designation of additional MPA sites in order to become fully ecologically coherent and representative. For these sites to become part of a well-managed network of MPAs (in conjunction with coastal MPAs), many challenges lie ahead. Sustained political will, increased human and financial capacity, and improved governance and stakeholder engagement and compliance will need to be secured. Nonetheless, this is a significant global step forward in the process of multi-sectoral and sustainable management of the high seas.

4.9. Acknowledgements

The original scoping report was conducted in association with the Scottish Association of Marine Science and the PP Shirshov Institute of Oceanology in the Russian Federation. The authors would like to thank Andrew Davies, Tina Molodtsova and Melanie O'Rourke for their contributions of data and advice during the nomination proforma stage. We would also like to acknowledge the OSPAR Contracting Parties (and the members of the OSPAR MPA group) for their efforts throughout the designation process.

The manuscript benefitted from a presentation given to the International Seabed Authority in November 2010 and a report submitted to the United Nations Environment Program as a briefing paper in September 2010.

The map provided as Figure 21 in this article, has been prepared by Mirko Hauswirth of the German Federal Agency for Nature Conservation (BfN).

Chapter 5.

Is Sustainable Management of the Eastern and Western Populations of Atlantic Bluefin Tuna (*Thunnus thynnus*) Possible?

5.1. Abstract

The Atlantic bluefin tuna (*Thunnus thynnus*) has become the quintessential example of overfishing and general mismanagement of the world's fisheries, suffering from political quota adjustment, a lack of scientific integration into management and other problems such as illegal, unreported and unregulated (IUU) fishing. In this chapter I develop a spatial model of the eastern and western stocks of Atlantic bluefin tuna to explore its sustainable management. I investigate the importance of accounting for spatial information within management parameters and model the predicted recovery timeframe for the two stocks to reach their biomass corresponding to maximum sustainable yield (B_{MSY}) under different management scenarios within a deterministic regime. My results highlight the importance of taking area and stock movement into consideration when determining total allowable catches for the Atlantic bluefin tuna fisheries. In particular, my results suggest that the western bluefin stock is more sensitive to assumptions of stock movement and mixing than the eastern populations, corroborating previous research. My results also indicate that to maximise the total catches of bluefin in perpetuity, it may be better to cease fishing in the western Atlantic and to only target individuals in the eastern Atlantic. The estimated timeframes for recovery are found to be medium to long term if fishing were halted today (20 years for both stocks to attain their B_{MSY}) and it is estimated that a 34% reduction in fishing mortality on both stocks is the minimum required decrease to ensure recovery.

5.2. Introduction

The Atlantic bluefin tuna (bluefin) is one of the most highly prized and argued over species in the world. Large individuals may reach over 650kg (Block *et al.* 2005), can live up to 40 years (Boustany *et al.* 2008) and are heavily targeted for the Japanese raw fish market where they may fetch prices of over $100,000 each at auction (Stokstad 2010). The current record price was set in January 2012 at $736,000[56].

The species inhabits the pelagic environment of the North Atlantic in both the temperate and tropical waters. While long seasonal migrations to foraging grounds

[56] http://www.bbc.co.uk/news/world-asia-pacific-16421231

throughout this range have been observed, two distinct stocks are believed to exist; one in the western Atlantic and Gulf of Mexico, and one in the eastern Atlantic and Mediterranean. Larval surveys indicate two major breeding grounds, the Mediterranean Sea and the Gulf of Mexico and natal homing has been observed through tagging studies (Block *et al.* 2005; Boustany *et al.* 2008). Recent genetic evidence has also come to light supporting these concepts (Carlsson *et al.* 2007). To complicate matters for both stock assessments and fisheries management, the two populations mix when foraging. Mediterranean-spawning fish may therefore be caught while foraging in the western Atlantic (Block *et al.* 2001) and vice versa. Consequently, accurate stock assessments for the different populations are difficult and may be artificially inflated as a result of mixing (Reeb 2010).

The decline of both stocks of Atlantic bluefin tuna is well documented with the eastern Atlantic adult population falling by 90% (Walli *et al.* 2009) and the western population by 82% since 1970 (ICCAT 2010a), leading to both the eastern and western Atlantic populations being classified as 'critically endangered' (IUCN[57] criterion) (Druon 2010). Bluefin are vulnerable to concentrated fishing efforts due to their predictably high concentrations at feeding and spawning locations. For instance, Walli *et al.* (2009) observed that only 10% of all trans-Atlantic (i.e. those fish tracked to the east of the 45º management line) bluefin tuna that were tagged and returned to the Mediterranean Sea to spawn were not caught. Other natural characteristics such as late reproduction, large size at reproduction and a long life span decrease the resilience of this species to high rates of exploitation (Ottolenghi *et al.* 2004; Fromentin and Ravier 2005).

Whilst continuing exploitation increases the risk of collapse, the huge commercial importance of the Atlantic bluefin tuna means there is a lack of political will to take effective action to rebuild the populations (Fromentin and Ravier 2005). This is illustrated by the recent failure (18 March 2010) to implement an Appendix I CITES listing and a temporary international trade ban (Stokstad 2010). Although this failure has led to more stringent management measures being implemented, there remains a great deal of international concern over the fate of the bluefin and its fishery (e.g. Webster 2011). Management of the bluefin currently falls under the remit of the

[57] International Union for Conservation of Nature

International Commission for the Conservation of Atlantic Tunas (ICCAT) which was established in the late 1960s. Throughout its history ICCAT has encountered problems in implementing sustainable bluefin management. These include uncertainty over stock assessments, political pressure to maintain high catches as a result of the tuna's high value and illegal, unreported and unregulated (IUU) fishing (e.g. Fromentin and Ravier 2005; Webster 2011; Sumaila and Huang 2012). Political adjustment to the total allowable catch (TAC) setting process and IUU fishing have compounded the problems of overcapacity and overexploitation (De Stefano and Van der Heijden 2007; ICCAT 2010a; Sumaila and Huang 2012). For example, between 2003 and 2010 the eastern fishery was subjected to average political adjustment levels of 110% (calculated with the political adjustment index presented in Chapter 2 using data from Sumaila and Huang 2012). Generally, the fishing industry continues to catch well above these levels (ICCAT 2010a). Estimates of IUU fishing are uncertain but it has been suggested that catch levels may actually be double the TACs set by ICCAT (ICCAT 2010a). In addition, non-ICCAT members may also fish for bluefin, contributing to the IUU catch (Safina 1993; Sumaila and Huang 2012). Whilst recovery plans for both populations have been put in place (ICCAT 1998, 2006) it has been suggested that the stocks remain at risk from over-harvesting unless new conservation measures are implemented (e.g. MacKenzie *et al.* 2008; Armsworth *et al.* 2010; ICCAT 2010b).

Bluefin tuna has become the most demanded and expensive tuna species on the world market (Ottolenghi 2008). Atlantic bluefin tuna make up only a small proportion of the approximately 80 million tonnes of fish caught globally each year in marine fisheries (FAO 2010b). Nevertheless, they make up a disproportionate amount of the value in the sale of global marine resources (Greenpeace 2007) and once on the consumer market, the value of bluefin sales is likely to be in the billions of dollars (Issenberg 2007). The Atlantic bluefin tuna thus crystallises most of the problems of many fisheries (i.e. overcapacity; IUU fishing; high market value; open access in international waters; deficient governance) and has become the quintessential example of overfishing and general mismanagement of the world's fisheries. New management strategies are therefore needed in order to prevent the collapse of these stocks.

The increasing priority for more spatially explicit and integrated management between the two stocks of the Atlantic bluefin prompted ICCAT to develop alternatives to the two-stock, two-zone management regime (i.e. where the two stocks were considered separate with no mixing past the 45º boundary in the Atlantic). A two-stock, six-zone model was proposed in order to more accurately match the known life cycles of the two stocks and allow mixing rates between the two stocks to be considered within regulations (Apostolaki *et al.* 2003).

Several characteristics make this a useful operating model for population dynamics and the evaluation of spatial management systems for the Atlantic bluefin tuna. For example, the population dynamics of the two stocks are modelled in considerably more detail than standard stock assessment models; quarterly time steps allow for seasonal variations in the spatial distribution of fish and fishing effort. Also fish movements by age and stock can be simulated. More recently the Standing Committee on Research and Statistics (SCRS) of ICCAT presented a spatial, Multi-stock Age-Structured Tag-integrated stock assessment model (MAST) which models both stocks simultaneously in four areas[58] (Taylor *et al.* 2009). However, despite advances, ICCAT continues to use virtual population analysis (VPA) for stock assessments due to reliability issues (ICCAT 2008) and the difficulties in identifying plausible parameterisations to fit historic trends in bluefin abundance as well as time-specific area-movements of both fish and fishers (Apostolaki *et al.* 2004; McAllister *et al.* 2004). Whilst useful, VPA-based projections have a tendency to be positively biased to higher stock levels and to underestimate uncertainty (Patterson *et al.* 2000; McAllister and Babcock 2002).

The main findings of Apostolaki *et al.* (2003; 2004; 2005) were that assumptions regarding movement patterns can considerably affect model predictions for the status of both stocks. In particular, the western stock shows a higher sensitivity to changes in movement patterns and often predictions regarding current status and the prospects for rebuilding the western stock are dependent on the movement scenario used. Consequently they suggest that ignoring the complexity in bluefin movement patterns could lead to overestimation of the productivity or recovery potential of the western stock.

[58] SCRS/2008/097

In this chapter I develop a population dynamics model that tests the sustainability of the Atlantic bluefin tuna fishery under different biological and fishery scenarios, based on stock assessments and published biological data. The model simulates the dynamics of the two bluefin tuna stocks (eastern and western) in the North Atlantic, accounting for differences in the biology of the two populations and in their trans-Atlantic migrations.

5.3. Purpose and aims of the study

This research aims to develop a model incorporating the eastern and western Atlantic bluefin tuna stocks in order to provide insights into the dynamics of the two populations and their fisheries. The development of this model will allow the probable negative impact of political adjustment to the TACs of this fishery to be examined. In addition it will enable exploration of the potentially positive political effect of establishing marine protected areas, and investigation into the possibility that MPAs may buffer the effect of political adjustment, resulting in a sustainable fishery.

Development of this model will hopefully lead to further research into the sustainable management of the Atlantic bluefin tuna. Within this Chapter the model is used to evaluate the potential fishery yields that may be obtained by applying different management assumptions and two movement scenarios. Potential timeframes for recovery of the two stocks are also indicated for various catch scenarios.

Differing biological and exploitation scenarios may be considered, as the model accounts for the differences in the biology and behaviour of the two populations. However, within this chapter only one biological scenario is presented (described in the methodology section) in order to showcase some example results. *T. thynnus* has been selected as a case study for the following reasons; it represents a problem of immediate policy concern, it is subjected to political adjustment to TACs and MPAs have been suggested as one method that may aid the recovery of the species (Sumaila and Huang 2012). With increasing public awareness of the issues surrounding the bluefin tuna and with recent criticism of its management, ICCAT will have to act to restore public faith and to rebuild tuna populations. This study aims to provide a stimulus for further research into the Atlantic bluefin tuna and how the populations may be rebuilt to achieve sustainability.

The key research questions were:

1. What is the maximum sustainable yield (MSY) for both western and eastern populations, when the TAC for each is only removed from that respective population (i.e. taking the population as a whole in the absence of location considerations or mixing)?
2. How does fishing at the individual MSY level for each population affect the other population in the more realistic scenario where catches can be taken from either population depending on the location of the fish?
3. What is the MSY for each population when the TAC for each is only removed from the respective population and is taken based on historical catches for each area and proportionally from the population present in each area?
4. What are the optimum levels of exploitation when managing the stocks together, as opposed to management based on the calculation of separate MSYs?
5. How long would it take both populations to recover to their biomass corresponding to MSY and virgin biomass from their present day levels if fishing was (a) completely halted, (b) reduced by 10%, (c) reduced by 20% or (d) reduced by 50%?

These questions will be examined under a deterministic regime. In the longer term, resources subjected to stochasticity tend to require more conservative management and the steady state harvest becomes lower than the deterministic level (Poudel *et al.* 2011). Whilst the inclusion of stochastic processes within fisheries models has become more common, deterministic models remain a useful tool in analysing trends in population as a response to different events. Deterministic modelling outputs also provide valuable baselines which may be compared to stochastic results, particularly when uncertainty is present regarding the level of stochasticity a system experiences. For these analyses the deterministic baseline results are therefore calculated.

5.4. Methodology

An age-structured, five-area, monthly time step biomass model was developed to simulate the dynamics of the two bluefin tuna stocks in the North Atlantic, taking into account the mixing of the two stocks and their movement between areas. The use of an age-structured model allowed for differences in the biology, behaviour and exploitation of fish of different ages to be considered. A spatial model was developed in order to increase the realism in the simulation of fish movement and to examine the dynamics of the two stocks and the effects of exploitation on the stock sizes. In 2001 the SCRS stated that a spatial model should not exceed 5 or 6 areas and recommended that the key areas to distinguish were the Gulf of Mexico, Mediterranean Sea, Western, Central and Eastern Atlantic (SCRS 2002). At this meeting they suggested a model of 6 areas be considered. However, after researching the available data on movement patterns a 5 area model was designed. This was in order to simplify assumptions about movement patterns as there is limited tagging data for the eastern Atlantic population. As more accurate movement data becomes available the model could be extended to incorporate a greater number of areas.

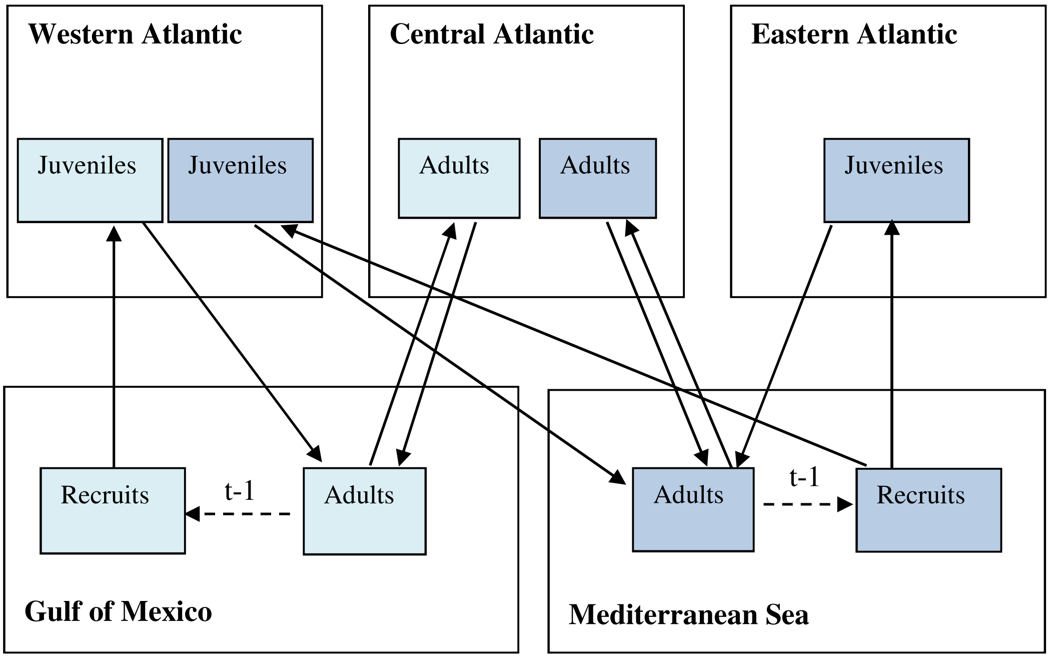
Input parameters were based on those used within the most recent Atlantic bluefin tuna stock assessment (ICCAT 2010b). Movement of the stocks was simplified from Block *et al.* (2005) and Rooker *et al.* (2007), the main findings of which are summarised in Table 7. The main life history traits of bluefin tuna are stated as follows: yearly spawning (1 cohort per year), life span of 20+ years, maturity occurs at 4 years for the eastern population and at 9 years for the western population (ICCAT 2010a). The natural mortality vector for each population was based on values adopted by the SCRS Bluefin Tuna Working Group (ICCAT 2010b). More detailed information is available for the natural mortality (μ) of the eastern population and is therefore age-specific but time- and area-invariant (i.e. μ=0.49 for age 1, μ=0.24 for ages 2 to 5, μ=0.2 for age 6, μ=0.175 for age 7, μ=0.15 for age 8, μ=0.125 for age 9 and μ=0.1 for ages 10+, in units of yr^{-1} in each case). For the western population mortality is assumed to be age-independent and is taken to be $0.14yr^{-1}$. The model framework and movement patterns of the two populations between areas are illustrated within Figure 22.

Table 7. Summary of Atlantic bluefin tuna movements as described by Block *et al*. (2005) and Rooker *et al*. (2007)

Block *et al*. 2005
• Identified two populations with distinct spawning areas; Gulf of Mexico and Mediterranean Sea. • Mixing zones were found to be primarily in the western and central Atlantic. • No mixing occurred in spawning areas. • Only adult tuna moved into known spawning grounds. *Western population* • Spawning is estimated to occur between April and June. • After leaving the western spawning area, tuna concentrate in the western and central Atlantic. • The central and eastern Atlantic are foraging areas for western spawners *Eastern population* • Tuna that migrated to the Mediterranean Sea resided in the western Atlantic foraging grounds for 0.5 to 3 years before leaving. • Spawning occurs between May and July, but may include August. • Once an eastern spawned bluefin tuna migrates to the Mediterranean Sea it is unlikely to return to the western Atlantic. • Summer movements of adults into and out of the Mediterranean Sea from the east Atlantic were observed, together with foraging in the central Atlantic.
Rooker *et al*. 2007
• Spawning occurs mainly in the Mediterranean Sea and Gulf of Mexico. • Adults may be found outside of spawning areas during spawning times. • Eastern Atlantic tagging efforts have been sporadic; total number of tags deployed is about one-third of that in the western Atlantic. • No mixing occurs in spawning areas. Substantial mixing along the North American coastline and central North Atlantic is noted. *Western population* • Spawning occurs in the Gulf of Mexico between April and June. • Over 99% of juveniles (< 4 years) tagged in the western Atlantic have been recaptured in the western Atlantic. Estimated west-east transatlantic migration rates are 22.8% and 12.6% for fish aged 4-8 and >8 respectively. *Eastern population* • Spawning occurs in the Mediterranean between May and July. • ~10% of juveniles tagged in the eastern Atlantic have been recaptured in the western Atlantic; significant exchange of juveniles between the Mediterranean Sea and the eastern Atlantic is indicated. • Data from tagging of fish in the eastern Atlantic is insufficient to characterise east to west movement of older individuals.

Only summaries of conclusions regarding movement patterns have been reproduced within this table.

Figure 22. Schematic diagram of the model framework



Solid lines represent movement patterns, dashed lines recruitment. Western populations occupy the Gulf of Mexico between April and June. 1 year old recruits enter the population and then move in July to the western Atlantic where they remain until they reach maturity and return as adults in April. Adults move into the Gulf of Mexico in April, spawn in May and move to the Central Atlantic to feed in July where they remain until they return to spawn the following year. Eastern populations occupy the Mediterranean Sea between May and July.1 year old recruits enter the population and then move in August to either the western Atlantic or the eastern Atlantic where they remain until they reach maturity and return as adults in May. Adults move into the Mediterranean in May, spawn in June and move to the Central Atlantic to feed in August where they remain until they return to spawn the following year. Natural mortality is independent of area and occurs at the beginning of every month.

The use of a detailed model requires information such as catch by area and time period and movement rates of fish by age, area and time. Not all of this information is available or it is characterised by considerable uncertainty. Therefore assumptions have to be made. Within this model these include:

1. The western and eastern stocks are considered to be separate populations that may mix while foraging but not contribute to each other's population in terms of recruitment.
2. The spawning grounds are solely utilised by the population which spawns in them (i.e. western population fish do not enter the Mediterranean Sea and vice versa.).

3. Juveniles do not remain or enter the two spawning areas.
4. Only two spawning areas exist within this model; sub-spawning areas such as those found within the Mediterranean Sea (Reeb 2010) are not considered. However, the model could be extended to include further spawning areas/sub-areas in the future.
5. Recruitment is based on the Beverton-Holt stock-recruitment curve and occurs at discrete time intervals (i.e. spawning times) and only within designated spawning areas.
6. Growth follows the von Bertalanffy growth relationship.
7. All adult fish spawn each year (no skipped spawning takes place).

Model structure

The spawning stock biomass of each population is calculated on an annual basis and a year consists of twelve time steps. Movement and natural mortality are instantaneous processes that take place at the beginning of every month where appropriate. Movement patterns follow those described in Figure 22. In its discrete form the population dynamics can be expressed as:

$$N_{i,a+1,r,t+1} = N_{i,a,r,t}\left(1 - \mu_{i,a,t}\right) + R_{i,t} \tag{5.1}$$

where the change in number of fish, $N_{i,a,r,t}$, of stock i and age a, in area r, at the beginning of time step t, after movement has occurred is known, is the result of survivorship from natural mortality ($\mu_{i,a,t}$) at age by stock per time step, and the recruitment to the population. All western individuals aged 9 or greater and eastern individuals aged 4 or greater are assumed to be mature. Parameters are defined in Table 8 and parameter values are given in Table 11.

The biomass of fish that spawn in each of the spawning areas at time, t, is calculated as the product of the total number of spawners (i.e mature fish) times the corresponding weight, $w_{i,a}$.

$$B_{i,a,t} = \sum_{a=a_{mat}}^{a_{max}}\left(N_{i,a,t} w_{i,a}\right) \tag{5.2}$$

The weight of fish at age a, $w_{i,a}$, of fish is calculated as a function of length (von Bertalanffy 1938):

$$l_{i,a,t} = l_{i,\infty}[1 - e^{-K_i(t-t_{0,i})}] \quad (5.3)$$

$$w_{i,a,t} = w_{i,\infty}[1 - e^{-K_i(t-t_{0,i})}]^{b_i} \quad (5.4)$$

where $l_{i,\infty}$, K_i, and $t_{0,I}$ are constants for each stock. These relationships allow fecundity to be weighted according to the size of the fish. By multiplying the population density in each age class by the specific weight of individuals in that class, the spawning stock biomass is weighted according to the distribution of fish across the population. For example, if the population is dominated by small young fish, the spawning stock biomass will be less than when there is a greater number of larger older fish.

Estimates of the stock recruitment relationship for the Atlantic bluefin tuna stocks remain highly uncertain within stock assessments (ICCAT 2008). However models are often based on the Beverton and Holt stock recruitment relationship because it represents a near constant level of recruitment at high spawner abundance and recruitment declines at low spawner abundance levels (Beverton and Holt 1957) (e.g. Apostolaki *et al.* 2003; ICCAT 2010b). For the Atlantic bluefin tuna the SCRS explore two models of spawner-recruit relationships: the two-line (low recruitment scenario) and the Beverton and Holt spawner-recruit formulation (high recruitment scenario) (ICCAT 2010b). The two-line model assumes recruitment increases linearly with SSB from zero with no spawners to a maximum value (R_{MAX}) when SSB reaches a certain threshold. This threshold is set at the average SSB during 1990-1995 as this is the period with the lowest estimated SSB. The SCRS calculates R_{MAX} as the geometric mean recruitment during 1976-2006 (the recruitment estimates for the previous three years are thought to be unreliable so were not used). The SCRS have fit the Beverton-Holt model to SSB and recruitment estimates for the period 1971-2006 (ICCAT 2010b). The model presented here is based on the Beverton-Holt relationship; however it would be possible to extend the model to test different spawner-recruit relationships such as those explored by Needle (2002).

The original Beverton-Holt stock-recruitment relationship $R_{i,t+1} = (\alpha_i B_{i,t,a})/(\beta_i + B_{i,t,a})$, was re-parameterised in terms of steepness (z) and virgin spawning biomass (females only) (B_0) (Mace and Doonan 1988). This was to allow changes in the fecundity characteristics of the stock to be explored without altering

B_0, enabling comparisons between the impact of management parameters and different stock recruitment relationships. The Beverton-Holt (1957) stock recruitment function related the biomass of spawners to the number of recruits at time, *t*:

$$R_{i,t+1} = \left(\frac{B_{i,t,a}}{\alpha+\beta B_{i,t,a}}\right) \tag{5.5}$$

where α and β are constants that can be calculated if the virgin spawning biomass, B_0, and recruitment, R_0, are known:

$$\alpha = B_0\left(\frac{1}{R_0} - \frac{5z-1}{4R_0 z}\right) \tag{5.6}$$

$$\beta = \frac{(5z-1)}{4(zR_0)} \tag{5.7}$$

where z is the steepness of the stock-recruit relationship and is equal to the fraction of the recruits under virgin conditions, R_0 (the recruitment corresponding to B_0), that are expected when the spawning biomass is reduced to 20% of B_0. In the absence of detailed information on steepness (z) the steepness of both stocks was set to 0.75. This value was based on the 0.7 value applied by Apostolaki *et al.* (2003) and the values of 0.75 and 0.9 used by Fromentin and Kell (2007). These values are considered to make biological sense for tuna (Kell and Fromentin 2009); the SCRS believe that 0.75 is the more appropriate estimate[59]. Sustainable yield estimates have been shown to depend on the steepness value chosen – the higher the steepness, the larger the yields and the better the recovery potential of the resource (Butterworth *et al.* 2003; Fromentin 2009). The value 0.75 was therefore chosen in order to avoid an overly high recovery potential for the two stocks and to conform to base scenarios run by the SCRS. The model was also run for steepness values of 0.7 and 0.9 to investigate the sensitivity of the stocks to assumptions regarding z, however the dynamics were found to be similar. With a higher z value (z = 0.9) the populations support a higher catch and recover faster once fishing pressures are removed than with a lower z value (z = 0.7). As it is the population dynamics of the stocks that is of interest within this study rather than the absolute values of catch, biomass or time,

[59] Extension of the 2009 SCRS meeting to consider the status of Atlantic bluefin tuna populations with respect to CITES biological listing criteria, Doc. No. PA2-604/3009.: http://iccat.int/Documents/Meetings/Docs/PA2-604%20ENG.pdf

the results are presented for the intermediate and more biologically appropriate value of $z = 0.75$.

The virgin biomass level (B_0) was set at 221,000 tonnes for the western population and 11 million tonnes for the eastern stock. These levels were based on recent SCRS estimates, however the results should not be considered to reflect the actual status of the stock as the virgin biomass conditions of the Atlantic bluefin tuna are controversial (SCRS 2010). However, the qualitative differences between the movement and management scenarios and between the eastern and western stocks are unlikely to be affected by this scale parameter.

R_0 is defined as $R_0 = \frac{\mu}{(1-\mu)^8} B_0$ for the western stock and $\sum\left(\mu_{a=4+,t_0} B_{a=4+,t_0}\right) + \sum\left(\mu_{a=1:3,t_0} B_{a=1:3,t_0}\right)$ for the eastern stock.

If R_0 is known then the number of fish at age, under virgin conditions, for the western stock is calculated as:

$$N_a^{vg} = \begin{cases} R_0 & a = a_1 \\ N_{a=2:a_{max-1}}^{vg} = N_{a=1}(1-\mu)^{1-a_{max-2}} & a = a_2 : a_{max-1} \\ N_{a_{max}}^{vg} = \left(\frac{(1-\mu)^{a_{max-1}} \cdot N_{a=1}}{\mu}\right) & a = a_{max} \end{cases} \tag{5.8}$$

The number of fish at age, under virgin conditions, for the eastern stock is calculated as:

$$N_a^{vg} = \begin{cases} R_0 & a = a_1 \\ N_{a=2:a_{max-1}}^{vg} = N_{a=1}\left(1-\mu_{a=1:a_{max-2}}\right)^{1-a_{max-2}} & a = a_2 : a_{max-1} \\ N_{a_{max}}^{vg} = \left(\frac{\left(1-\mu_{a_{mat-1}}\right)^{a_{max-1}} \cdot N_{a=1}}{\mu_{a=19}}\right) & a = a_{max} \end{cases} \tag{5.9}$$

The above equations assume that all fish of each stock occupy a single area. The spatial distribution of fish at the beginning of each simulation run was set according to the following rules:

- All western juveniles, aged between 1 and a_{mat-1}, were located in the western Atlantic;
- All western adults, aged between a_{mat} and a_{max}, were placed in the central Atlantic;

- Half of the eastern juveniles, aged between 1 and a_{mat-1}, were situated in the eastern Atlantic and half in the western Atlantic; and
- All eastern adults, aged between a_{mat} and a_{max}, were located in the central Atlantic.

The population in the first year of a simulation run is assumed to be a fraction, $\emptyset_{\mathrm{i,a}}$ of the virgin population:

$$N_{i,a,r,t} = \emptyset_{i,a} \cdot N_{i,a,r,t}^{vg} \tag{5.10}$$

If the time step (t) corresponds to the spawning period (April - June for the western stock and May-July for the eastern stock), it is assumed that the sexually mature fish move into the area in the first month, spawn in the second and exit in the third. Immature fish remain in the area that they occupied at the beginning of the spawning period. Adults from the western stock move to the Gulf of Mexico to spawn while those from the eastern stock move to the Mediterranean Sea. Movement between areas is an instantaneous process that takes place at the end of the appropriate time step (see Figure 22).

Table 8. List of parameters

Symbol	Parameter description
a	Age
i	Stock
t	Time
B_0	Spawning stock biomass under virgin conditions
μ	Natural mortality
z	Steepness of stock-recruitment curve, no units - dimensionless
R	Recruitment
R_0	Recruitment under virgin conditions
α, β	Constants of the Beverton-Holt stock recruitment function
l	Fish length
l_∞	von Bertalanffy parameter. Defines the asymptotic or maximum body length size.
K	von Bertalanffy parameter known as the Brady growth coefficient. Defines the growth rate toward the maximum.
t_0	von Bertalanffy parameter. Shifts the growth curve along the age axis to allow for apparent nonzero body length at age zero.
w	Fish weight.
w_∞	Defines the asymptotic or maximum body weight size.
b	von Bertalanffy parameter.

Fishing was assumed to take place annually and be an instantaneous process occurring in the middle of each year. Catches were based on averages of longline and purse seine catch data from 1998-2008 for the Atlantic bluefin tuna (the most recent database available from ICCAT[60]). Data were assigned to the model areas and averaged over the ten year period to gain a perspective into the true location of catches. Total catches were set to equal the TAC and the proportion of catches recorded in each area annually was calculated. However, disadvantages exist with using these data:

1. Catches (total and location) may be misreported by fishers.
2. Fish caught on the eastern side of the 45º line separating the management of the two stocks are recorded as fish from the eastern population and TAC, and vice versa.
3. This database does not include estimates of illegal fishing.

The MSY was calculated separately for each stock assuming the western TAC was taken only from the western population and vice versa, and that individual areas would not affect the MSY. The target for managing both fish stocks was set to be 50% of their respective virgin population sizes (Caddy and Mahon 1995) and catches were assumed to be taken proportionally from each age class. All age classes (1-20+) were assumed to be exploitable by fishers. MSY was calculated iteratively using a bisection method[61] (Press 1992).

Annual catches for each stock were then set to the corresponding MSY and the proportion of the MSY (or TAC) for each area was taken in the middle of the year after all spawning had taken place. Where no mixing took place catches were taken proportionally according to age class from either the western or eastern population present in each area (Table 9). In each area where the two stocks mixed (i.e. the western and central Atlantic (see Figure 22) catches were taken from both populations according to the proportion of each stock present at that time (Table 9). As no fish were present in the two spawning areas (the Gulf of Mexico and Mediterranean) at this time (all mature fish are assumed to spawn), these catches were taken from the respective adult populations present in the central Atlantic, i.e.

[60] This database is freely available from http://iccat.int/en/accesingdb.htm
[61] The bisection method is a root-finding which repeatedly bisects an interval and then selects a subinterval in which a root must lie for further processing (Press 1992).

the Gulf of Mexico catches were taken from the western stock and the Mediterranean catches were taken from the eastern stock. Table 10 lists the proportions of each TAC taken from each area annually. Table 9 describes the methodologies for taking catches based on area and mixing.

The optimum levels of catch required to maximise the catch for the western, eastern or total fishery yield were also calculated iteratively using a bisection method. The initial catch levels were set at zero for the minimum possible catch (C_{MIN}) and at the total exploitable population value for the maximum possible catch (C_{MAX}). At the end of each simulation run, the catch levels for the western and eastern Atlantic were recorded if both populations remained above $0.5B_0$. If either population fell below $0.5B_0$, no sustainable catch level was recorded. From the resulting matrix of sustainable combinations of western and eastern catch levels, the combination of western and eastern catches which provided the maximum western, eastern or total yield was found. Using these values the catch parameter could then be adjusted to reduce the interval between C_{MIN} and C_{MAX} until optimal combinations were found.

Note that despite the different growth curves for each stock, catches for each age class were assumed to be taken from the same age class of the other stock; i.e. if the western TAC for age class 1 was 2 units and 1 of these units was taken from the eastern population this unit would also be taken from age class 1 for the eastern population.

Table 9. Methodology for removing catches based on area with no stock mixing, and area with stock mixing

Methodology for removing catches from the two stocks separately (i.e. western TAC from western stock and vice versa) taking area into account:

$$TAC_A = Proportion\ of\ total\ TAC\ taken\ from\ area\ A\ (see\ Table\ 8)$$

$$Total\ population\ in\ area\ A = \sum A_{1:20}$$

$$Proportion\ of\ fish\ in\ each\ age\ class\ (a) = \frac{a_1}{Total\ population\ in\ A} \cdots \frac{a_2}{Total\ population\ in\ A} \cdots \frac{a_{20}}{Total\ population\ in\ A}$$

$$Catch\ in\ area\ A\ (proportionally\ by\ age\ class\ (a)) = TAC_A * Proportion\ of\ fish\ in\ each\ age\ class$$

$$Remaining\ population\ in\ area\ A\ after\ catches = A_{1:20} - Catch\ in\ area\ A$$

Methodology for removing catches from the two stocks taking area and mixing into account.

$$TAC_A = Proportion\ of\ total\ TAC\ taken\ from\ area\ A\ (see\ Table\ 8)$$

$$Total\ population\ in\ area\ A = \sum A_{s1,1:20} + \sum A_{s2,1:20}$$

where s1 and s2 represent the western and eastern stocks.

$$Proportion\ of\ western\ fish\ in\ each\ age\ class\ (a) = \frac{s1_{a=1}}{Total\ population\ in\ A} \cdots \frac{s1_{a=2}}{Total\ population\ in\ A} \cdots \frac{s1_{a=20}}{Total\ population\ in\ A}$$

$$Proportion\ of\ eastern\ fish\ in\ each\ age\ class\ (a) = \frac{s2_{a=1}}{Total\ population\ in\ A} \cdots \frac{s2_{a=2}}{Total\ population\ in\ A} \cdots \frac{s2_{a=20}}{Total\ population\ in\ A}$$

$$Catch\ of\ western\ fish\ in\ area\ A\ \big(proportionally\ by\ age\ class\ (a)\big) = TAC_A * Proportion\ of\ western\ fish\ in\ each\ age\ class$$

$$Catch\ of\ eastern\ fish\ in\ area\ A\ \big(proportionally\ by\ age\ class\ (a)\big) = TAC_A * Proportion\ of\ eastern\ fish\ in\ each\ age\ class$$

$$Remaining\ western\ population\ in\ area\ A\ after\ catches = A_{1:20} - Catch\ of\ western\ fish\ in\ area\ A$$

$$Remaining\ eastern\ population\ in\ area\ A\ after\ catches = A_{1:20} - Catch\ of\ eastern\ fish\ in\ area\ A$$

Table 10. Proportion of total allowable catch (TAC) taken annually from each area and stock

Area	Proportion of TAC
Eastern Atlantic	0.14*eastern TAC
Western Atlantic	0.55*western TAC
Central Atlantic	0.03*eastern TAC + 0.35*western TAC
Mediterranean*	0.83*eastern TAC
Gulf of Mexico*	0.1*western TAC

* These catches were taken from their respective populations in the central Atlantic.

Input parameters and assumptions

The values of the input parameters used for the calculations are shown in Table 11. These values are intended to accurately reflect current knowledge about the Atlantic bluefin tuna biology and are the values used by ICCAT in its most recent stock assessment (ICCAT 2010b). However, there is considerable uncertainty within these values. For example, it has been suggested that the age of maturity for the western population may be as late as 11 (Rooker *et al.* 2007) or 12 (Safina and Klinger 2008) and that the virgin biomass may fall between 80,000-221,000 tonnes for the western stock and 825,000-2.81 million tonnes for the eastern stock (although for a steepness level of 0.75 estimates usually fall between 1-11.7 million tonnes for the eastern stock) (SCRS 2010).

Fishery catches were based on average historical catches east and west of 45º from 1998-2008. Catches were available in quarterly format and were totalled to produce annual catches.

Table 11. Input parameter values and assumptions used in the model

Parameter	Western Stock Value	Eastern Stock Value
Age at maturity	9	4
Natural mortality	0.14 yr^{-1}	0.49 age 1, 0.24 age 2-5, 0.2 age 6, 0.175 age 7, 0.15 age 8, 0.125 age 9, 0.1 age 10+
Virgin spawning stock biomass (B_0)	221,000t	11,000,000t
B_{MSY}	0.5 B_0	0.5 B_0
Spawning season	April-June	May-July
Spawning area	Gulf of Mexico	Mediterranean Sea
Length (L)/age relationship	$l_\infty = 315$ K = 0.089 $t_0 = -0.093$	$l_\infty = 319$ K = 0.093 $t_0 = -0.093$
Weight (W) (Kg) /length (cm) relationship	$W = 2.861 \cdot 10^{-5} * L^{2.929}$	< 100 cm: $W = 2.95 \cdot 10^{-5} * L^{2.899}$ ≥ 100 cm: $W = 1.96 \cdot 10^{-5} * L^{3.009}$
Longevity	20+	20+
Stock-recruitment relationship	Beverton-Holt	
Steepness (z)	0.75	0.75

See ICCAT (2010b) for detailed references for each parameter.

Scenarios considered

1. Scenario 1

   No juvenile mixing of the two populations occurs in any area at any point of the simulation.

2. Scenario 2

   10% of eastern Atlantic juveniles move to the western Atlantic where they mix with the juveniles of the western stock.

Movement of stocks was simplified from data published by Block *et al.* (2005) and Rooker (2007). Within both scenarios all adults of both western and eastern stocks reside in the central Atlantic apart from during the spawning season where they move to their respective spawning grounds. Catches are always taken proportionally to the population present in each age class in each area and are based on the average percentage calculated from the last 10 years of available catch data (1998-2008) of the TAC (or MSY for these scenarios) that are taken from each area. Both scenarios use the same biological characteristics (i.e. B_0, z, μ, age of maturity, target biomass) as described above. The aim of presenting the results for these two scenarios is to begin to examine the effect of assumptions regarding movement and mixing of the two stocks on sustainable management and to showcase some initial results for this model.

All simulations were run in Matlab (Mathworks 2008).

5.5. Results

<u>*Question 1:*</u> *What is the maximum sustainable yield (MSY) for both western and eastern populations, when the TAC for each is only removed from that respective population (i.e. taking the population as a whole in the absence of location considerations or mixing)?*

The MSY for each stock was determined assuming the western TAC was taken only from the western population and vice versa. The proportion of the total population in each age class for each stock was calculated and the TAC was taken proportionally based on age class. By taking the population of each stock as a whole, no consideration of the spatial distribution of the fish was made. Based on the biological characteristics described above (i.e. B_0, z, μ, age of maturity, target biomass) MSY was calculated iteratively using a bisection method to be approximately 6,000t for the western stock and 216,000 tonnes for the eastern stock giving a total theoretical catch of 222,000 tonnes. This allowed the western population to be maintained at the target biomass (or biomass corresponding to MSY) of 111,000 tonnes and the eastern at 6 million tonnes. This remains the same for both movement scenarios because catches are only taken from the whole population irrespective of location. It is worth noting however that these values are theoretical based on the estimated virgin biomass of the two populations and that currently present values of biomass in the wild are far lower than these (ICCAT 2010b).

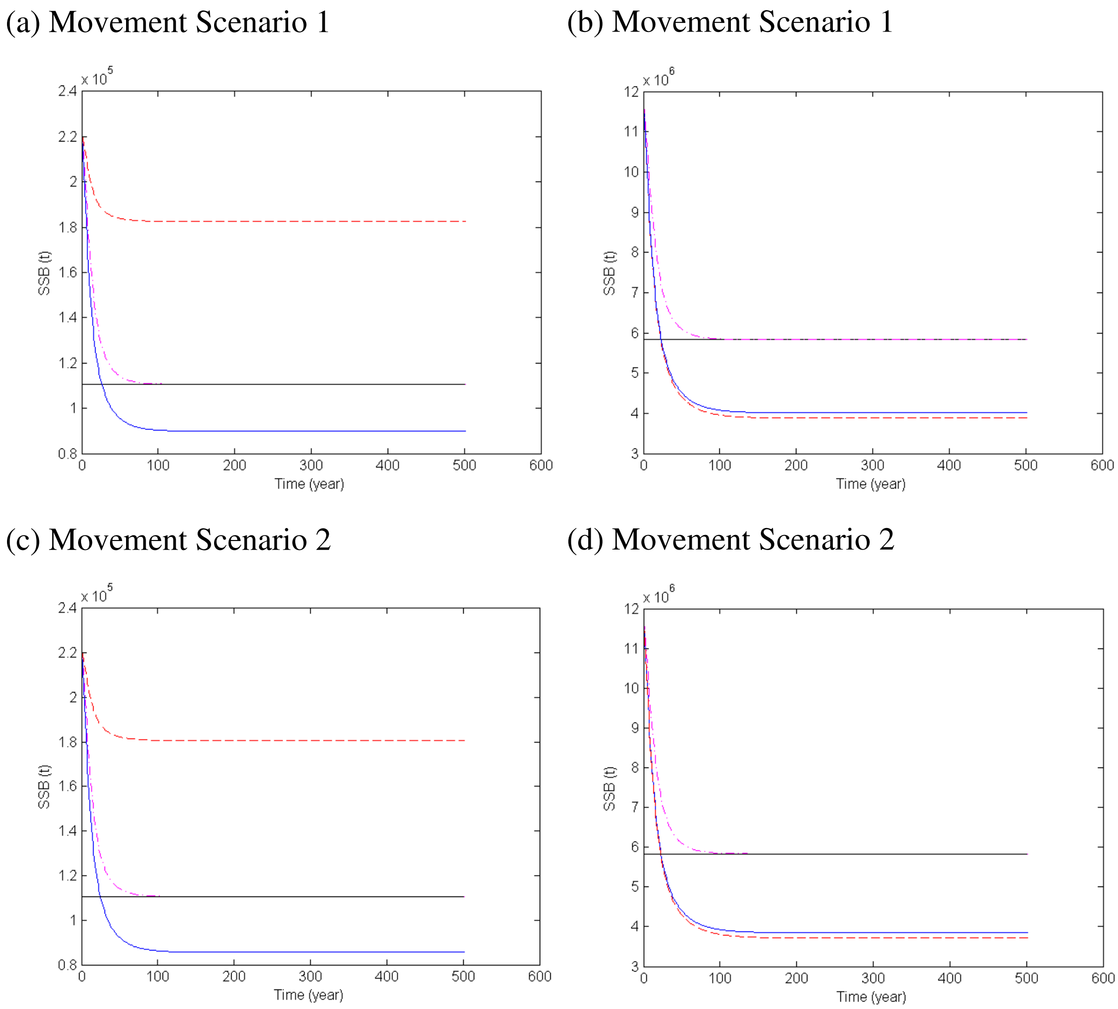
Question 2: How does fishing at the individual MSY level for each population affect the other if catches can be taken from either population depending on the location of the fish?

Figure 23 shows the impact on spawning stock biomass when the MSY catch was taken from the two populations where (a) area was accounted for but the two MSY catch levels were still only taken from their respective populations, and (b) where the catch was taken proportionally between area and the stock present in each area (see Table 9 for methodology) according to the quotas established in Table 10 for both mixing scenarios.

The results for both movement scenarios are very similar. This indicates that the movement of 10% of eastern Atlantic juveniles to the western Atlantic area is not enough to have a significant impact on the status of either population. Instead the results indicate that it is the movement patterns and population localities as a whole that are important within management (compare the blue solid and the red dashed lines in Figure 23) rather than a small percentage of juveniles moving across, for these scenarios. Figure 23 shows that when either area or mixing between the two stocks (with the subsequent capture of western individuals being counted towards the eastern TAC and vice versa) is not taken into account into the calculation, neither population of Atlantic bluefin tuna is maintained at B_{MSY}, for either movement scenario.

When area is taken into account (Table 9) during the extraction of catches, but the western TAC is still only taken from the western stock and vice versa, both populations fall below the target biomass of $0.5B_0$ (Figure 23 solid blue line). When the catches are taken proportionally from each age class according to both populations present in each area (Figure 23 red dashed line) the western stock is shown to be buffered by the eastern population allowing it to maintain a higher SSB (Figure 23a and 3c) whereas the eastern population is only marginally affected by the western stock (Figure 23d).

Figure 23. Impact of taking the MSY catch, when calculated without taking area or mixing into account, for the (a and c) western and (b and d) eastern stocks with and without mixing for the two movement scenarios



The black solid line represents the B_{MSY} for each stock. The magenta dash-dot line indicates the SSB corresponding to the MSY when catches are taken from the population as a whole not taking into account area or the mixing of the two stocks. The blue solid line indicates the SSB when the MSY catch is taken proportionally from each age class of the appropriate stock in each area. The red dashed line indicates the SSB when the MSY catch is taken proportionally from each age class according to both populations present in each area.

<u>Question 3:</u> What is the MSY for each population when the TAC for each is only removed from the respective population and is taken based on historical catches for each area and proportionally from the population present in each area?

In order to validate these results, the MSY for each stock was determined assuming the western TAC was taken only from the western population and vice versa and that catches were taken proportionally from the population present in each area (Table 9).

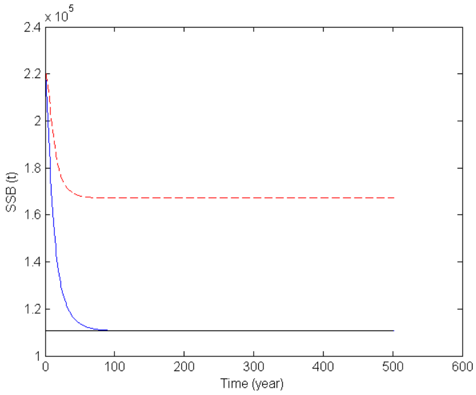
Based on the same biological characteristics as used in the previous calculation, MSY was calculated to be 5,000t for the western stock and 165,000t for the eastern stock under both movement scenarios. This gives a total theoretical sustainable yield of 170,000t - a decrease of 52,000t from the total MSY calculated when area and mixing were not taken into account. This decrease stems from the additional mortality of western juveniles and eastern adults as a consequence of the distribution of catches (Table 10).

Figure 24 shows the impact of setting the TAC to the MSY catches for the western and eastern stocks for both movement scenarios (a) ensuring catches only come from their respective populations present in each area and (b) taking the catches proportionally from both populations present in each area.

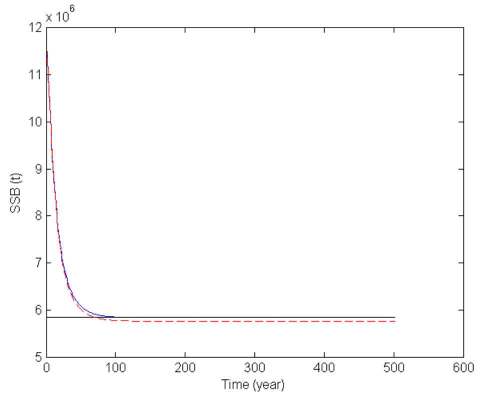
The trends for both movement scenarios were found to be the same – the western SSB is maintained at a much higher level than its B_{MSY} while the eastern SSB falls below target. This supports the results drawn from Figure 23, that the western population is buffered substantially by the eastern population allowing it to maintain a higher SSB than its respective TAC would allow. These results also indicate that the western stock has a higher sensitivity to assumptions about movement than the eastern stock.

Figure 24. Impact of taking the MSY catch, when calculated with no mixing of the two stocks assumed to take place but area being taken into account, on spawning stock biomass of the (a and c) western and (b and d) eastern stocks when the model is run taking into account both area and stock mixing

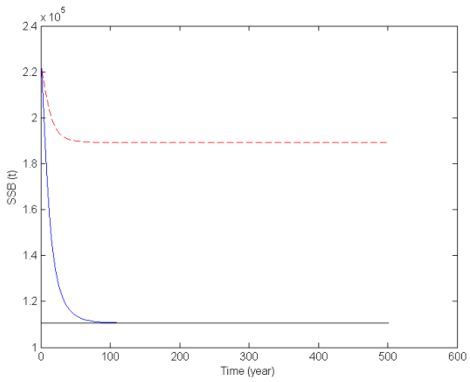
(a) Movement Scenario 1



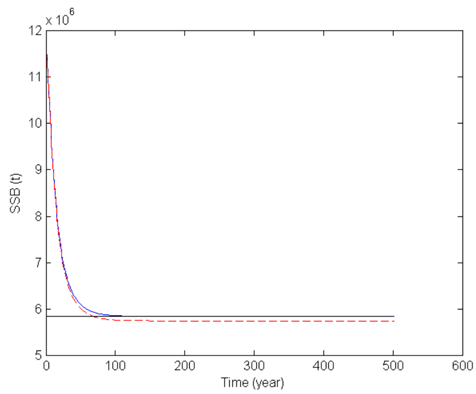
(b) Movement Scenario 1



(c) Movement Scenario 2



(d) Movement Scenario 2



The black solid line represents the B_{MSY} for each stock. The blue solid line shows SSB maintained when the model is run with no mixing being taken into account by the MSY calculation. The red dashed line shows the SSB that is maintained when the model is run using the same MSY levels for each stock but taking mixing of the two populations into account.

<u>Question 4:</u> What are the optimum levels of exploitation when managing the stocks together as opposed to their separate MSYs?

The above results show that neither stock is maintained at B_{MSY} when extracting the catches calculated when area but not mixing is taken into account. However, the total yield of Atlantic bluefin tuna is not maximised. Table 12 presents a summary of the MSY totals calculated for each stock according to two mixing scenarios (1 - no mixing of juveniles, and 2 - 10% of eastern Atlantic juveniles move to the western Atlantic). The model was run with the aim of maximising (a) the total yield taken from the western Atlantic[62], (b) the total yield taken from the eastern Atlantic and (c) the total combined yield across the Atlantic when both stocks may be caught (see p126 for methodology).

Table 12. Summary of the MSY calculated for each stock according to the mixing scenarios of (1) no mixing and (2) 10% of eastern Atlantic juveniles move to the western Atlantic

MSY Scenario	Movement Scenario	Western Atlantic MSY (t)	Eastern Atlantic MSY (t)	Total Theoretical Yield (t)
(a) Calculated annually taking area and stock mixing into account, but trying to maximise the western Atlantic catch	1	21,910	0	21,910
	2	41,470	0	41,470
(b) Calculated annually taking area and stock mixing into account, but trying to maximise the eastern Atlantic catch	1	0	1,091,817	1,091,817
	2	0	1,091,824	1,091,824
(c) Calculated annually taking area and stock mixing into account, but trying to maximise the catch of both fisheries	1	10,730	161,269	171,999
	2	15,174	157,325	172,499

Overall, the increase in MSY for the western stock from original calculations (see results for Questions 1 and 3) under MSY scenarios a and c is unsurprising. Previous simulations have shown that the western stock is buffered by the eastern stock (see Figure 23 and Figure 24). The increase in catch is therefore likely to be due to the presence of the eastern population in the western and central Atlantic, from which 90% of total western fisheries catches are taken.

The sensitivity of the eastern Atlantic stock to the two movement scenarios presented here appears to be much lower than that of the western. Very little variation is seen in the MSYs calculated in scenarios b and c. For scenario b, the small increase in yield found for movement scenario 2 is probably due to the additional biomass of juveniles (1-3 year olds) surviving to adulthood as a result of their movement out of the eastern Atlantic fishery to the western Atlantic. For scenario c, the slight

[62] Due to mixing of the two stocks being taken into account MSY is now being calculated for the western and eastern Atlantic catches rather than the two stocks.

decrease in yield is the result of fewer available juveniles in the eastern Atlantic as well as the increased western catch made possible by the buffering effect of the eastern population on the western stock. The reduction in catches for the eastern Atlantic would therefore be the result of additional eastern population catches being taken from the western and central Atlantic areas but being counted towards the western Atlantic catch. However, in a stochastic world these small variations are unlikely to matter due to the natural variations in recruitment and survivorship.

<u>Question 5:</u> *How long would it take both populations to recover to their biomass corresponding to MSY and virgin biomass from their present day levels if fishing was (a) completely halted, (b) reduced by 10%, (c) reduced by 20% or (d) reduced by 50%?*

While these results should not be taken as absolute values for the fishery they do indicate that catches of *T.thynnus* could be much greater than present day yields[63] if the two stocks were allowed to recover to their B_{MSY}. It is therefore interesting to examine the question of how long it would take both populations to recover from their present day levels if all fishing were completely halted or reduced.

Estimates indicate that the western and eastern populations have fallen by 82% and 90% respectively since 1970 (Walli et al. 2009; ICCAT 2010a). As these stocks were being exploited prior to 1970 (Fonteneau 2009) it is likely that both populations have experienced an even greater total decline. However, depleting the stocks by these amounts and projecting recovery timeframes from present day levels should provide an indication of the expected timeframes for recovery.

Initially, the fishing effort appropriate to produce present day biomass levels needed to be identified. During simulation runs threshold SSBs were found. In movement scenario 1, when the western stock attained approximately 83,000 tonnes and the eastern 2 million tonnes both stocks became unable to support further catches sustainably. In movement scenario 2, these threshold levels were lower for the western stock at 63,000 tonnes and higher for the eastern stock at 2.4 million tonnes. The difference in these levels highlights the effect of mixing. The changes seen in the threshold biomasses between movement scenarios stem from the shifting

[63] TACs for 2011 were set at 12,900t for the eastern Atlantic stock and 1,750t for the western Atlantic stock.

interdependencies of the stocks depending upon the movement assumptions made. When no eastern Atlantic juveniles move to the western Atlantic (movement scenario 1) the western stock is no longer buffered by the eastern population and the biomass available for capture in the western Atlantic is reduced. The threshold biomass is therefore higher under movement scenario 1 as western catches are lower which maintains a greater biomass. Under movement scenario 1 eastern juveniles are not caught and counted towards the western TAC. Consequently there is a greater availability of juvenile biomass in the eastern Atlantic and the stock can support higher catches which drives the SSB lower. If fishing levels were set to those that reduce the stocks to their present day levels under either movement scenario a sustainable biomass was therefore not achieved and both stocks continued to decline.

Figure 25 shows the predicted future biomasses of each population for both movement scenarios according to a number of catch scenarios, (1) a status quo scenario, (2) if fishing were reduced overall by (a) 10% (b) 20% and (c) 50% from present day levels and (3) if fishing were completely halted.

Movement scenario 1 results:

In the status quo scenario the SSB of both stocks were found to reduce to zero within 40 years. If all fishing were halted today, the model predicts that the western stock would recover to its B_{MSY} within 8 years and to its virgin biomass level (SSB_0) in 65 years. The eastern stock is predicted to recover to its B_{MSY} in 19 years and SSB_0 in 88 years with the complete cessation of fishing. However, it is unlikely to become politically or economically acceptable for fishing of Atlantic bluefin tuna to be completely halted. Consequently it is worth examining the effect of reducing present day fishing levels. If the entire fishery (east and west) underwent a 10% reduction in total catches both stocks would still fail to recover to a more sustainable level and instead this only doubles the time it takes for both stocks to reach zero. If the whole Atlantic bluefin tuna fishery underwent a 20% reduction in catches the western fishery would recover to B_{MSY} in 67 years however the eastern stock would fail to reach B_{MSY}. A 50% reduction would ensure both stocks recovered to B_{MSY} within 50 years at which point the total catches would be able to increase to MSY.

While the expected timeframe for recovery is relatively short (from a conservation point of view) it is unlikely to be socially or politically acceptable to put half the fishery out of business. In order to eventually reach B_{MSY} and therefore be able to maximise catches, the model predicts that the western stock would require a 15% reduction in total catch to be able to rebuild to its B_{MSY} in 180 years. However, the eastern stock would remain below its B_{MSY} at this level. In order for both stocks to rebuild to their respective B_{MSY} a reduction of 34% of the total catch would be necessary. At this level, the eastern stock would rebuild within 124 years and the western within 25 years.

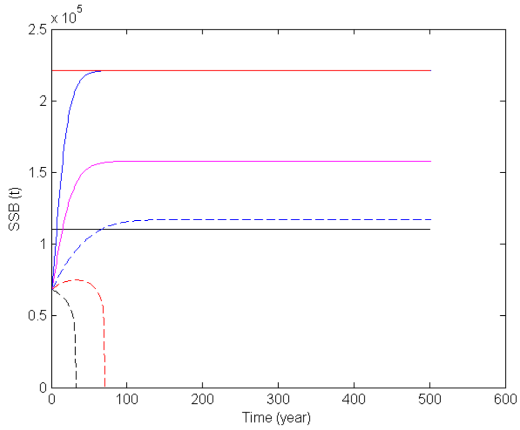
Movement scenario 2 results:

In the status quo scenario the SSB of both stocks were found to reduce to zero within 50 years. If all fishing were halted today, the model predicts that the western and eastern stocks would recover to their B_{MSY} within 10 and 18 years respectively and to their virgin biomass levels in 73 and 77 years. If the entire fishery (east and west) underwent a 10% or 20% reduction in total catches both stocks would recover to a more sustainable level within 200 years. However, the catches would have to remain at this level in order to maintain this biomass as any increase would lead to both stocks declining. In order to be able to maximise catches in the future both stocks would need to rebuild to B_{MSY}. A 50% reduction in the fishery now would allow this target to be reached within 50 years.

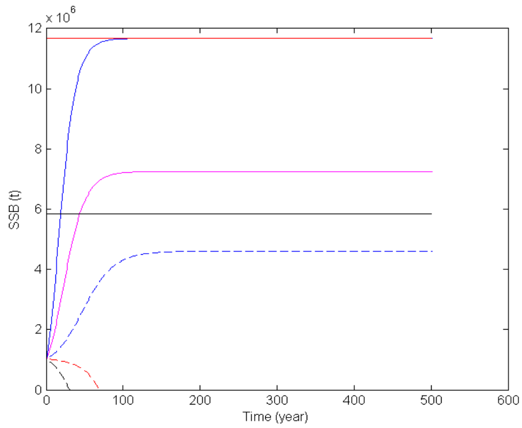
In order to minimise the total loss to the fishery and allow both stocks to eventually reach B_{MSY} the model predicts that the western stock would require a 23.5% reduction in total catch to be able to rebuild to its B_{MSY} in 180 years. However, the eastern stock would remain below its B_{MSY} at this level. In order for both stocks to rebuild to their respective B_{MSY} a reduction of 33.5% of the total catch would be necessary. At this level, the eastern stock would rebuild within 180 years and the western within 40 years.

Figure 25. Predicted SSB of (a and c) the western and (b and d) the eastern stock from present day levels according to 3 catch scenarios, (1) status quo (black dashed line), (2) 10% (red dashed line), 20% (blue dashed line) and 50% (magenta solid line) reduction in catches from present day levels and (3) if fishing were completely halted (blue solid line)

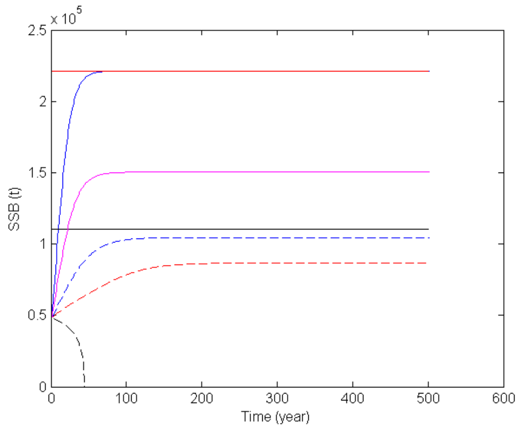
(a) Movement Scenario 1



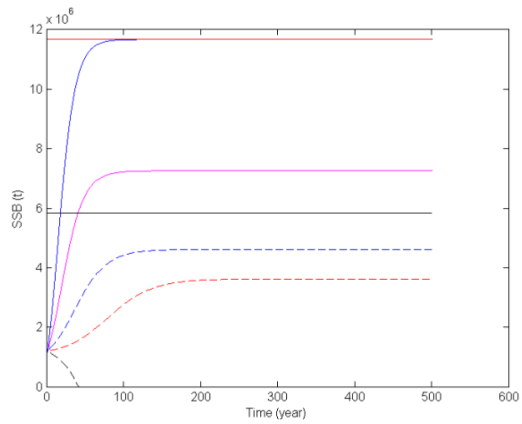
(b) Movement Scenario 1



(c) Movement Scenario 2



(d) Movement Scenario 2



The black solid line indicates the B_{MSY} for each stock. The red solid line indicates the virgin biomass level. N.B. The starting biomass for the western population movement scenario 1 had to be set at $0.3SSB_0$ rather than $0.1SSB_0$ as no catch levels could be found that reduced both populations to their target levels. If the western catch level was taken to reduce the stock to $0.18SSB_0$ the eastern population would collapse. Consequently the western stock starting biomass was set higher and the eastern fishery preferentially set to present day levels as this fishery is considered more important economically. The results presented in (a) therefore are likely to be biased towards a quicker recovery than if the starting biomass was $0.18SSB_0$.

5.6. Discussion

This chapter develops a model incorporating the eastern and western Atlantic bluefin tuna stocks for use in assessing the inter-dynamics of the two populations of the Atlantic bluefin tuna and their fisheries. In this section the results to the 5 research questions posed are summarised and further work is discussed.

Question 1: *What is the maximum sustainable yield (MSY) for both western and eastern populations, when the TAC for each is only removed from that respective population (i.e. taking the population as a whole in the absence of location considerations or mixing)?*

Under these assumptions the western Atlantic stocks MSY is 6,000t and the eastern Atlantic stocks is 216,000t giving a total theoretical yield of 222,000t.

Question 2: *How does fishing at the individual MSY level for each population affect the other if catches can be taken from either population depending on the location of the fish?*

When the model is run taking average area catches into account but no mixing of stocks, both population's SSB fall below the target biomass of B_{MSY}. This suggests that once more complex spatial considerations are taken into account this method of MSY calculation is inappropriate.

The results for both movement scenarios are very similar. When either area or mixing between the two stocks is not taken into account in the calculation, neither population of Atlantic bluefin tuna in either movement scenario is maintained at B_{MSY}. When the catches are taken from whichever stock is present in each area, the western stock is shown to be buffered by the eastern population allowing it to maintain a higher SSB. The eastern population, however, is only marginally affected by the western stock maintaining a slightly lower SSB than B_{MSY}. The results for the movement scenarios examined here suggest that the western population is more sensitive to assumptions regarding movement than the eastern stock.

<u>Question 3:</u> What is the MSY for each population when the TAC for each is only removed from the respective population and is taken based on historical catches for each area and proportionally from the population present in each area?

With catches taken proportionally from the population present in each area, the MSY for both stocks decreases in either movement scenario. The MSY for the western stock decreases by 1,000t to 5,000t, and the MSY for the eastern stock decreases by 50,000t to 165,000t. This gives a total theoretical sustainable yield of 171,000t - a decrease of 52,000t from the MSY calculated when area and movement were not taken into account.

The declines seen in the MSYs are due to the spatial distribution of the two stocks. Whereas previously the catches were taken from their respective populations proportionally across all age classes, in this calculation the catches are divided into the different areas (Table 10). Consequently, for the western stock, 55% of the total catch is taken from the western Atlantic, where only juveniles are present, and 45% is taken from the adult population. As a greater proportion of the catch is taken from the juvenile population, fewer make it to spawn, so the MSY is lower. For the eastern stock, 14% of the total catch is taken from the eastern Atlantic and 86% is taken from the adult population. Consequently the fishing mortality rate on juveniles and adults is altered from the even mortality rate experienced in both the juvenile and adult populations in a non-spatially adjusted model to an uneven rate.

<u>Question 4:</u> What are the optimum levels of exploitation when managing the stocks together as opposed to separately?

In order to maximise the total yield from the species as a whole, analyses suggest that it would be better to maximise the eastern Atlantic catch and halt fishing in the western Atlantic. Maximising the eastern Atlantic catch produces a total yield about 6 times greater than that taken when the total fisheries yield across the whole Atlantic is maximised and 26 times greater than that taken when the western Atlantic catch is maximised. When catches are only taken from the eastern population the western stock is able to remain above B_{MSY} and may be maintained even with higher eastern Atlantic catches because the majority of catches are taken from the Mediterranean where the western stock is never present. When catches are taken from both populations the western stock reaches B_{MSY} much earlier than the eastern stock and falls below B_{MSY} if greater catches are taken from the eastern stock (due to the buffering of the western stock to eastern catches). The results provide additional support to the conclusion that the western population has a greater sensitivity to assumptions regarding movement than the eastern stock.

Question 5: *How long would it take both populations to recover to their biomass corresponding to MSY and virgin biomass from their present day levels if fishing was (a) completely halted, (b) reduced by 10%, (c) reduced by 20% or (d) reduced by 50%?*

If all fishing were completely halted it would take less than 20 years for both populations to recover to their respective B_{MSY} levels and up to 88 years to rebuild to their virgin biomass levels when following a deterministic path under either movement scenario. While this expected timeframe for recovery is short (from a biological and conservation point of view), a complete moratorium on Atlantic bluefin tuna fishing is unlikely to be socially or politically acceptable. In addition, multi-decadal timescales are very long in the scheme of fisheries management and when perceived in terms of a fisher's livelihood. This perception of time can be seen in ICCAT's 2010 Atlantic bluefin tuna stock assessment where scientists state that "For a long lived species such as bluefin tuna, it will take some time (> 10 years) to realize the benefit [of reducing catches]". For many people outside of the fishing industry, reducing catches to enable the recovery of a population nearing collapse would see 10 years as a reasonable timeframe. However, from a fisher's point of view, the same time period would mean almost a generation of fisherman affected. Managers will therefore have to balance these socio-economic concerns with the issue of rebuilding populations. Table 13 summarises the recovery timeframes for different levels of catch reduction.

Table 13. Summary of the predicted timeframes for recovery following a reduction in present day fishing effort

Fishing scenario	Stock	Movement scenario	Time to reach B_{MSY} (years)	Time to reach SSB_0 (years)
(a) completely halted	Western	1	8	65
		2	10	73
	Eastern	1	19	88
		2	18	77
(b) -10% off total catches	Western	1		
		2		
	Eastern	1		
		2		
(c) -20% off total catches	Western	1	67	
		2		
	Eastern	1		
		2		
(d) -50% off total catches	Western	1	15	
		2	22	
	Eastern	1	44	
		2	41	

Where no number is recorded, the population failed to reach the target levels.

In order to minimise the disruption to the fishing industry whilst allowing both stocks to rebuild to B_{MSY}, catch reductions of about 34% would be required under either movement scenario. However, the lower reduction in catches means that the impact of reduced catches will be felt for longer as it will take up to 180 years for both stocks to rebuild to B_{MSY}. This highlights the need for the fishing industry and politicians managing fisheries to determine the balance between short-term losses and long-term gains.

Main conclusions from results

From these results the main conclusions based on the two movement scenarios presented are:

- The western stock appears more sensitive to assumptions of stock movement and mixing than the eastern population supporting suggestions from previous analyses (e.g. Apostolaki *et al.* 2003; Apostolaki *et al.* 2004; Apostolaki *et al.* 2005; Safina and Klinger 2008).
- It is important to take area and stock movement considerations into account as much as possible when determining total allowable catches for the Atlantic bluefin tuna fisheries.
- In order to maximise the total catches of Atlantic bluefin tuna it may be better to cease fishing in the western Atlantic and to try to maximise the total catch from the eastern stock. While retaining both the western and eastern Atlantic fisheries may be more socially and politically acceptable, aims to maximise the total catch from each population result in a much reduced total yield.
- If a moratorium were placed on both Atlantic bluefin fisheries today, the timeframe for recovery (assuming no negative environmental conditions or capture within other fisheries) is medium to long-term (approximately 20 years for the eastern stock and 10 years for the western stock).
- A 34% reduction is the minimum required decrease in total catches for both stocks to rebuild to B_{MSY}. However, this increases the timeframe for recovery substantially; 124 years for the eastern stock and 25 years for the western population under movement scenario 1, and 180 years for the eastern stock and 40 years for the western stock under movement scenario 2.

Further work

The model developed within this chapter may become a powerful tool to determine possible outcomes of management scenarios on the recovery and sustainable management of the Atlantic bluefin tuna. The results presented here answer the initial research questions posed, however the model has considerable potential for further development.

Of particular interest would be an examination of the impact of political adjustment on total allowable catches (TACs). As discussed previously, the Atlantic bluefin tuna TACs have been subjected to high levels of political adjustment to scientifically advised TACs. Based on the results presented in Chapter 3 it is hypothesised that this can only have had a negative impact on these stocks. Therefore the political legalisation of overexploitation in these fisheries may take a considerable share of the blame when discussing the current poor status of the stocks. It would be interesting to run the model using the actual percentage overshoots each year from advised TACs to examine the potential historical impact that these decisions may have had. However, no database currently exists examining the historical trends of political adjustment for the Atlantic bluefin tuna. The availability of these data from ICCAT is limited and would require significant time spent searching stock assessment papers and advice from SCRS as well as the cooperation of ICCAT to gain documents published prior to 2009 as these are not available online[64].

To continue the theme of this thesis it would be interesting to run the model again adding the historical levels of political adjustment and incorporating marine protected areas into the management of the stocks. Chapter 4 showed that the establishment of high seas MPAs can be a positive, scientifically based political process and with current targets to establish these areas this has now become a feasible management strategy. This model may be used to identify key areas and timings (i.e. particular months) for protection and that MPAs may be able to buffer stocks against the negative process of political adjustment. The model may then be run into the future to examine the potential impact of implementing MPAs on stock recovery in a situation of status quo political TAC setting and scientifically based TAC setting.

Further movement, biological and exploitation scenarios may be considered by this model. Table 14 lists the model scenarios and variables that may be varied in future model simulations. In particular, scenarios that include movement of the western population into the east Atlantic might have very different implications for the status of the western stock. In addition to investigating additional movement scenarios, more complex management issues could also be explored under both deterministic and stochastic regimes. For the results presented within this chapter, fishing gears were amalgamated. Future development of the model could separate these into separate gears (e.g. purse seine, longline, etc) in order to allow differences in gear-specific catchabilities and localities to be considered. Economic analyses might then be conducted.

[64] http://www.iccat.int/en/assess.htm

Further research areas might therefore include: the impact of political adjustment; the application of marine protected areas as part of a management plan together with investigations into the most beneficial locations and total area to protect; the relationship between catches by gear and stock rebuilding; and economic analyses.

Table 14. List of model scenarios and variables that may be varied under different simulation scenarios

Movement Scenarios

- Movement patterns between areas
- Age dependent movement
- Stochastic movement

Biological Scenarios

- Beverton-Holt steepness value
- Type of stock-recruitment relationship
- Stochastic stock-recruitment relationship
- Proportion of the adult populations that spawn each year (to investigate the impact of the skipped spawning hypothesis)
- Age at maturity
- Growth mode, length-weight conversions

Exploitation Scenarios

- Separation of gear types
- Gear specific localities and selectivities
- Catch bias to favour older larger individuals
- Age of exploitation
- Marine protected areas – displacement (redistribution) of the effort from these areas or the removal of this effort
- TAC system introduced – political adjustment levels

Economic Scenarios

- Future work should also develop the model in order to be able to examine the economic impact of political adjustment, marine protected areas, etc.

5.6. Conclusions

The system simulated here is an approximation of the real Atlantic bluefin tuna fishery based on available data. It cannot encompass the full complexity of the actual biological or fisheries dynamics and management across the Atlantic. However, it provides a useful starting point for the analysis of complex management issues and an investigation into the impacts of management strategies such as spatial management - experiments that are hard to conduct in the field and require vast amounts of research. While the model presented here is not yet reliable for stock assessments due to a paucity of data, it is useful for determining trends, proportional changes and the response of populations to different management strategies.

The aim of developing this model was to help advance research into sustainable management of the Atlantic bluefin tuna as this is an issue that urgently needs attention. I have presented evidence on two currently conflicting political influences on fisheries management – the largely negative impact of political adjustment to total allowable catches and the positive impact of establishing marine protected areas. In the future this model may be used to examine whether these forces might be more closely aligned in order to create viable fisheries in the present whilst enabling recovery of stocks to ensure long-term sustainability.

Chapter 6.

Summary, Recommendations and Conclusions

6.1. Introduction

Worldwide, natural resources are suffering from overexploitation and unsustainable use. As human populations increase, so will the demand for resources. Marine resource managers must therefore find ways to achieve sustainable targets for marine populations in order to ensure the continuous supply of seafood long into the future. In order to achieve this, resource managers and policy makers will have to re-evaluate their relationship with science together with their priorities for management. In this thesis I have examined the difficult relationship between science and politics within the context of the total allowable catch (TAC) system of Europe and the process of establishing marine protected areas (MPAs) in the high seas. I also developed a model with the aim of bringing together these issues in the management of a trans-oceanic species, the Atlantic bluefin tuna.

This final chapter presents an overview of the scientific contributions and policy implications of these studies. The potential for further work on each of these research topics is discussed and several important requirements for sustainable marine resource management are identified.

6.2. Political adjustment in European fisheries

The extent to which European politicians have adhered to scientific recommendations on annual total allowable catches (TACs) from 1987-2011 was analysed in Chapter 2. Whilst political adjustment to TACs can be seen worldwide, Europe was chosen as a case study due to the planned reform of the Common Fisheries Policy (CFP) in 2012. For the 11 stocks examined, TACs were set higher than scientific recommendations in 68% of decisions. On average, politically-adjusted TACs averaged 33-37% above scientifically recommended levels and there was no evidence that the previous reform of the CFP in 2002 improved decision-making. This analysis was expanded in Chapter 3 to model the effects of such politically-driven decision-making on stock sustainability and it was found that political adjustment of scientific recommendations dramatically increase the probability of a stock collapsing within 40 years. Scenarios showed that that basing TACs on MSY targets is insufficient to maintain sustainable stocks under environmental uncertainty and bycatch. This problem is then further compounded by politically adjusting recommended TACs, reducing the likelihood of maintaining stocks at a sustainable level. Appendix 2, Appendix 3 and Appendix 4 present the published results from this research.

The practice of political adjustment reveals the negative political impact on management and its failure to integrate science into management. However, there are instances where political decisions for management are driven by, and based on, scientific knowledge. The establishment of the marine protected areas in areas beyond national jurisdiction of the North-East Atlantic is one such instance.

6.3. Establishing marine protected areas in areas beyond national jurisdiction

Marine protected areas, or fisheries closures, may be implemented as part of the precautionary principle. It has been suggested that whilst providing valuable habitat protection (Lubchenco *et al.* 2003) reserves may act as insurance against failed management, either through human miscalculation, stochastic events or environmental changes and may act to reduce uncertainty and instability within fisheries (Bohnsack 1996; Lauck *et al.* 1998; Grafton and Kompas 2005). In general, modelling studies have shown that as migration increases, the ability of MPAs to protect stocks from collapse decreases (Le Quesne and Codling 2009). However some studies provide evidence that reserves produce greater than expected yields irrespective of migration level (West *et al.* 2009) and that fisheries closures, when combined with other management measures to improve the sustainability of a stock, would increase the long term profits of the fishery (Armsworth *et al.* 2010). The general consensus in the scientific community is that MPAs are a useful tool for biodiversity conservation (as they increase the density, diversity, biomass and size of organisms, and allow the restoration of habitats (Halpern 2003; Stewart *et al.* 2009; Lotze *et al.* 2011)) while the merits of closures to benefit fisheries remains debated (e.g. Bess and Rallapudi 2007; Gray 2010). Despite this, targets have been set to establish a global network of MPAs[65] and consequently there are growing efforts to choose and establish these areas.

[65] For example, in 1995 Convention on Biological Diversity (CBD) Jakarta Mandate on Marine and Coastal Biodiversity obliged Parties to establish a global network of MPAs. This commitment was further elaborated to include the deadline of 2012 for the establishment of representative networks as agreed by the World Summit on Sustainable Development (WSSD) in 2002.

The process of establishing MPAs however is not simple, particularly in areas beyond national jurisdiction (ABNJ). MPAs are legal constructions created for the purpose of regulating human activity in specific geographical areas. Ideally, successful planning and establishment of MPAs depends on involvement of all affected parties from the start of the planning process as well as a firm scientific and legal basis. Chapter 4 discusses the process undertaken to establish the beginnings of a network of MPAs in ABNJ in the North-East Atlantic together with the scientific and legal challenges that had to be overcome. This Chapter emphasises lessons learned which may aid similar efforts elsewhere. However, while overcoming the challenges described and designating the set of MPAs presented in Chapter 4 is a significant step forwards, the potential benefits of these sites will only be realised if they are well-managed and ultimately this will require sustained political will, sufficient human and financial capacity, strong governance and stakeholder engagement as well as compliance from those persons utilising the North-East Atlantic.

This success story was driven by strong political commitment and based on the best available science. The positive integration of science into management by politicians ultimately ended up with a significant outcome.

6.4. Sustainable management of the Atlantic bluefin tuna

Political adjustment and competitive bargaining is endemic in Regional Fisheries Management Organisations across the globe. The performance of the International Commission for the Conservation of Atlantic Tunas (ICCAT) in managing stocks, such as the Atlantic bluefin tuna (*Thunnus thynnus*), has been highly unsatisfactory (Cullis-Suzuki and Pauly 2010). Competitive bargaining has led to quotas for this species becoming so over-inflated that they are now delivering what many see as the institutionalised extinction of a species (Safina and Klinger, 2008; Korman, 2011).

It has been identified that new management strategies are needed in order to prevent the collapse of the Atlantic bluefin tuna (e.g. MacKenzie *et al.* 2008; Armsworth *et al.* 2010; ICCAT 2010b). Possible strategies range from the listing of the species on Appendix 1 of CITES (although the most recent attempt in 2010 failed to achieve this) to stronger enforcement of the current TAC system to the creation of marine protected areas/fisheries closures. Should ICCAT apply fisheries closures as part of a rebuilding strategy for the Atlantic bluefin tuna similar challenges to those presented in Chapter 4 may have to be overcome.

This study develops a spatial model of the two stocks of Atlantic bluefin tuna with the aim of continuing research into the integration of science into a political management system to create a sustainable fishery. Initial results suggest that assumptions regarding the movement patterns of the two stocks may considerably affect model predictions for the status of both stocks, although particularly for the western stock. In addition, if fishing were reduced simulations indicate that the timeframe for recovery is often within a human lifespan and that once recovered catches, and therefore profits, will be much greater. If politicians were to reprioritise their main concerns from short-term aims to longer-term goals and to more fully incorporate science into management sustainable and more profitable Atlantic bluefin tuna fisheries might be achieved. However, greater application and analysis of this model is needed before policy may be suggested.

6.4. Policy implications

Political adjustment in European fisheries

The European Commission, on 22 April 2009, published a Green Paper on the Common Fisheries Policy which marked the start for the next reform, likely to be in effect by 2013 (CEC 2009). Following a consultation period, proposals to reform were released that include a phase out of fish discarding, broadening of multi-year species management plans, better data collection and a move to ecosystem based management[66]. Although the decision-making process was criticised in the Green Paper (2009), and the impact of “high political pressure to increase short-term fishing opportunities at the expense of […] sustainability” (CEC 2009: p7) was identified, proposed reforms leave untouched political competitive bargaining over total allowable catches. Consequently, the new proposals place no obligation on decision makers to achieve the sustainability to which the policy aspires. The results presented in Chapter 2 suggest that without this obligation the pattern of political adjustment is unlikely to change, as seen in the 2002 reform of the CFP. The model

[66] http://ec.europa.eu/fisheries/reform/

analysis of Chapter 3 provides support to this suggesting that the 2012 reform is likely to be similarly ineffective unless decision-making is changed so that catch allocations are based on science rather than politics. The implication of this research stretches beyond European fisheries and the 2012 CFP reform as competitive bargaining and political adjustment may be seen in fisheries around the world. This research shows that robust science should play an integral role in the drive towards fisheries sustainability and is thus essential, not optional. From this research it is suggested that scientific assessments should be used to set politically binding limits on catch sizes within which politicians can negotiate to divide scientifically appropriate quotas among member states.

Establishing marine protected areas in areas beyond national jurisdiction

The major goal and policy outcome of this research was the designation of the first network of high seas marine protected areas created under the OSPAR Convention. However, with global targets to establish well-managed networks of MPAs throughout the oceans this forms only the first step in this process. It is hoped that by disseminating the process undertaken to achieve this goal this will aid further establishment of MPA networks by providing inspiration on what can be achieved and guidance on how to overcome some of the challenges likely to be faced.

Sustainable management of the Atlantic bluefin tuna

The model presented within this thesis may become a powerful tool to determine possible outcomes of management scenarios on the recovery and sustainable management of the Atlantic bluefin tuna. Further investigation into the impact of political adjustment on stock status together with an examination of the potential benefits of MPAs for recovery would provide useful information for ICCAT and may provide impetus for future policy changes.

The current management of the two stocks of Atlantic bluefin tuna assumes very little mixing (1-2%) of the populations takes place (Apostolaki *et al.* 2003; ICCAT 2010b). Estimates of mixing are based on a few tagged individuals and are highly variable ranging from 2-20% for the western stock and 1-10% for the eastern population (Rooker *et al.* 2007). The two movement scenarios analysed in Chapter 5 show the effect that assumptions regarding movement may have on management parameter estimates and targets. It is therefore important for ICCAT to consider assumptions regarding movement in future policies. Further investigation into the movement patterns through tagging studies of both stocks of Atlantic bluefin tuna would therefore be useful.

6.5. Further work

Political adjustment in European fisheries

The analysis of total allowable catches in European fisheries presented in Chapter 2 is based upon averages on political adjustment for the 11 species examined. Future work to investigate the impact on specific species and stocks would be interesting as it would allow the economic impact of political adjustment to be determined, i.e. the cost of not following scientific advice could be determined. This would provide a valuable contribution to this work as the impact on fisheries and people would be made more relevant rather than simply presenting the biological evidence for fish species.

The model designed for Chapter 3 shows that political decisions to adjust advised TACs undermines their use and their scientific basis. However, models are always simplifications of reality and this one is no exception. The specific limitations to this model are listed on page 83 of this thesis. Further work could be completed by extending this model to incorporate greater uncertainty (for example with scientific advice), to investigate the impact of different life-history (e.g. fecundity, growth) characteristics, and to set political adjustment levels stochastically around a mean to try to incorporate the reality of fluctuating levels of adjustment each year.

Whilst these extensions might improve the 'reality' of the model, it is still felt that the general results presented here would not change. More interesting would be to develop the model into an age- and size-structured one which would allow the model to then investigate the impact of fisheries selectivity on stocks under a politically-driven total allowable catch system. In addition the incorporation of economics into the model would allow future trends to be predicted with regards to the continuing impact of setting total allowable catches higher than scientifically recommended and allow the comparison to a policy which fully incorporates scientific recommendations.

Establishing marine protected areas in areas beyond national jurisdiction

The network established represents the first elements of an ecologically coherent network of MPAs in areas beyond national jurisdiction. However, this is only a start and in order to create a representative network additional complementary sites will be needed. In addition, without the proper management MPAs are unlikely to deliver in their objectives and therefore appropriate management plans need to be developed. Nonetheless, it is hoped that this initial process has strengthened political will and will pave the way for future sites both in the North-East Atlantic and elsewhere.

Sustainable management of the Atlantic bluefin tuna

The results presented in Chapter 5 begin to explore the implications of movement assumptions on the predicted recovery of the two stocks of Atlantic bluefin tuna. However, the movement, biological and exploitation scenarios considered here are limited by the available data and as such the model has the potential to be explored much further.

Chapter 5 details specific additional research questions that may be answered by this model in the future however, to tie in with the theme of this thesis regarding the relationship between science and politics it is felt that immediate analysis of the impact of political adjustment on stock status would be interesting. Comparing these results to the status of stocks had science been followed would provide a basis for economic analysis to be carried out to investigate the cost of failing to follow scientific advice. In addition, future management strategies to rebuild the tuna populations may include the application of high seas marine protected areas. In Chapter 4 I have shown that the process of establishing MPAs may foster a positive relationship between science and politics. Within ICCAT themselves spatial management measures have already been discussed. Consequently, this model may provide useful data regarding the potential benefits of locating MPAs in different areas and at different times.

6.6. Concluding remarks

Overexploitation and unsustainable use characterises many fish stocks around the world. Improvements to management and utilisation of the world's marine resources are required. In this thesis I have shown the negative and positive relationship that science and politics may have in marine resource management. In the case presented in Chapter 4, I have shown that the integration of science into a political goal can have a major positive outcome. Where science fails to be integrated into policy, the negative effects can be seen as has been shown in the European decision-making system of total allowable catches. The research conducted within this thesis has contributed to the goal of sustainable marine management. In addition this work has also directly influenced policy. From this research several important requirements for sustainable fisheries management have been identified which include:

- **A scientific basis for fisheries management.** Scientific stock assessments and the resulting recommendations are unbiased and based on best available knowledge and data. In the EU and many other regional management bodies total allowable catch limits are often set undermining this advice to prevent short-term economic losses. However, in the long run this leads to unsustainable levels of exploitation, declining stocks and greater. Scientific advice for catch limits therefore must be followed and the practice of political adjustment to these limits must be eliminated.
- **Stakeholder engagement and cooperation**: Establishing new management strategies depends on cooperation amongst all affected parties. During the OSPAR process this was found to be essential not only for the designation of the network but also for their future management. As currently being seen in the Common Fisheries Policy reform this is likely to result in considerable debate and not all parties will be satisfied with all aspects of the final outcome. However, consultation provides a platform where compromises and assurances can be made and consequently greater buy in to changes is likely to be achieved.
- **Strong political commitment and willingness** are required to change policies and improve management. Without these targets and deadlines are unlikely to be established and management is unlikely to improve. Without

such commitment conflicts and complexities may become intractable and used as reasons to deter engagement.

This thesis has shown that when there is a positive relationship between science and politics significant outcomes can be achieved while a negative relationship may be detrimental to both the fish stocks and the fisheries which rely on them. It is hoped that future research into the Atlantic bluefin tuna using the model presented in Chapter 5 will provide insights into ways to change the largely negative relationship seen to date within ICCAT into a positive one resulting in sustainable fisheries while limiting the short-term socio-economic impacts.

Appendix 1.

Appendix 1 details the Mann-Whitney U values to complement Figure 6 and Figure 15 in Chapter 2 on pages 40 and 50 respectively.

Mann-Whitney U values for comparisons of zones where p<0.01

Management Zones	U, Z and N values	p value and effect size (r)	Mean rank
Baltic Sea (sub-areas 22-32) **Clyde & Rockall (VI)**	U = 1882, Z = 3.61, N = 119	$p = 0.0002$, $r = 0.33$	145.81 111.19
Baltic Sea (sub-areas 22-32) **Iceland (Va)**	U = 612, Z = 4.67, N = 126	$p = 1x10^{-6}$, $r = 0.42$	179.75 77.25
Baltic Sea (sub-areas 22-32) **Northeast Arctic (sub-areas I & II)**	U = 549, Z = -3.84, N = 103	$p = 8x10^{-5}$, $r= 0.38$	174.85 82.15
Baltic Sea (sub-areas 22-32) **North Sea (IV)**	U = 946, Z = 3.28, N = 55	$p = 0.0009$, $r = 0.44$	179.48 77.52
Baltic Sea (sub-areas 22-32) **Skagerrak & Kattegat (IIIa)**	U = 772, Z = -2.95, N = 112	$p = 0.0029$, $r = 0.29$	175.45 81.55
Baltic Sea (sub-areas 22-32) **West of the British Isles & English Channel (VII)**	U = 2659.5, Z = 3.48, N = 158	$p = 0.0004$, $r = 0.28$	181.75 75.25
Iceland (Va) **Irish Sea (VIIa)**	U = 7211.5, Z = 3.77, N = 214	$p = 0.0001$, $r = 0.26$	128.82 128.18
Northeast Arctic (sub-areas I & II) **Spain, Portugal & the Bay of Biscay (VIII/IX)**	U = 2151, Z = -5.42, N = 188	$p = 3x10^{-8}$, $r = 0.40$	132.52 124.48
Skagerrak & Kattegat (IIIa) **Spain, Portugal & the Bay of Biscay (VIII/IX)**	U = 3056.5, Z = -4.11, N = 197	$p = 3x10^{-5}$, $r = 0.50$	130.61 126.39
Spain, Portugal & the Bay of Biscay (VIII/IX) **Clyde & Rockall (VI)**	U = 2971, Z = -4.80, N = 204	$p = 5x10^{-7}$ $r = 0.35$	132.4 124.55
Spain, Portugal & the Bay of Biscay (VIII/IX) **Iceland (Va)**	U = 2496, Z = -6.70, N = 211	$p = 4x10^{-12}$, $r = 0.46$	142.45 114.55
Spain, Portugal & the Bay of Biscay (VIII/IX) **Irish Sea (VIIa)**	U = 5057, Z = -3.37, N = 235	$p = 0.0007$, $r = 0.22$	144.00 112.99
Spain, Portugal & the Bay of Biscay (VIII/IX) **North Sea (IV)**	U = 3766.5, Z = -4.74, N = 140	$p = 2x10^{-6}$, $r = 0.40$	141.51 115.49
Spain, Portugal & the Bay of Biscay (VIII/IX) **West of the British Isles & English Channel (VII)**	U = 4543, Z = -5.02, N = 243	$p = 4x10^{-7}$, $r = 0.32$	155.21 101.79

Mann-Whitney U values for comparisons of mean PAI values for species where $p<0.01$

Species	U, Z and N values	p value and effect size (r)	Mean rank
Megrim **Plaice**	U = 232, Z = 3.25, N = 36	p = 0.0007, r = 0.54	36.28 14.72
Megrim **Haddock**	U = 232, Z = -3.25, N = 36	p = 0.0007, r = 0.54	36.28 14.72
Megrim **Saithe**	U = 225, Z = 3.01, N = 36	p = 0.002, r= 0.50	36.00 15.00
Whiting **Plaice**	U = 387, Z = -4.47, N = 42	p = $1x10^{-6}$, r = 0.69	36.48 14.52
Whiting **Sole**	U = 356, Z = -3.68, N = 42	p = 0.0001, r = 0.57	35.24 15.76
Whiting **Haddock**	U = 399, Z = -4.78, N = 42	p = $9x10^{-8}$, r = 0.74	36.96 14.04
Whiting **Herring**	U = 377, Z = -4.22, N = 42	p = $6x10^{-6}$, r = 0.65	36.08 14.92
Whiting **Saithe**	U = 388, Z = -4.50, N = 42	p = $8x10^{-7}$, r = 0.69	36.52 14.48
Cod **Plaice**	U = 507, Z = 3.77, N = 50	p = $9x10^{-5}$, r = 0.53	33.28 17.72
Cod **Sole**	U = 454, Z = 2.75, N = 50	p = 0.005, r = 0.39	31.16 19.84
Cod **Haddock**	U = 520, Z = 4.03, N = 50	p = $2x10^{-5}$, r = 0.57	33.80 17.20
Cod **Herring**	U = 477, Z = 3.19, N = 50	p = 0.001, r = 0.45	32.08 18.92
Cod **Saithe**	U = 512, Z = 3.87, N = 50	p = $5x10^{-5}$, r = 0.55	33.48 17.52
Nephrops **Plaice**	U =461, Z = 4.82, N = 45	p = $1x10^{-7}$, r = 0.72	36.44 14.56
Nephrops **Sole**	U = 436, Z = 4.25, N = 45	p = $6x10^{-6}$, r = 0.63	35.44 15.56
Nephrops **Haddock**	U = 474, Z = -5.12, N = 45	p = $7x10^{-9}$, r = 0.76	36.96 14.04
Nephrops **Herring**	U = 452, Z = -4.61 N = 45	p = $5x10^{-7}$, r = 0.69	36.08 14.92
Nephrops **Saithe**	U = 458, Z = 4.75, N = 45	p = $2x10^{-7}$, r = 0.71	36.32 14.68

Appendix 2.

Appendix 2 reproduces a research paper published in Marine Pollution Bulletin 2011 based on the work presented in chapters 2 and 3 of this thesis. The reference of this article is: O'Leary, B.C. Smart, J.C.R., Neale, F.C., Hawkins, J.P., Newman, S., Milman, A.C., Roberts, C.M., 2011. Fisheries Mismanagement. Mar. Pollut. Bull. 62,2642-2648.

Fisheries Mismanagement

Bethan C. O'Leary[*a], James C.R. Smart[a], Fiona C. Neale[a], Julie P. Hawkins[a], Stephanie Newman[a1], Amy C. Milman[a], Callum M. Roberts[a]

[a] Environment Department, University of York, York, YO10 5DD, UK.
bcol500@york.ac.uk, jcs@dmu.dk, f_c_neale@hotmail.co.uk, julie.hawkins@york.ac.uk, SNewman@ieep.eu, amymilman@gmail.com, callum.roberts@york.ac.uk, Tel: 01904 434327, Fax: 01904 432998.

*Corresponding author.

---

[1] Present address: Institute of European Environmental Policy, 15 Queen Anne's Gate, London, SW1H 9BU, UK.

Abstract

We analysed the extent to which European politicians have adhered to scientific recommendations on annual total allowable catches (TACs) from 1987 to 2011, covering most of the period of the Common Fisheries Policy (CFP). For the 11 stocks examined, TACs were set higher than scientific recommendations in 68% of decisions. Politically-adjusted TACs averaged 33% above scientifically recommended levels. There was no evidence that the 2002 reform of the CFP improved decision-making, as was claimed at the time. We modelled the effects of such politically-driven decision-making on stock sustainability. Our results suggest that political adjustment of scientific recommendations dramatically increases the probability of a stock collapsing within 40 years.

In 2012 European fisheries policy will undergo a once-a-decade reform. Ten years ago radical reforms were promised but the changes failed to improve sustainability. It is likely that the 2012 reform will be similarly ineffective unless decision-making is changed so that catch allocations are based on science rather than politics.

Keywords

Common Fisheries Policy,

Fisheries management,

Total allowable catch, quotas, TAC

Political adjustment

1. Introduction

The complex and expensive system for managing European fisheries is one of the world's least successful, with many stocks now seriously depleted (Condé *et al.,* 2010). The last reform of the Common Fisheries Policy (CFP) in 2002 did little to halt stock declines and the degradation of European seas (Froese and Proelß, 2010). In recognition of these high profile deficiencies, the European Commission Green Paper on the 2012 Reform of the CFP, published in 2009 (CEC, 2009), called for radical reform, and since then the CFP has been under review. Much attention has focussed on discarding, with a high profile media campaign against this practice, and on economic incentives such as catch shares to encourage the industry to fish more responsibly (Costello *et al.,* 2008). However, based on the analyses we present in this paper, we contend that none of these measures will succeed in achieving sustainability unless the process for catch allocation is reformed to place science at the heart of decision-making.

Total allowable catches (TACs) and quotas have been the cornerstone of resource management within the CFP. Even with the continuing shift to multi-annual plans, TACs remain the practical basis for management, with the TACs being determined by fisheries ministers annually according to a set of rules laid out in each multi-annual plan (EU 2009). Scientists assess stocks in European waters each year and recommend TACs to the European Commission for each stock and fishing zone. Fisheries ministers then meet to negotiate TACs and allocate quotas amongst member nations. This decision-making process leads to the paradox of ministers' protecting national interests while attempting to allocate quotas among member states for mutual benefit and to achieve conservation goals. As the Rt. Hon. John Gummer, a former UK Fisheries Minister and Secretary of State for the Environment, put it, "If you are a fisheries minister you sit around the table arguing about fishermen - not about fish. You are there to represent your fishermen. You are there to ensure that if there are 10 fish you get your share and if possible a bit more. The arguments are not about conservation, unless of course you are arguing about another country” (NIA, 2001). This type of stance has led to ministers' consistently setting TACs higher than advised scientifically (Piet *et al.,* 2010; Villasante *et al.,* 2010).

We analysed historical TAC setting between 1987 and 2011 for 11 species across nine management zones to assess the degree to which politicians choose to adjust scientifically recommended TACs. Failure to follow scientific advice may be a major weakness, and possibly a fatal flaw, of the CFP. To assess the degree to which political adjustment may affect stock status, we then modelled the effects of politically driven decision-making for two different fish life histories under different scenarios of background environmental variability and juvenile bycatch.

2. Methods

2.1. Data collection

Data on advised and agreed TACs were obtained from the ICES online advice archives[67] and official EU Council Regulations (EU 2010a) and bilateral agreements (EU 2010b). In total, data were collected for 44 stocks of 11 fish species across nine management zones. The species analysed were cod (*Gadus morhua*), plaice (*Pleuronectes platessa*), haddock (*Melanogrammus aeglefinus*), megrim (*Lepidorhombus* spp.), saithe (*Pollachius virens*), herring (*Clupea harengus*), sole (*Solea* spp.), hake (*Merluccius merluccius*), nephrops (*Nephrops norvegicus*), sprat (*Sprattus sprattus*) and whiting (*Merlangius merlangus*). These species were chosen because the EU manages them under a TAC system. Stocks for which there was a mismatch between the areas for which TACs were advised by ICES and those for which TACs were set by ministers were excluded from our analysis. For example if the advice referred to area VIId and the TAC was set for areas VIId and VIIe these stocks were removed from the calculations. The nine management zones considered were the Baltic sub-areas 22-32, Skagerrak and Kattegat (IIIa), North Sea (IV), Northeast Arctic subareas I and II, Icelandic division Va, Clyde and Rockall (VI), West of the British Isles and the English Channel (VII), Irish Sea (VIIa), Spain, Portugal and the Bay of Biscay (VIII and IX).

2.2. Statistical analysis

A political adjustment index (PAI) was calculated as a measure of the degree to which ministers adjusted scientifically recommended TACs. This was the percentage

[67] http://www.ices.dk/advice/icesadvice.asp

by which the official TACs differed from the advised TACs for all decisions made for all stocks in all years (PAI = ((agreed TAC – advised TAC)/advised TAC) x 100. If science advised a TAC of zero (i.e. a moratorium), it was not possible to calculate the PAI as the percentage deviation from the advised TAC. Therefore, where a positive TAC was implemented when a moratorium had been advised in the same year, the PAI was arbitrarily set to 100%. In order to account for this arbitrary assignment, the data were analysed with moratoria years both included and excluded. A chi-squared test was used to determine whether decisions in which TACs were adjusted upwards significantly outweighed those in which TACs were adjusted downwards or those subjected to no adjustment.

Statistical comparisons of political adjustment amongst species and management zones were made using the Mann-Whitney U test in order to determine whether some species and areas were more prone to adjustment than others. The average adjustment across the stocks of each species was calculated for each year by taking the mean of the stocks' PAIs across the different management areas. In the case of hake, only one stock was included in the analysis. These figures were used to investigate temporal trends in political adjustment for each species. Differences in levels of political adjustment among management zones were compared by taking the mean of the PAIs for stocks present in the respective zones annually.

To determine the effect of the CFP reform in 2002, and the expansion of the EU in 2004, on political adjustment, the PAIs were compared for the years prior to 2004 and for the period 2004-2011. Whilst the CFP reform was finalised in 2002, it only came into force on January 1st 2003 and therefore the first TAC negotiations to take place under the new policy were those held in December 2003 concerning the TACs for the 2004 fishery.

All statistical tests were implemented using R (R Development Core Team, 2008).

2.3. Simulation modelling

A stochastic, single-species biomass model was developed to investigate the consequences of political adjustment on the recommended TACs. The model was based on a lagged recruitment, survival and growth model (Hilborn and Mangel, 1997) and extended to incorporate a TAC management system (see Appendix for mathematical framework and parameter values). Two life histories were examined: an early maturing species with prolific recruitment, such as herring, and a late maturing species with lower recruitment, such as cod. The impacts of continuously exceeding scientifically recommended TACs by fixed percentages over a management timeframe of 40 years were examined under two scenarios: the first included juvenile bycatch mortality, the second excluded it.

Recruitment each year was calculated according to the Beverton Holt (1957) stock-recruitment relationship and stochasticity was introduced in this component of the model to represent natural environmental variability. In the model, TACs were set annually and simulated scientific recommendations were generated as the equivalent harvest level required to maintain stocks at the biomass which corresponded to the MSY (B_{MSY}) for the species and stock concerned. Should the biomass of the stock in the model fall below a species- and stock-specific minimum biomass limit (B_{LIM}) the scientifically recommended quotas assumed that a proportional decrease in fishing mortality would be required to recover biomass. For biomasses above B_{LIM}, scientifically recommended TACs were set to MSY and then reduced (proportional to the undershoot below B_{LIM}) once the biomass reached, or fell below, B_{LIM} in order to rebuild the stock to B_{MSY}.

In the results reported here B_{LIM} was set at B_{MSY}. This level was chosen because in a well-managed fishery the catch target is set to maintain the stock at B_{MSY} and therefore once the stock biomass falls below B_{MSY} the TACs should be adjusted to allow the stock to rebuild to this level. Political adjustment was modelled by multiplying the scientifically recommended TAC by the adjustment level. Simulations were run for political adjustment levels of 0-50% in 1% increments. The level of political adjustment was held constant over the full timeframe for each simulation.

We also considered the effect of juvenile bycatch as in many fisheries juveniles are taken as bycatch and discarded. In general, demersal fisheries typically experience higher by-catch and discard rates than pelagic fisheries (Alverson *et al.,* 1994), with estimates reaching up to 94% for cod (ICES, 2009) compared to 11% for herring (Pierce *et al.,* 2002). Our modelling used an intermediate bycatch value of 50%. Juveniles were considered to be immature fish which are not subject to the TAC.

Juvenile bycatch was introduced into the system through recruitment adjustments during each year of a simulation run.

The model was run for each species until the stock collapsed (defined as depletion to 10% or less of the unexploited biomass) or for a maximum of 100 years. For each scenario 10,000 simulations were completed. The model was implemented in MATLAB (Mathworks, 2010).

3. Results

3.1. Levels of political adjustment of TACs

Table 1 summarises decisions made between 1987 and 2011 for 11 quota-managed fish species across nine management zones. In 68% of decisions ministers set TACs higher than the scientific recommendation. Only in 14% of cases were TACs set lower than advised; the remainder followed scientific advice. The difference in these proportions is significant [$\chi^2(2, N = 877) = 480.45$, $p < 0.001$] indicating that the process of political adjustment leads to ministers increasing scientifically advised TACs far more often than setting TACs at or below the scientifically advised level. On average, politically-adjusted TACs were 33% above the catch levels recommended as being safe by scientists. However, for some stocks TACs were routinely set more than 100% above scientific advice, and in one case – hake (Spain, Portugal and the Bay of Biscay management zone, sub-areas VIIIc and IXa) – scientific advice was exceeded by 1100% (Table 2).

Table 1. Decision-making and use of scientific advice under the Common Fisheries Policy from 1987-2011. Figures are shown ± 1 standard error.

	Pre-2002 Reform	**Post-2002 Reform**	**Overall**
Average Political Adjustment Index (PAI) (%)	27 ± 12	38 ± 14	33 ± 8
Average PAI (%) by fishing area*			
1. Spain, Portugal & Bay of Biscay	121 ± 24	92 ± 30	107 ± 19
2. Baltic	53 ± 8	104 ± 33	69 ± 12
3. Irish Sea	23 ± 3	44 ± 10	30 ± 4
4. Skagerrak & Kattegat	24 ± 4	25 ± 6	25 ± 3
5. West of the British Isles & English Channel	10 ± 2	61 ± 11	25 ± 4
6. North Sea			
7. Clyde & Rockall	19 ± 3	28 ± 6	22 ± 3
8. N.E. Arctic	14 ± 3	35 ± 7	19 ± 3
9. Iceland	24 ± 6	3 ± 3	17 ± 5
* Listed in descending order of average overall PAI	8 ± 2	11 ± 7	9 ± 2
% of decisions TACs set higher than recommended	71 ± 3	64 ± 6	68 ± 3
% of decisions TACs set lower than recommended	10 ± 2	18 ± 5	14 ± 3
% of decisions TACs set according to scientific advice	19 ± 4	18 ± 4	18 ± 3
Number of moratoria implemented/total number recommended	0/13	0/59	0/72

Table 2. Minimum and Maximum PAI for each species in the period 1987-2011.

Species	Extremes of PAI in time series (%)	
	Minimum	Maximum
Cod	-6	98
Plaice	-16	50
Haddock	-8	40
Megrim	-2	286
Saithe	-10	65
Herring	-5	51
Sole	1	51
Hake	-5	1100
Nephrops	9	283
Sprat	-12	58
Whiting	13	111

The 2002 CFP reform was meant to improve decision-making but our analysis found no evidence of progress (Fig. 1 and Fig. 2), other than a slight decrease in the percentage of TACs set higher than scientific advice and greater concordance with advice in the North-East Arctic area (Table 1). Instead, the majority of areas actually showed an increase in political adjustment to scientifically recommended TACs after the 2002 reform. However, Mann-Whitney U tests to compare the pre 2002-reform and post-reform PAIs found no significant difference in the level of PAI between the years before the reform and those after. This was the case for PAI averaged across all species ($U=545$, $N = 873$, $p=0.13$) and across management zones ($U=513$, $N = 825$, $p>0.1$). In addition, no individual species or management zones had significantly different levels of PAI during the pre- and post-reform periods. In addition, post-2002 there was a sharp increase in the number of fishing moratoria advised (which were all overruled), indicating continued disregard of scientific advice (Table 1 and Fig. 3).

Fig. 1. Extent to which scientific advice was accepted or adjusted by the Council of Ministers each year between 1987-2011 (expressed via a political adjustment index, PAI =((agreed TAC – advised TAC)/advised TAC)*100).

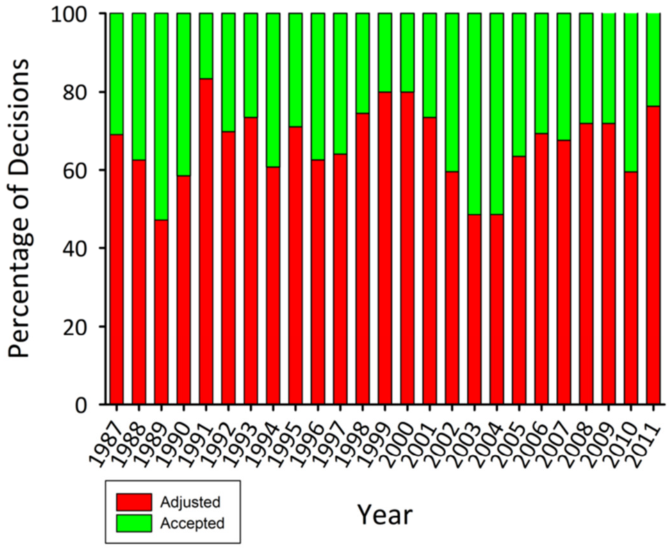


Fig. 2. Average political adjustment index across species between 1987 and 2011.

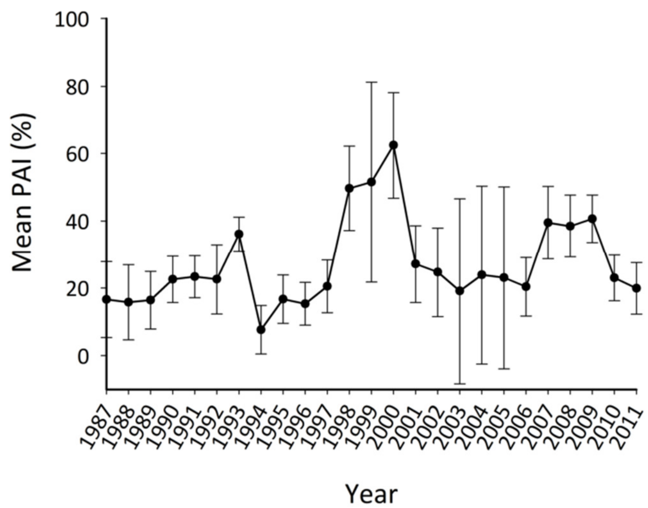
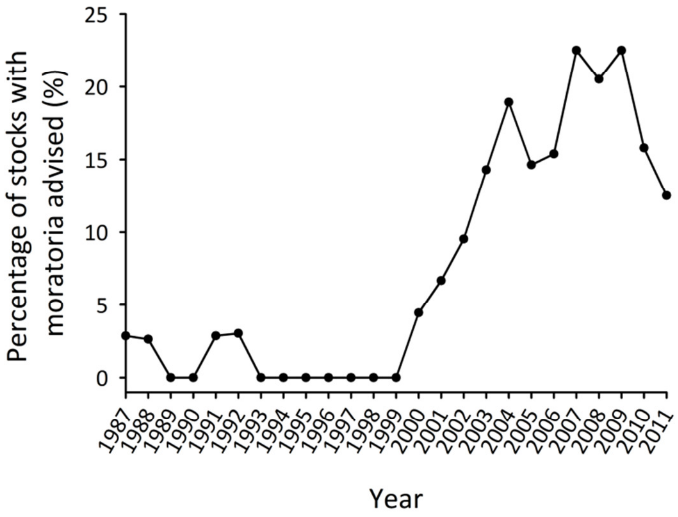


Fig. 3. Percentage of stocks for which moratoria were advised between 1987 and 2011.



A significant difference was found in the level of political adjustment between management zones using the Kruskal-Wallis test ($H8=72.52$, $p<0.01$). The two management zones most prone to political adjustment are the Spanish, Portuguese and Bay of Biscay zone (division VIII/IX) (107%) and the Baltic Sea (sub-areas 22-32) (69%). The waters around Iceland (division Va) are the least prone to political adjustment (9%) (Table 1). These results indicate that regional differences, such as differences in management, may be driving average PAI for different management zones.

It is possible that the potential benefits of the CFP reform have not been realised due to the expansion of the EU to include ten new member states in 2004[68]. New members are likely to have participated in the TAC decision-making process and this could have increased political adjustment simply by increasing the number of delegates, each bringing their own interests to the table (Thomson 2009). With subsequent expansion of the EU in 2007 to include Romania and Bulgaria this issue is likely to have become more prominent. The near coincidence in the timing of CFP reform and EU expansion meant that the two events could not be separated in our analysis.

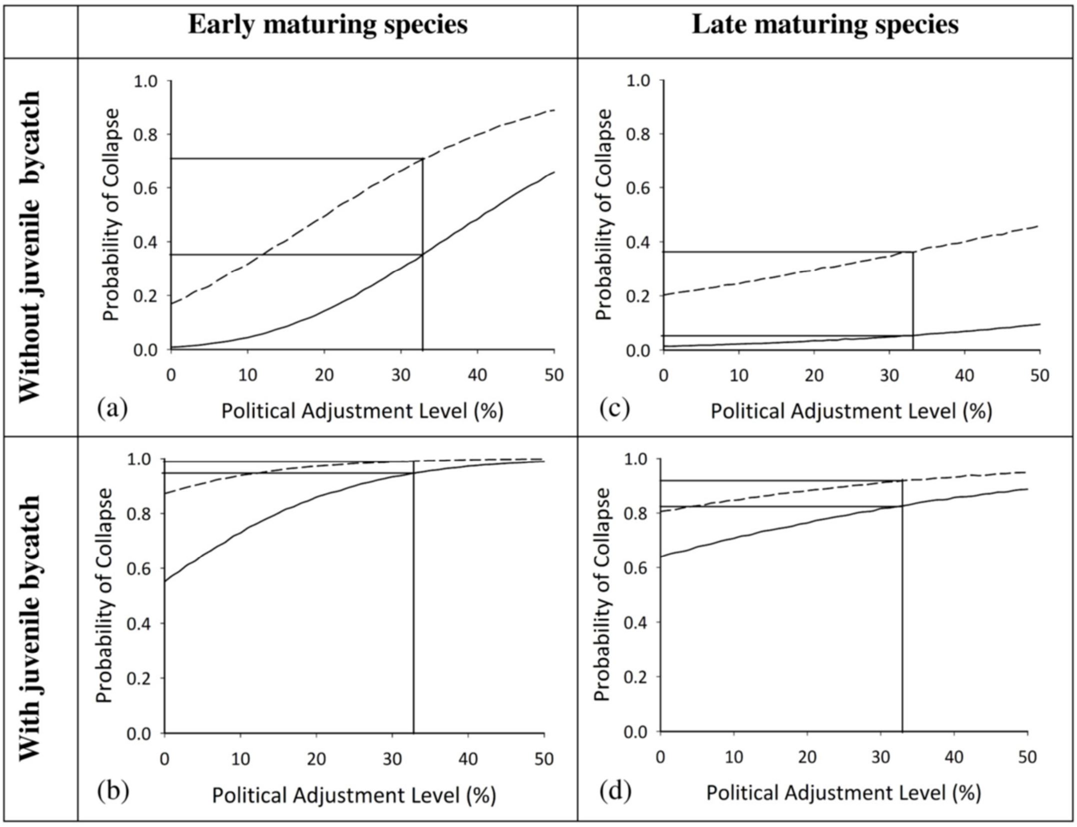
[68] Cyprus, Czech Republic, Estonia, Hungary, Latvia, Lithuania, Malta, Poland, Slovakia and Slovenia

The percentage of stocks for which moratoria were advised increased from 5% in 2000 to a maximum of 22% in 2007 and 2009 (Fig. 3). Advising a moratorium is a serious action, often regarded as a last resort by stock assessors. However, politicians appear not to appreciate the imminent risk of stock collapse which such a move implies as, for the stocks examined here, this zero-catch advice was not followed on any occasion; a TAC was always implemented in contradiction to the scientific advice. This could suggest naivety in managing natural resources, a lack of trust in scientific assessments or it may reflect the short-term goals of politicians and the industry which they oversee. The high number of moratoria advised in recent years paints a worrying picture of stock deterioration.

3.2. Simulation modelling

Figure 4 shows the calculated probabilities of stock collapse within 40 years, with and without political adjustment of TACs, for the early and late maturing fish life histories modelled.

Fig. 4. The effect of political adjustment on the probability of stock collapse within 40 years in modelled scenarios. (a) Probability of collapse for an early maturing species (such as herring). (b) Probability of collapse for an early maturing species when 50% of juveniles are caught as bycatch before recruitment to the targeted fishery. (c) Probability of collapse for a late maturing species (such as cod). (d) Probability of collapse for a late maturing species when 50% of juveniles are caught as bycatch before recruitment to the targeted fishery. All scenarios show lower resilience to political adjustment when subject to high environmental variability (solid lines) compared to low environmental variability (dashed lines). The probability of collapse at average levels of political adjustment observed under the Common Fisheries Policy (33%) is indicated by the solid vertical and horizontal lines. Details of the model can be found in the Appendix.



Our results suggest that environmental variability (modelled as variable recruitment success) increases the risk of fishery collapse, even when scientific advice is followed for both fish life histories. However, at both the levels of variability modelled, following scientific advice reduces the risk of collapse and promotes sustainability. Including juvenile bycatch in model simulations increases the risk of collapse within 40 years, in some cases increasing that risk to almost 100%. This emphasises the importance of efficient and target-specific fishing techniques and the need to limit juvenile bycatch within fisheries.

Under all scenarios, early maturing species appear to have less resilience to political adjustment than the later maturing species on the basis of probability of collapse within 40 years. The model takes into account species-specific fecundity parameters through the Beverton-Holt stock-recruitment relationship. Consequently, B_{MSY} for the late maturing species is a higher proportion of unexploited biomass than for the early maturing species. Because of this, management of early maturing species is more precarious since the gap between B_{MSY} and a state of collapse is lower, in proportional terms.

Our simulation results also confirm what managers already know: even if scientific advice is followed, it is difficult to manage fisheries sustainably in variable natural environments (Shelton and Mangel 2011). However, when we introduced TAC setting which mirrors the political adjustment seen under the CFP, the probability of stock collapse over a 40 year time horizon increases dramatically. When juvenile bycatch is included to reflect the situation in mixed species fisheries such as those for cod and haddock, the probability of collapse within 40 years for both of the modelled life-histories rose to between 83% and 99% (Fig. 4b and 4d). More worrying still is that these scenarios are 'best-case', in that the scientific advice within our model is based on perfect knowledge of stock status. In reality advice is generated in the face of uncertain stock sizes and fishing mortality rates.

Often the justification which fisheries ministers quote when adjusting scientifically recommended TACs is that uncertainty is inherent within stock assessment (Sovacool 2009, Khalilian *et al.* 2010), and that scientific advice is based solely on the biological aspects of the fishery and neglects socio-economic issues (Aps *et al.* 2007). While the uncertainty inherent in stock assessment cannot be denied (e.g. Hauge 2011), and it is agreed that TAC calculations ignore socio-economic issues, it could be argued that in the long-term the industry would be best served by following scientific advice to re-establish stocks from which higher yields could be extracted sustainably in the future.

4. Discussion

Since 1983, when the CFP came into force, sustainability has been the stated core goal of EU fishery policy. However, management under the CFP has driven many stocks further away from this objective. A meta-analysis of European fish stocks during the past half century showed that the CFP has failed to have any clear positive effect on marine fish stocks (Sparholt *et al.,* 2007). The decline of European fish stocks is well-documented, with 88% now overexploited relative to maximum sustainable yield targets and 46% fished outside safe biological limits (Condé *et al.,* 2010). Recent studies (Piet *et al.,* 2010; Villasante *et al.,* 2010) and our own research, reveal the scale of overshoot that competitive ministerial bargaining driven by short-term national interests has produced. It is clear from our analysis that political mismanagement must bear a considerable share of the responsibility for this decline. Decision-making by competitive bargaining has shown a reckless disregard for scientific advice which has been produced at a high cost to taxpayers. The remainder of the blame could be attributed to other management failures such as the shortcomings of quota management, heavy subsidies, high levels of discarding, high grading and non-compliance with fishery regulations. No matter how well these other deficiencies are addressed in the forthcoming reform of the CFP, productive and sustainable fisheries will not be achieved if Fisheries Ministers' cavalier disregard for scientific advice continues. Over the long-term, our simulation modelling suggests that such behaviour virtually guarantees the collapse of fish stocks, thus the practice amounts to institutionalised mismanagement of fisheries.

The EU commits considerable expenditure towards promoting resource conservation in fisheries through scientific research, stock assessments and in ensuring compliance with regulations and quota allocations; US$1.2 billion in 2003, according to one estimate (Sumaila *et al.,* 2010). In view of the sidelining of science in decision-making, it has to be questioned whether these expenditures are being allowed to deliver value for money. Our results, and those of others (e.g. Cardinale and Svedäng 2008), show that even with the uncertainty inherent in stock assessments, scientific research and advice should nevertheless be accepted because this provides the best opportunity to minimise the risk of stock collapse and maximise the long-term benefits from sustainable fishing. In reality, fisheries science is currently forced to act as a stakeholder around the political negotiating table rather than the lynchpin of management success. Consequently, political decisions are made in denial of biological reality.

Decision-making under the CFP should engage in a wider debate on the proper role of science in the management of renewable natural resources; should scientific advice be just another angle of a many-faceted political calculus, or should it be the foundation of a rationality which constrains political bargaining within the scope of natural productivity? There are good reasons to conclude the latter. No matter how much politicians might wish it otherwise, you cannot cheat nature out of more than she can produce, nor can you negotiate with her to increase production. If scientific advice is not followed, no amount of fine-tuning of harvest control rules or fishing methods will safeguard fisheries.

Similar failings extend far outside the European Union. Competitive bargaining is endemic in regional fisheries management organisations across the globe. For example, the performance of the International Commission for the Conservation of Atlantic Tunas (ICCAT) in managing stocks, such as the Atlantic bluefin tuna, has been highly unsatisfactory (Cullis-Suzuki and Pauly, 2010). Competitive bargaining has led to quotas for this species becoming so over-inflated that they are now delivering what many see as the institutionalised extinction of a species (Safina and Klinger, 2008; Korman, 2011).

In direct contrast, Icelandic and Norwegian fisheries have been subject to lower levels of political adjustment than those managed by the European Commission (Table 1). Aside from progressive policies on discards and the use of individual transferable quotas, fishery decision making in these regions appears to show a high level of respect for scientific advice and competitive bargaining appears much reduced (Eliasen *et al.* 2009). For example, Iceland has a long history of pre-emptively cutting quotas for groundfish, such as cod and saithe, following scientific concern regarding poor recruitment (Christensen *et al.* 2009). Such respect for science may have helped to maintain stocks and prevent overfishing when the number of recruits entering the fishery is low. Iceland and Norway are members of the European Economic Area rather than the European Union, and consequently their fisheries are subject to singular rather than multi-jurisdictional management

(Johnsen and Eliasen, 2011). There are therefore fewer 'players' around the negotiating table competing for quotas and the improved cohesion between science and politics which appears to have resulted has contributed to largely healthier stocks.

Fishers face multiple risks to their businesses: environmental variability, environmental change, unpredictable future catch levels, market prices and costs as well as a dependence on management decisions, many of which are erratic. While the environment is beyond the managers' control, science-based management offers the best way to reduce uncertainty about future catches and increase long-term revenues and sustainability.

The 2012 reform of the CFP is critical to the future of European fisheries because in another ten years some stocks and fisheries may be depleted beyond recovery. To accomplish the vision set out in the European Commission's 2009 Green Paper, "rampant overfishing" needs to end. This must begin with a respect for scientific advice. We suggest that scientific assessments should be used to set politically binding limits on catch sizes within which politicians can negotiate, not with nature, but with their peers to divide scientifically appropriate quotas among member states. If scientific advice continues to be sidelined, the 2012 reform of the CFP will once again fail to deliver sustainable and productive fisheries. On an increasingly crowded planet, it is now imperative that we adopt decision-making processes that enhance rather than undermine the sustainability of food production and natural capital.

Acknowledgements

The authors thank Bryce Beukers-Stewart and Jon Pitchford for their helpful comments. Funding was provided by the Economic and Social Research Council (ESRC) and the National Environment Research Council (NERC).

Appendix

Fish population dynamics are described in terms of biomass. In its discrete form this model can be expressed as:

$$B_{t+1} = sB_t + R_t - C_t \tag{A.1}$$

Where *s* describes the change in biomass (*B*) of the stock from one year (*t*) to the next (*t+1*) as a result of survivorship in the face of natural mortality only; *R* represents the recruitment to the population in year *t* and *C* is the catch taken from the stock in year *t*.

Recruitment (R_t) is derived from a re-parameterised version of the Beverton-Holt (1957) stock recruitment relationship which includes a steepness parameter (*z*) (Mace and Doonan, 1988) to represent recruitment, relative to the recruitment at equilibrium in the absence of fishing, that occurs when spawner abundance has been reduced to 20% of its virgin level (B_0). A high steepness value (*z*= 0.99) indicates almost constant recruitment that is essentially independent of the spawning stock biomass, while a low steepness value (*z*= 0.20) produces a proportional relationship between recruitment and spawning stock (Mace and Doonan, 1988). This method of re-parameterising the stock recruitment relationship allows the steepness of the curve to be altered while maintaining the same carrying capacity for the environment. Stochasticity (σ) is introduced into the system via the recruitment equation for each year of a simulation run. Recruitment is calculated using a deterministic equation which is then multiplied by a value drawn randomly from a uniform distribution spanning –1 to 1.

The reparameterisation of Beverton-Holt to include a *z* value defines the parameters α and β (necessary to calculate recruitment (Eq. (A.4)) as:

$$\alpha = \frac{B_0}{R_0}\left(1 - \left(\frac{z-0.2}{0.8z}\right)\right) \quad \text{(A.2)}$$

$$\beta = \frac{z-0.2}{0.8R_0} \quad \text{(A.3)}$$

Recruitment can thus be expressed as:

$$R_t = \left(\frac{B_{t-L}}{\alpha+\beta B_{t-L}}\right)\sigma \quad \text{(A.4)}$$

where $R_0 = B_0(1-s)$. (A.5)

Here, L refers to the time lag in years between birth and recruitment to the fishery. Recruitment in year t therefore depends on the stock biomass L years earlier. Consequently, two distinct age groups are modelled; recruits, i.e. fish aged less than L, and fish older than L years which are fecund, i.e. capable of producing new biomass.

Two levels of stochasticity were investigated, high and low. High stochasticity was set according to the level of variation that led to a 10% chance of stock collapse within 20 years. This scenario therefore produces highly variable recruitment which is realistic for many species (Needle, 2002). The low level of stochasticity was defined by the variation that led to a 0.5% probability of collapse within 20 years, producing a correspondingly low variation in recruitment.

Within this model the biomass at MSY (B_{MSY}), and the MSY itself, are defined as:

$$B_{MSY} = \frac{1}{\beta}\sqrt{\frac{\alpha}{1-s}} - \alpha \quad \text{(A.6)}$$

$$MSY = B_{MSY}\left(s - 1 + \frac{1}{\alpha+\beta B_{MSY}}\right) \quad \text{(A.7)}$$

A second dynamic element was integrated into this model by incorporating fluctuating catch quotas in direct response to the stock biomass in each year to represent political adjustment of the TAC system. The TAC was calculated by Eq. (A.9) where B_{LIM} is defined as B_{MSY} and $0.1B_0$ represents the 'collapse' threshold.

$$TAC = \left(\frac{B_t - 0.1B_0 \times MSY}{B_{LIM} - 0.1B_0}\right) \quad \text{(A.8)}$$

Juvenile bycatch is introduced into the model by adjusting recruitment during each year of a simulation run. Recruitment is calculated initially via Eq. (A.4) and then scaled by the juvenile bycatch rate (j) to produce:

$$R_t = \left(\left(\frac{B_{t-L}}{\alpha+\beta B_{t-L}}\right)\sigma\right) \quad \text{(A.9)}$$

Table 3 shows the parameter values used for our simulations.

Table 3. Parameter values for simulations

Parameter	Value	Description
Virgin biomass, B_0	100	Also referred to as environmental carrying capacity. Biomass is referred to throughout as a percentage of carrying capacity.
Survivorship, s	0.88	Survivorship was taken to be a function of natural mortality, taken as 0.2 based on estimates by Pauly (1980), and growth in mass of surviving individuals each year, taken to be 10%. Units: survival probability per year per individual.
Steepness of stock-recruitment curve, z	0.4, 0.7	Represents how steeply the Beverton-Holt stock recruitment curve ascends. No units - dimensionless.
α	0.8929, 3.125	Recruit production parameter.
β	0.0744, 0.0521	Recruit production parameter.
Lag time, L	2, 5	Lag time in years between reproduction and recruitment to the fishery.
B_{LIM}	B_{MSY}	The biomass level at which the TAC system is implemented.
Population collapse	$0.1B_0$	10% of virgin biomass (Worm *et. al.*, 2006).
Juvenile bycatch rate, j	0.5	Proportion of juveniles caught as bycatch.

References

Alverson, D.L., Freeberg, M.H., Murawski, S.A., Pope, J.G., 1994. A global assessment of fisheries bycatch and discards. FAO, 233p. http://www.fao.org/DOCREP/003/T4890E/T4890E00.HTM

Aps, R., Kell, L.T., Lassen, H., Liiv, I., 2007. Negotiation framework for Baltic fisheries management: Striking the balance of interest. ICES J. Mar. Sci. 64, 858-861.

Beverton, R.J.H., Holt, S.J., 1957. On the dynamics of exploited fish populations. London, Chapman and Hall.

Cardinale, M., Svedäng, H., 2008. Mismanagement of fisheries: Policy or science? Fish. Res. 93, 244 - 247.

CEC, 2009. Green Paper on the Reform of the Common Fisheries Policy. Commission of the European Communities. Brussels, COM(2009)163**:** 28p.

Christensen, A-S., Hegland, T.J., Oddsson, G., 2009. The Icelandic ITQ system, in: Hauge, K.H., Wilson, D.C. (Eds.), Comparative evaluations of innovative fisheries management: Global experiences and European prospects. Springer, pp. 97-118.

Condé, S., Jones-Walters, L., Torre-Marin, A., Romão, C., 2010. EU 2010 Biodiversity Baseline. No 12/2010:126p. http://www.eea.europa.eu/publications/eu-2010-biodiversity-baseline/

Costello, C., Gaines, S.D., Lynham, J., 2008. Can catch shares prevent fisheries collapse. Science. 321, 1678-1681.

Cullis-Suzuki, S., Pauly, D., 2010. Failing the high seas: A global evaluation of regional fisheries management organisations. Mar. Policy. 34, 1036-1042.

Eliasen, S., Sverdrup-Jensen, S., Holm, P., Johnsen, J.P., 2009. Nordic experience of fisheries management: Seen in relation to the reform of the EU Common Fisheries Policy. Nordic Council of Ministers. Copenhagen, TemaNord.

EU, 2009. Planning for the long term, in The Common Fisheries Policy: A user's guide**,** pp. 15-16.

EU, 2010a. COUNCIL REGULATION (EU) No 23/2010 of 14 January 2010 fixing for 2010 the fishing opportunities for certain fish stocks and groups of fish stocks, applicable in EU waters and, for EU vessels, in waters where catch limitations are required and amending Regulations (EC) No 1359/2008, (EC) No 754/2009, (EC) No 1226/2009 and (EC) No 1287/2009. O. J.

EU, 2010b. Agreed record of conclusions of fisheries consultations between the European Union and Norway for 2010. Brussels.

Froese, R., Proelß, A., 2010. Rebuilding fish stocks no later than 2015: Will Europe meet the deadline? Fish. Fish. 11, 194-202.

Hauge, K.H., 2011. Uncertainty and hyper-precision in fisheries science and policy. Futures. 43, 173-181.

Hilborn, R., Mangel, M., 1997. Management of hake fisheries in Namibia, in: The ecological detective: Confronting models with data. New Jersey, Princeton University Press: 235-262.

ICES, 2009. Cod in Subarea IV (North Sea), Division VIId (Eastern Channel), and IIIa West Skagerrak. http://www.ices.dk/products/icesadvice/2009/ICES%20ADVICE%202009%20Book%206.pdf

Johnsen, J.P., Eliasen, S., 2011. Solving complex fisheries management problems: What the EU can learn from the Nordic experiences of reduction of discards. Mar. Policy. 35, 130-139.

Khalilian, S., Froese, R., Proelss, A., Requate, T., 2010. Designed for failure: A critique of the Common Fisheries Policy of the European Union. Mar. Pol. 34, 1178-1182.

Korman, S., 2011. International management of a high seas fishery: Political and property-rights solutions and the Atlantic bluefin. VA. J. Int. Law. 51, 697-748.

Mace, P.M., Doonan, I., 1988. A generalised bioeconomic simulation model for fish dynamics. New Zealand Fishery Assessment Research Document no. 99/4. Wellington, New Zealand.

Mathworks, 2010. Matlab version 7.1. 2010a. The Mathworks Inc.

Needle, C.L., 2002. Recruitment models: Diagnosis and prognosis. Rev. Fish. Biol. Fish. 11, 95-111.

NIA, 2001. The Common Fisheries Policy: Issues, Impact and Future of the CFP. Northern Ireland Assembly, Research Paper 37/01:22p. http://www.niassembly.gov.uk/io/research/3701.pdf

Pauly, D., 1980. On the interrelationships between natural mortality, growth parameters, and mean environmental temperature in 175 fish stocks. ICES J. Mar. Sci. 39, 175-192.

Pierce, G.J., Dyson, J., Kelly, E., Eggleton, J.D., Whomersley, P., Young, I.A.G., Santos, M.B., Wang, J., Spencer, N.J., 2002. Results of a short study on by-catches and discards in pelagic fisheries in Scotland (UK). Aquat. Living. Resour. 15, 327-334.

Piet, G.J., van Overzee, H.M.J., Pastoors, M.A., 2010. The necessity for response indicators in fisheries management. ICES J. Mar. Sci. 67, 559-566.

R Development Core Team, 2008. R: A language and environment for statistical computing. Vienna, Austria, R Foundation for Statistical Computing.

Safina, C., Klinger, D.H., 2008. Collapse of bluefin tuna in the western Atlantic. Conserv. Biol. 22, 243-246.

Shelton, A.O., Mangel, M. 2011. Fluctuations of fish populations and the magnifying effects of fishing. PNAS 108, 7075-7080.

Sovacool, B., 2009. A game of cat and fish: How to restore the balance in sustainable fisheries management. Ocean Dev. Int. Law 40, 97-125.

Sparholt, H., Bertelsen, M., Lassen, H., 2007. A meta-analysis of the status of ICES fish stocks during the past half century. ICES J. Mar. Sci. 64, 707-713.

Sumaila, U.R., Khan, A.S., Dyck, A.J., Watson, R., Munro, G., Tydemers, P., Pauly, D., 2010. A bottom-up re-estimation of global fisheries subsidies. J. Bioecon. 12, 201-225.

Thomson, R., 2009. Actor alignments in the European Union before and after alignment. Eur. J. Pol, Res. 48, 756-781.

Villasante, S., García, M.d.C., González-Laxe, F., Rodríguez, G.R., 2010. Overfishing and the Common Fisheries Policy: (un)successful results from TAC regulation. Fish. Fish. 12, 34-50.

Worm, B., Barbier, E.B., Beaumont, N., Duffy, E., Folke, C., Halpern, B.S., Jackson, J.B.C., Lotze, H.K., Micheli, F., Palumbi, S.R., Sala, E., Selkoe, K.A., Stachowicz, J.J., Watson, R., 2006. Impacts of biodiversity loss on ocean ecosystem services. Science. 314, 787-790.

Appendix 3.

Appendix 3 reproduces a letter published in Nature 2011 based on the work presented in chapters 2 and 3 of this thesis and published in response to Froese (2011). The reference of this article is: O'Leary, B.C., Roberts, C.M., 2011. Fishery reform: ban political haggling. Nature. 475: 454.

Fishery reform: ban political haggling

We applaud proposals by the European Commission to reform the Common Fisheries Policy by phasing out fish discarding, broadening multi-year species-management plans, improving data collection and moving to ecosystem-based management (Nature 475, 7; 2011). But one vital reform has been missed: bargaining over total allowable catches should be banned and decision-makers should be compelled to follow scientific advice.

Politicians have habitually overruled scientific advice on fisheries since inception of the EC policy in the 1980s, setting total allowable catches one-third higher than recommended levels. Placing short-term political expediency and industry lobbying ahead of long-term sustainability threatens food security and the health of future generations.

Science provides the best tools for maximizing immediate benefits from fishing without squandering future opportunities. Let politicians argue for their national share of what nature can provide, rather than adopting policies that undermine the biological basis of food production. Politicians must cede their power over fisheries if they are properly to serve the public interest.

Bethan C. O'Leary, Callum Roberts *University of York, UK.*
callum.roberts@york.ac.uk

Appendix 4.

Appendix 4 reproduces a Response to a Comment on the article 'Fisheries Mismanagement' by Cook *et al*. (in review).

Response to Cook *et al*. Comment on "Fisheries Mismanagement"

Bethan C. O'Leary[*a], James C.R. Smart[a], Fiona C. Neale[a], Julie P. Hawkins[a], Stephanie Newman[a˺], Amy C. Milman [a], Samik Datta[b] Callum M. Roberts[a]

[a] Environment Department, University of York, York, YO10 5DD, UK. bcol500@york.ac.uk, jcs@dmu.dk, f_c_neale@hotmail.co.uk, julie.hawkins@york.ac.uk, SNewman@ieep.eu, amymilman@gmail.com, callum.roberts@york.ac.uk, Tel: 01904 434327, Fax: 01904 432998.

[b] School of Life Sciences, Gibbet Hill Campus, The University of Warwick, Coventry, CV4 7AL. S.Datta@warwick.ac.uk

*Corresponding author.

In O'Leary *et al.* (2011) we presented an analysis of 24 years of historical data regarding political decision-making on fisheries total allowable catches (TACs) within Europe, together with a simple stochastic model examining the impact that this could have had on the status of European fish stocks. In response to Cook *et al.* (in review) we agree some minor corrections to our article 'Fisheries Mismanagement' are appropriate. However, the major premise of their comments that our conclusions are flawed is incorrect.

˺ Present address: Institute of European Environmental Policy, 15 Queen Anne's Gate, London, SW1H 9BU, UK. Tel: 020 7340 0928

Cook *et al.* (in review) state that from our simulations we had concluded that the consequence of continuing to set total allowable catches (TACs) above scientific advice will lead to the collapse of stocks in the next 40 years. They misinterpreted the paper as we did not state this. Instead the model indicated, based on our calculated values of political inflation of TACs above scientific advice, that such decision making would substantially increase the likelihood of stock collapse over a forty-year timeframe. We therefore concluded that political decision making has likely contributed to the poor state of European fisheries resources today.

Cook *et al.* (in review) made six key comments on our article. These are reiterated below for clarity prior to giving our response to each.

1. There are errors in the equations that describe the model.

Cook *et al.* (in review) did spot typographical errors in the equations that describe our model. We have rectified these and the amended equations may be found in the appendix to this paper (equations A.1 to A.9). However, the results in the paper were based on the correct equations and so are unaffected. In addition, the following print errors were made. The results presented in Table 1 were shifted down a line so that the North Sea results and those below it were not in line with the appropriate side headings. This blank space should actually fall where it is stated that the results are "listed in descending order of average overall PAI". The heading for Figure 4 should read that the high stochasticity results are represented by dashed lines and the low stochasticity results by solid lines.

As recognised by Cook *et al.* (in review) equation A.4 as written in the original paper 'Fisheries Mismanagement', would lead to low levels of recruitment due to the presence of the Beverton-Holt parameters alpha and beta in the denominator. The corrected equation for beta (Equation A.3 as written in the appendix) limits this effect due to the presence of z within the denominator. With regards to the zero and negative values that may be produced by this equation (due to sigma being able to take values between -1 and 1) this was included to allow the equation to represent particularly bad years of recruitment due to environmental variability or other negative shock events.

2. *The stochastic noise used to simulate recruitment variability is unrealistic.*

As Cook *et al.* (in review) points out, we used a uniform error distribution where all events within a specified range are equally likely. While Cook *et al.* (in review) highlight that this means that very poor recruitment is as likely as average recruitment, it also means that very good recruitment is just as likely. However, log-normal or gamma distributions are often used and so we ran the model again with this formulation of recruitment variability, as recommended by Cook *et al.* (in review). We present in Figure 1 and Table 1 the results of applying our reformulated model to the species cod and herring using a lognormal distribution for recruitment variability, based on data from Myers *et al.* 1999. Table 2 lists the parameter values used for these simulations.

The effect of changing the method of applying stochasticity and using specific recruitment variability parameters acts to lower the probabilities of collapse. When compared to the original (non-species specific) results presented in O'Leary *et al.* (2011) the general trends are similar in that the probability of collapse increases with increasing political adjustment and that increasing unaccounted juvenile mortality increases the probability of collapse sharply for both species (Figure 1 and Table 1**).** The exact probabilities of collapse are however not directly comparable due to the different life history characteristics being examined. Whilst the probabilities of collapse are found to be lower using our reformulated model our conclusions hold. In reality, as there is likely to be greater levels of uncertainty, both within the stock-assessments that determine scientific advice and the mortality associated with the fisheries catches themselves, the higher unaccounted mortality scenarios are felt to more realistically represent true fishery conditions.

3. *The model is forced to produce high stock collapse rates without reference to real parameter values.*

We manipulated the amount of recruitment variability to represent the impact of exploiting fish stocks subject to both stable and highly variable environmental conditions. Because the fish stocks originally modelled were hypothetical examples, assignation of specific recruitment variability parameters was as an issue. We also gave consideration to the fact that fishing can elevate the variability of recruitment in exploited species (Hsieh *et al.* 2006). Our conclusion was that selecting a 10%

chance of stock collapse under management at maximum sustainable yield would represent a combination of environmental and anthropogenically-caused variability. However, we accept the concern of Cook *et al.* (in review) that this is an arbitrary choice of level of collapse and their argument that a more realistic way to approach the parameterisation of environmental variability is to parameterise the recruitment error distribution from observations. Consequently our new simulations specifically use cod and herring as examples with recruitment variability parameterised according to Myers *et al.* (1999) (Figure 1, Table 1).

The data analysed within O'Leary *et al.* (2011) provides ample evidence of the historical trends of risky fisheries decision-making. Over the period that the Common Fisheries Policy has been in place, 68% of the decisions set TACs higher than scientific recommendations. The aim of our model was to provide evidence that politicians' consistent disregard for scientific advice has contributed to the current status of European fish stocks by examining several simple scenarios. Maximum sustainable yield (MSY) was used as a base fishing rate for these scenarios due to its simple theoretical basis. The parameters within our model were set according to the concept of MSY and based upon Mace and Doonan's (1988) re-parameterisation. The scenarios investigated within our paper were not intended as complex stock assessment predictions but provide accessible simulations of factors which have contributed to the current situation.

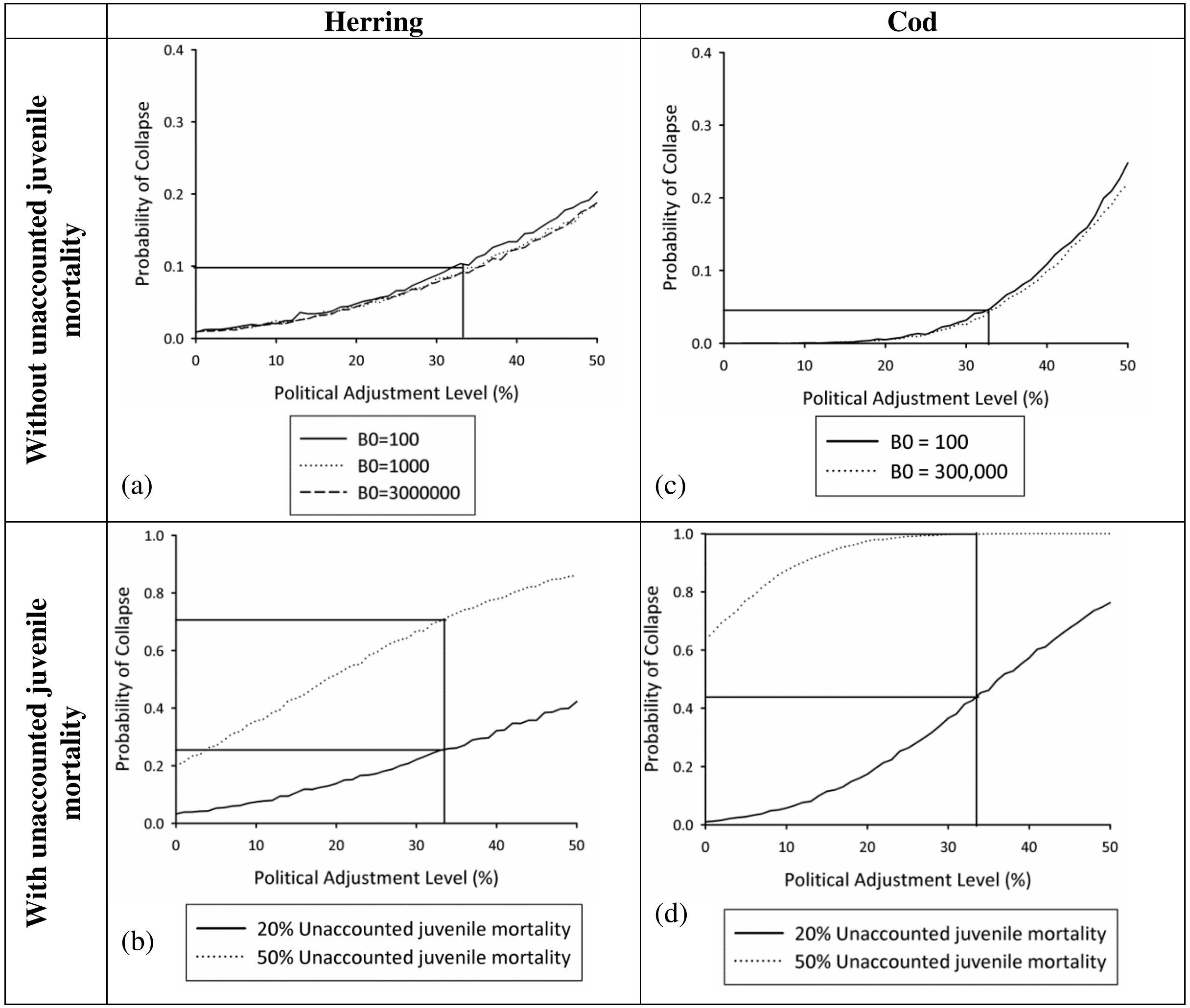
4. *The life history characteristics of the model fish stocks are not representative of the fish species concerned.*

In our paper 'Fisheries Mismanagement' we write that according to the model simulations, "early maturing species *appear* to have less resilience to political adjustment than the later maturing species". We then go on to clarify that management of the "early maturing species is more precarious" because of the application of specific fecundity parameters through the Beverton-Holt stock-recruitment relationship. However, Cook *et al.* (in review) state that we argue that late maturing species are more vulnerable to fishing than early maturing species. This was not our conclusion. Following on from this, Cook *et al.* (in review) make note of our use of virgin biomass or carrying capacity and provide a full explanation of the effect this has on the relative biomass at MSY (B_{MSY}) of each life history characteristic. Within our paper we allude to this by stating that the "B_{MSY} for the late maturing species is a higher proportion of unexploited biomass than for the early maturing species". Cook *et al.* (in review) only provide a more detailed description of this effect. We agree that the result is that we compare the performance of two values of steepness and the effect of different ages of maturity, rather than full life history traits. However, in order to compare the effect on two differing species directly, other characteristics such as age-related mortality and fecundity would have to be taken into account. This goes beyond the aim of the model. In addition, it is worth noting that the examples of cod and herring as a late and early maturing species respectively were intended to provide an illustration for the general reader rather than specific case studies. The simulations presented in this response intend to go some way further into representing more realistic populations of and herring.

In addition, Cook *et al.* (in review) argue that the arbitrary (and admittedly unrealistic) value of 100 assigned to the virgin biomass of both species fails to capture full life-history traits and their comment suggests that allowing these values to vary by species would alter the results of the model. However, further simulations presented in Figure 1(a and b) indicate that the model is unaffected by changing levels of virgin biomass. Instead, the probability of collapse remains constant (slight variations are due to the stochastic nature of this model) for each species at all levels of virgin biomass.

Figure 1 and Table 1 show that when no additional juvenile mortality is present the probability of collapse for herring is 1% when scientific advice is followed (MSY) and 10% at a political adjustment level of 33%. The probability of collapse for cod is 0% at MSY and 5% at a political adjustment level of 33%. When unaccounted juvenile fishing mortality is applied, the probability of collapse increases dramatically to between 23-41% under a 20% mortality scenario and between 68-100% under a 50% mortality scenario (Figure 1 and Table 1). Whilst these are lower than those from our original results, once even the lower level of unaccounted juvenile mortality is applied they are still worryingly high and political adjustment represents a poor management strategy.

Figure 1. The effect of political adjustment on the probability of stock collapse within 40 years in modelled scenarios



(a) Probability of collapse for herring where unexploited biomass, B_0 = 100, 1000 and 3,000,000, (b) Probability of collapse for herring when 20% and 50% of juveniles experience unaccounted mortality before recruitment to the targeted fishery where B_0 = 100, (c) Probability of collapse for cod where B_0 = 100 and 300,000, (d) Probability of collapse for cod when 20% and 50% of juveniles experience unaccounted mortality before recruitment to the targeted fishery where B_0 = 100. Scenarios a and c show that the value assigned to B_0 has no effect on the probability of collapse results. Scenarios b and d only show the results for B_0 = 100 to improve the clarity of presentation. Full results for all values of B_0 are presented in Table 15. The probability of collapse at average levels of political adjustment observed under the Common Fisheries Policy (33%) is indicated by the solid vertical and horizontal lines. Details of the model can be found in the Appendices of O'Leary et al. 2011 and this paper (recruitment equation A.10 replaces equation A.9 for this simulation).

Table 1. Probability of Collapse (%) when Scientific Advice is Followed (MSY) and when TACs are adjusted by the average PAI (33%) with and without unaccounted juvenile mortality for herring and cod.

Herring						
	No unaccounted juvenile mortality		**Unaccounted juvenile mortality level**			
			20%		**50%**	
	MSY	**33%**	**MSY**	**33%**	**MSY**	**33%**
$B_0 = 100$	0.82	10.35	3.23	25.37	20.12	70.28
$B_0 = 1000$	0.84	8.94	2.81	23.70	19.52	68.10
$B_0 = 3{,}000{,}000$	0.91	9.07	2.98	22.78	19.92	68.21
Cod						
	No unaccounted juvenile mortality		**Unaccounted juvenile mortality level**			
			20%		**50%**	
	MSY	**33%**	**MSY**	**33%**	**MSY**	**33%**
$B_0 = 100$	0	4.78	1.04	42.73	63.48	99.80
$B_0 = 300{,}000$	0	4.53	1.02	40.51	57.54	99.59

Table 2. Parameter values for simulations

Parameter	Value		Description
	Herring	Cod	
Virgin biomass, B_0	100 1000 3,000,000	100 300,000	Also referred to as carrying capacity. We modelled the virgin biomass of herring at 100, 1000 and 3,000,000 and cod at 100 and 300,000. These values were chosen according to the estimates of North Sea cod and herring suggested by Cook *et al.* (in review) of 300,000 tonnes and 3,000,000 tonnes respectively. The value of 1000 for herring was to examine whether there was any difference from our initial value of 100 and a biomass an order of magnitude greater.
Survivorship, s	0.88	0.88	Survivorship was taken to be a function of natural morality, taken as 0.2 based on estimates by Pauly (1980), and growth in mass of surviving individuals each year, taken to be 10%.
Steepness of stock-recruitment curve, z	0.74	0.84	Represents how steeply the Beverton-Holt stock recruitment curve ascends. Values taken from Myers *et al.* (1999).
α	0.3968	0.732	Recruit production parameter
β	0.0794 0.0794 2.65^{-5}	0.076 0.0076 2.53^{-6}	Recruit production parameter
Lag time, L	3	4	Lag time in years between reproduction and recruitment to the fishery. Estimates for North Sea cod by Nash *et al.* (2010) indicate the age-at-50% maturity falling between 2 to 4.5 years with females maturing older than males. ICES estimates that on average North Sea Herring matures at age 3[69]).
B_{LIM}	B_{MSY}		The biomass level at which the TAC system is implemented.
Lognormal distribution parameters	Mean = 0.73, Variance = 1.31	Mean = 0.84 Variance = 0.37	See Myers *et al.* (1999). Mean values are logged.
Population collapse	$0.1B_0$		10% of virgin biomass (Worm *et al.* 2006).
Unaccounted juvenile mortality rate, j	20% 50%		Proportion of juveniles caught through unaccounted mortality.

[69] ICES HAWG Report 2010, Annex 3 – Stock Annex North Sea Herring. Available at: http://www.ices.dk/reports/ACOM/2010/HAWG/Annex-03%20Stock%20Annex%20-%20North%20Sea%20Herring.pdf

5. *The assumption that increasing the TAC always increases the out-turn catch is incorrect.*

Cook *et al.* (in review) highlight the fact that the TAC is often taken from both mature and immature fish and argue that consequently our simulations of juvenile bycatch were unrealistic. We agree that landings are often made which include juvenile fish and these count towards the overall TAC for that species. It is also true that juvenile fish may be a component of the calculated MSY. In fact, it has been suggested that it would be better to exploit the whole age structure of a species rather than targeting the larger older individuals of a population (Law *et al.* 2012). However, whilst this may be the case we modelled additional juvenile bycatch (and mortality) to the 'adult' TAC as often juveniles are discarded due to being undersized or low value (known as high grading) (Gillis *et al.* 1995; Catchpole *et al.* 2005). Even if the value of 50% used within our simulations is high as claimed by Cook *et al.* (in review) the point of running these scenarios was to illustrate that over-quota mortality, be it juvenile mortality, the result of high grading or illegal, unreported and unregulated fishing, will worsen the situation created by politicians when they consistently ignore scientific advice in decision-making. The simulations presented here show that even if the unaccounted juvenile mortality rate is lower at 20%, the probability of collapse still rises considerably with political adjustment (Figure 1 and Table 1).

Cook *et al.* (in review) also argue that the uncertainties relating to stock assessment, modelling and implementation are such that the simplistic target of managing fisheries at B_{MSY} may not succeed in maintaining a sustainable fishery. We do not dispute this and also recognise that ICES harvest control rules (HCRs) are tested to include various considerations of uncertainty. Within our simulations, for simplicity we assumed scientists have perfect knowledge regarding the status of stocks. Consequently, for our purposes B_{MSY} was deemed to be an appropriate target. In addition, Cook *et al.* (in review) argue that the simple HCR we applied to our model in O'Leary *et al.* (2011) may be inherently risk prone. The HCR used is based on the concept of MSY, with a precautionary biomass of B_{MSY}. TACs decline proportionally when stocks fall below B_{MSY} and are set to zero when stock size reaches 10% of B_0. In fact, a similar HCR was presented by Froese *et al.* (2011) as a more sustainable method of calculating total allowable catches within European fisheries. Froese *et al.* (2011) showed that the application of this type of rule could have prevented past fisheries collapses and would be able to deal with strong cyclic variations in recruitment. Consequently, while our HCR may not reflect those used in the management of many stocks at present, we think it is a valid generic model to evaluate trends over time.

6. The assumption that juvenile fish are excluded from the TAC is incorrect.

Further, taking examples from the North Sea and the West of Scotland, Cook *et al.* (in review) argue that while our dataset shows that for the stocks studied 68% of decisions set total allowable catches higher than scientifically recommended, actual out-turn catch achieved is often below the overall TAC set, taking examples from the North Sea and the West of Scotland. The mean shortfall of catch they report is -0.17, a value that is approximately half of the mean TAC inflation factor that we report in O'Leary *et al.* (2011). Consequently, their argument that this catch shortfall balances out our TAC inflation factor is wanting. In addition, it is worth asking why these TACs are not filled. Presumably this is not because fishers note the inflation factor and seek to reduce their catches back in line with scientific advice. Instead, it is more likely that catches are misreported (Simmonds 2007; Zeller *et al.* 2011), TACs are too high for the stocks present (either scientifically or politically) resulting in apparent underutilisation (Karagiannakos 1996), or catches may be constrained by TACs for other targeted fisheries or by TACs for bycatch species (Andersen *et al.* 2009).

Summary

In summary, while Cook *et al.* (in review) highlight some inaccuracies in our paper (i.e. typographical errors in the equations, the application of stochasticity) their main criticisms of our paper are invalid having misinterpreted both the conclusions that we made and the aim of our model. The aim of our model was to explore the implications of politically-motivated fisheries decision making, and its conclusion was that this amount to systematic fisheries mismanagement. Cook *et al.* (in review) claim that we "blame managers and politicians for mis-management based on a poorly constructed model". Actually, we blame managers and politicians for mismanagement on the basis of our analysis of historical data on decisions. Of this historical data analysis Cook *et al.* (in review) make no mention except to show that often the actual (reported) landings are below the agreed TAC. While this may be the case in some areas, this is unlikely to mean that fishing mortality (direct or indirect, legal or illegal) falls below those TACs. Our model does not predict the collapse of stocks within the next 40 years as Cook *et al.* (in review) interpreted. Instead it was designed to illustrate that systematically exceeding scientific advice regarding total allowable catches may have serious consequences for the status of stocks. The reanalyses with the modified model show here indicate that our main conclusions are robust: past decision-making by fisheries ministers' has contributed considerably to the poor state of many fish stocks within European waters and that unless politicians are legally bound to follow scientific recommendations after the 2012 reform of the Common Fisheries Policy, the reform is unlikely to produce sustainable and productive fisheries in the future.

Appendix

The model described within O'Leary et al (2011) 'Fisheries Mismanagement' and this paper is summarised here. Equations A.1 to A.9 provide the structure of the original model corrected for typographical errors. Equations A.1 to A.8 and A.10 describe the model presented within this manuscript for cod and herring.

Fish population dynamics are described in terms of biomass. In its discrete form this model can be expressed as:

$$B_{t+1} = sB_t + R_t - C_t \tag{A.1}$$

Where s describes the change in biomass (B) of the stock from one year (t) to the next ($t+1$) as a result of survivorship in the face of natural mortality only; R represents the recruitment to the population in year t and C is the catch taken from the stock in year t.

Recruitment (R_t) is derived from a re-parameterised version of the Beverton-Holt (1957) stock recruitment relationship which includes a steepness parameter (z) (Mace and Doonan, 1988) to represent recruitment, relative to the recruitment at equilibrium in the absence of fishing, that occurs when spawner abundance has been reduced to 20% of its virgin level (B_0). The reparameterisation of Beverton-Holt to include a z value defines the parameters α and β (necessary to calculate recruitment as:

$$\alpha = \frac{B_0}{R_0}\left(1 - \left(\frac{z-0.2}{0.8z}\right)\right) \tag{A.2}$$

$$\beta = \frac{z-0.2}{0.8zR_0} \tag{A.3}$$

Recruitment can thus be represented deterministically as:

$$R_t = \left(\frac{B_{t-L}}{\alpha+\beta B_{t-L}}\right) \tag{A.4}$$

where $R_0 = B_0(1-s)$. (A.5)

Here, L refers to the time lag in years between birth and recruitment to the fishery. Recruitment in year t therefore depends on the stock biomass L years earlier. Consequently, two distinct age groups are modelled; recruits, i.e. fish aged less than L, and fish older than L years which are fecund, i.e. capable of producing new biomass.

Within this model the biomass at MSY (B_{MSY}), and the MSY itself, are defined as:

$$B_{MSY} = \frac{1}{\beta}\left(\sqrt{\frac{\alpha}{(1-s)}} - \alpha\right) \tag{A.6}$$

$$MSY = B_{MSY}\left(s - 1 + \frac{1}{\alpha+\beta B_{MSY}}\right) \tag{A.7}$$

The TAC was calculated by equation A.8 where B_{LIM} is defined as B_{MSY} and $0.1B_0$ represents the 'collapse' threshold.

$$TAC = \left(\frac{B_t - 0.1B_0 \times MSY}{B_{LIM} - 0.1B_0}\right) \tag{A.8}$$

In our original model, stochasticity (σ) was introduced into the system via the recruitment equation for each year of a simulation run. Recruitment was calculated using the deterministic equation which was then multiplied by a value drawn randomly from a uniform distribution spanning –1 to 1. Juvenile bycatch was then introduced into the model by adjusting recruitment during each year of a simulation run (equation A.9).

$$R_t = \left(\left(\frac{B_{t-L}}{\alpha + \beta B_{t-L}} \right) . (1 + \sigma) \right) j \tag{A.9}$$

For the model presented within this manuscript, recruitment variability was assumed to follow a lognormal distribution and applied through the multiplication of a randomly drawn number (ln(r)). Unaccounted juvenile mortality (u) (previously referred to as juvenile bycatch) was introduced by adjusting recruitment accordingly.

$$R_t = \left(\left(\frac{B_{t-L}}{\alpha + \beta B_{t-L}} \right) . ln(r) \right) u \tag{A.10}$$

References

Andersen, J.L., Nielsen, M., Lindebo, E., 2009. Economic gains of liberalising access to fishing quotas within the European Union. Mar. Policy. 33, 497-503.

Catchpole, T.L., Frid, C.L.J., Gray, T.S., 2005. Discards in North Sea fisheries: causes, consequences and solutions. Mar. Policy 29, 421-430.

Cook, R.M., Needle, C.L., Fernandes, P.G. (in review). Comment on "Fisheries Mismanagement". Mar. Pollut. Bull.

Froese, R., Branch, T.A., Proelß, A., Quaas, M., Sainsbury, K., Zimmermann, C., 2011. Generic harvest control rules for European fisheries. Fish Fish. 12, 340-351.

Gillis, D.M., Pikitch, E.K., Peterman, R.M., 1995. Dynamic discarding decisions: foraging theory for high-grading in a trawl fishery. Behav. Ecol. 6, 146-154.

Hsieh, C.-h., Reiss, C.S., Hunter, J.R., Beddington, J.R., May, R.M., Sugihara, G., 2006. Fishing elevates variability in the abundance of exploited species. Nature. 443, 859-962.

Karagiannakos, A., 1996. Total Allowable Catch (TAC) and quota management system in the European Union. Mar. Pol. 20, 235-248.

Law, R., Plank, M.J., Kolding, J., 2012. On balanced exploitation of marine ecosystems: results from dynamic size spectra. ICES J. Mar. Sci. 69, 602-614.

Mace, P.M., Doonan, I., 1988. A generalised bioeconomic simulation model for fish dynamics. New Zealand Fishery Assessment Research Document no. 99/4. Wellington, New Zealand.

Myers, R.A., Bowman, K.G., Barrowman, N.J., 1999. Maximum reproductive rate of fish at low population sizes. Can. J. Fish. Aquat. Sci. 56, 2404-2419.

Nash, R.D.M., Pilling, G.M., Kell, L.T., Schön, P.-J., Kjesbu, O.S., 2010. Investment in maturity-at-age and -length in northeast Atlantic cod stocks. Fish. Res. 104, 89-99.

O'Leary, B.C., Smart, J.C.R., Neale, F.C., Hawkins, J.P., Newman, S., Milman, A.C., Roberts, C.M., 2011. Fisheries Mismanagement. Mar. Pollut. Bull. 62, 2642-2648.

Pauly, D., 1980. On the interrelationships between natural mortality, growth parameters, and mean environmental temperature in 175 fish stocks. ICES J. Mar. Sci. 39, 175-192.

Simmonds, E.J., 2007. Comparison of two periods of North Sea herring stock management: success, failure and monetary value. ICES J. Mar. Sci. 64, 686-692.

Worm, B., Barbier, E.B., Beaumont, N., Duffy, E., Folke, C., Halpern, B.S., Jackson, J.B.C., Lotze, H.K., Micheli, F., Palumbi, S.R., Sala, E., Selkoe, K.A., Stachowicz, J.J., Watson, R., 2006. Impacts of biodiversity loss on ocean ecosystem services. Science. 314, 787-790.

Zeller, D., Rossing, P., Harper, S., Persson, L., Booth, S., Pauly, D., 2011. The Baltic Sea: Estimates of total fisheries removals 1950-2007. Fish. Res. 108, 356-363.

Acronyms and Abbreviations

ABNJ	Areas Beyond National Jurisdiction
ACOM	Advisory Committee (ICES)
B_{LIM}	Biomass limit reference point
B_{MSY}	Biomass corresponding to MSY
B_{pa}	Biomass precautionary approach reference point
CBD	Convention on Biological Diversity
CFP	Common Fisheries Policy
CGFZ	Charlie Gibbs Fracture Zone
C_{MAX}	Maximum possible catch
C_{MIN}	Minimum possible catch
CPUE	Catch per unit effort
EBFM	Ecosystem-Based Fishery Management
EBSA	Ecologically Biologically Significant Area
EEZ	Exclusive Economic Zone
EU	European Union
FAO	Food and Agriculture Organisation
HCR	Harvest Control Rule
HELCOM	Helsinki Commission (Baltic marine environment protection commission)
ICCAT	International Commission for the Conservation of Atlantic Tunas
ICES	International Council for the Exploration of the Sea
IMO	International Maritime Organisation
ISA	International Seabed Authority
ITQ	Individual Transferable Quota
IUU	Illegal, Unregulated and Unreported Fishing
MAST	Multi-stock Age-Structured Tag-integrated stock assessment model
MPA	Marine Protected Area
MSY	Maximum Sustainable Yield
NAFO	Northwest Atlantic Fisheries Organisation
NAMMCO	North Atlantic Marine Mammal Commission
NASCO	North Atlantic Salmon Conservation Organisation
NEAFC	North-East Atlantic Fisheries Commission

OSPAR	Oslo-Paris Convention. (Conservation body responsible for the marine environment of the North-East Atlantic)
PAI	Political Adjustment Index
RFMO	Regional Fisheries Management Organisation
SCRS	Standing Committee on Research and Statistics
SSB	Spawning Stock Biomass
STECF	Scientific, Technical and Economic Committee on Fisheries
TAC	Total Allowable Catch
VME	Vulnerable Marine Ecosystem
VPA	Virtual Population Analysis
WSSD	World Summit on Sustainable Development
WWF	Worldwide Fund for Nature
UNCLCS	United Nations Commission on the Limits of the Continental Shelf
UNCLOS	United Nations Convention on the Law of the Sea
UNGA	United Nations General Assembly

Bibliography

Albuquerque, A.T.M.d., 2010. Presentation on the workload of the Commission on the Limits of the Continental Shelf (CLCS). 20th Meeting of States Parties, 14-18 June, 2010.

Alverson, D.L., Freeberg, M.H., Murawski, S.A., Pope, J.G., 1994. A global assessment of fisheries bycatch and discards. FAO, 233p. http://www.fao.org/DOCREP/003/T4890E/T4890E00.HTM

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

Anderson, J.L., Nielsen, M., Lindebo, E., 2009. Economic gains of liberalising access to fishing quotas within the European Union. Mar. Policy. 33, 497-503.

Anon., 2011. EC proposes science-based, sustainable fishing opportunities for 2011. http://europa.eu/rapid/pressReleasesAction.do?reference=IP/10/1501&format=HTML&aged=0&language=EN&guiLanguage=en

Apostolaki, P., Babcock, E., McAllister, M.K., 2003. A scenario based framework for the stock assessment of North Atlantic bluefin tuna taking into account trans-Atlantic movement, stock mixing and multiple fleets. ICCAT Col. Vol. Sci. 55, 1055-1079.

Apostolaki, P., Babcock, E., McAllister, M., 2004. Further investigation of the effects of stock mixing on estimates of the size of North Atlantic bluefin tuna population using the six-area population dynamics model presented in SCRS/2002/088. ICCAT Col. Vol. Sci. 56, 1121-1133.

Apostolaki, P., Hillary, R., 2009. Harvest control rules in the context of fishery-independent management of fish stocks. Aquat. Living Resour. 22, 217-224.

Apostolaki, P., McAllister, M., Babcock, E., 2005. Preliminary evaluation of the effects of North Atlantic bluefin tuna mixing: Summary of main findings. ICCAT Col. Vol. Sci. 58, 1380-1387.

Aps, R., Kell, L.T., Lassen, H., Liiv, I., 2007. Negotiation framework for Baltic fisheries management: Striking the balance of interest. ICES J. Mar. Sci. 64, 858-861.

Ardron, J., Gjerde, K., Pullen, S., Tilot, V., 2008. Marine spatial planning in the high seas. Mar. Policy. 32, 832-839.

Ardron, J.A., 2008. Three initial OSPAR tests of ecological coherence: Heuristics in a data-limited situation. ICES J. Mar. Sci. 65, 1527-1533.

Armstrong, C.W., 2007. A note on the ecological-economic modelling of marine reserves in fisheries. Ecol. Econ. 62, 242-250.

Armsworth, P.R., Block, B.A., Eagle, J., Roughgarden, J.E., 2010. The economic efficiency of a time-area closure to protect spawning bluefin tuna. J. Appl. Ecol. 47, 1-11.

Arnason, R., Kelleher, K., Willmann, R., 2009. The sunken billions: The economic justification for fisheries reform. Joint publication of the Word Bank and the FAO., 88p. http://siteresources.worldbank.org/EXTARD/Resources/336681-1224775570533/SunkenBillionsFinal.pdf

Auster, P.J., Gjerde, K., Heupel, E., Watling, L., Grehan, A., Rogers, A.D., 2011. Definition and detection of vulnerable marine ecosystems on the high seas: Problems with the "move-on" rule. ICES J. Mar. Sci. 68, 254-264.

Bailey, D.M., Collins, M.A., Gordon, J.D.M., Zuur, A.F., Priede, I.G., 2009. Long-term changes in deep-water fish populations in the northeast Atlantic: A deeper reaching effect of fisheries? Proc. R. Soc. B. 276, 1965-1969.

Ban, N.C., Hansen, G.J.A., Jones, M., Vincent, A.C.J., 2009. Systematic marine conservation planning in data-poor regions: Socioeconomic data is essential. Mar. Policy. 33, 794-800.

Barrowman, N.J., Myers, R.A., 2000. Still more spawner-recruitment curves: The hockey stock and its generalisations. Can. J. Fish. Aquat. Sci. 57, 665-676.

Baudron, A., Ulrich, C., Nielsen, J.R., Boje, J., 2010. Comparative evaluation of a mixed-fisheries effort-management system based on the Faroe Islands example. ICES J. Mar. Sci. 67, 1036-1050.

Beddington, J.R., Agnew, D.J., Clark, C.W., 2007. Current problems in the management of marine fisheries. Science. 316, 1713-1716.

Bellido, J.M., Santos, M.B., Pennino, M.G., Valeiras, X., Pierce, G.J., 2011. Fishery discards and bycatch: solutions for an ecosystem approach to fisheries management. Hydrobiologia. 670, 317-333.

Benn, A.R., Weaver, P.P., Billet, D.S.M., van den Hove, S., Murdock, A.P., Doneghan, G.B., Le Bas, T., 2010. Human activities on the deep seafloor in the North East Atlantic: An assessment of spatial extent. PLoS ONE. 5, e12730.

Berkeley, S.A., Chapman, C., Sogard, S.M., 2004. Maternal age as a determinant of larval growth and survival in a marine fish, *Sebastes melanops*. Ecology. 85, 1258-1264.

Bess, R., Rallapudi, R., 2007. Spatial conflicts in New Zealand fisheries: The rights of fishers and protection of the marine environment. Mar. Policy 31, 719-729.

Beverton, R.J.H., Holt, S.J., 1957. On the dynamics of exploited fish populations. London, Chapman and Hall.

Biju Kumar, A., Deepthi, G.R., 2006. Trawling and by-catch: Implications on marine ecosystem. Curr. Sci. 90, 922-931.

Birkeland, C., Dayton, P.K., 2005. The importance in fishery management of leaving the big ones. Trends. Ecol Evol. 20, 356-358.

Block, B.A., Dewar, H., Blackwell, S.B., Williams, T.D., Prince, E.D., Farwell, C.J., Boustany, A., Teo, S.L.H., Seitz, A., Walli, A., Fudge, D., 2001. Migratory movements, depth preferences, and thermal biology of Atlantic bluefin tuna. Science. 293, 1310-1314.

Block, B.A., Teo, S.L.H., Walli, A., Boustany, A.M., Stokesbury, M.J.W., Farwell, C.J., Weng, K.C., Dewar, H., Williams, T.D., 2005. Electronic tagging and population structure of Atlantic bluefin tuna. Nature. 434, 1121-1127.

Bohnsack, J.A., 1996. Maintenance and recovery of reef fishery productivity, in: Polunin, N.V.C., Roberts, C.M. (Eds.), Reef fisheries. London, Chapman & Hall**:** 283-313.

Boustany, A.M., Reeb, C.A., Block, B.A., 2008. Mitochondrial DNA and electronic tracking reveal population structure of Atlantic bluefin tuna (*Thunnus thynnus*). Mar. Biol. 156, 13-24.

Brander, K., 2010. Impacts of climate change on fisheries. J. Marine Syst. 79, 389-402.

Brewin, P.E., Stocks, K.I., Menezes, G., 2007. A history of seamount research, in: Pitcher, T.J., Morato, T., Hart, P.J.B., Clark, M.R., Haggan, N., Santos, R.S. (Eds.), Seamounts: Ecology, conservation and management. Oxford, Blackwell.

Brito, D., 2010. Overcoming the Linnean shortfall: Data deficiency and biological survey priorities. Basic. Appl. Ecol. 11, 709-713.

Brooks, T.A., Mittermeier, R.A., Mittermeier, C.G., Da Fonseca, G.A.B., Rylands, A.B., Konstant, W.R., Flick, P., Pilgrim, J., Oldfield, S., Magin, G., Hilton-Taylor, C.H., 2002. Habitat loss and extinction in the hotspots of biodiversity. Conserv. Biol. 16, 909-923.

Browman, H.I., Stergiou, K.I., 2004. Perspectives on ecosystem-based approaches to the management of marine resources. Mar. Ecol. Prog. Ser. 274, 269-270.

Bundy, A., Fanning, P., Zwanenburg, K.C.T., 2005. Balancing exploitation and conservation of the eastern Scotian Shelf ecosystem: Application of a 4D ecosystem exploitation index. ICES. J. Mar. Sci. 62, 504-510.

Butterworth, D.S., Ianelli, J.N., Hilborn, R., 2003. A statistical model for stock assessment of southern bluefin tuna with temporal changes in selectivity. Afr. J. Mar. Sci. 25, 331-361.

Cacaud, P., 2005. Fisheries laws and regulations in the Mediterranean: A comparative study. Studies and Revies. General Fisheries Commission for the Mediterranean. No 75. Rome, FAO**:** 40p.

Caddy, J.F., Garibaldi, L., 2000. Apparent changes in the trophic composition of world marine harvests: the perspective from the FAO capture database. Ocean Coast. Manage. 43, 615-655.

Caddy, J.F., Mahon, R., 1995. Reference points for fisheries management. FAO. ftp://ftp.fao.org/docrep/fao/003/v8400e/v8400e.pdf

Cadigan, N.G. 2009. Sensitivity of common estimators of management parameters derived from stock-recruit relationships. Fish. Res. 96, 195-205.

Cardinale, M., Svedäng, H., 2008. Mismanagement of fisheries: Policy or science? Fish. Res. 93, 244-247.

Carlsson, J., McDowell, J.R., Carlsson, J.E.L., Graves, J.E., 2007. Genetic identity of YOY bluefin tuna from the eastern and western Atlantic spawning areas. J. Hered. 98, 23-28.

CBD, 1995. Decision II/10 - Conservation and sustainable use of marine and coastal biological diversity, also known as the Jakarta Mandate. Proceedings on the World Summit on Sustainable Development.

CBD, 2008. Options for preventing and mitigating the impact of some activities on selected seabed habitats. Subsidiary Body on Scientific, Technical and

Technological Advice Thirteenth meeting. FAO, Rome, 18-22 February 2008.

CEC, 2001. Green paper on the future of the common fisheries policy. Commission of the European Communities. Brussels, COM (2001) 135.

CEC, 2009. Green Paper on the Reform of the Common Fisheries Policy. Commission of the European Communities. Brussels, COM(2009)163**:** 28p.

Chen, Y., Chen, L., Stergion, K.I., 2003. Impacts of data quantity on fisheries stock assessment. Aquat. Sci. 65, 92-98.

Cheung, W.W.L, Pitcher, T.J., Pauly, D., 2005. A fuzzy logic expert system to estimate intrinsic extinction vulnerabilities of marine fishes to fishing. Biol. Cons. 124, 97-111.

Cheung, W.W.L., Watson, R., Morato, T., Pitcher, T.J., Pauly, D., 2007. Intrinsic vulnerability in the global fish catch. Mar. Ecol. Prog. Ser. 333, 1-12.

Christiansen, S., 2006. Marine protected areas in areas beyond national jurisdiction. Proposed high seas MPAs in the north-east Atlantic by WWF 1998-2006. WWF North-East Atlantic Program. 38pp. http://www.ngo.grida.no/wwfneap/Projects/Reports/WWF_NEA_HSMPA_Proposals.pdf

Clark, C.W., 1973a. The economics of overexploitation. Science 40, 2080-2091.

Clark, C.W., 1973b. Profit maximisation and the extinction of animal species. J. Polit. Econ. 81, 950-961.

Clark, C.W., 1990. Mathematical bioeconomics : the optimal management of renewable resources. New York; Chichester, Wiley.

Condé, S., Jones-Walters, L., Torre-Marin, A., Romão, C., 2010. EU 2010 Biodiversity Baseline. No 12/2010:126p. http://www.eea.europa.eu/publications/eu-2010-biodiversity-baseline/

Conover, D.O., Munch, S.N., Arnott, S.A., 2009. Reversal of evolutionary downsizing caused by selective harvest of large fish. Proc. R. Soc. B. 276, 2015-2020.

Cook, R.M., Needle, C.L. and Fernandes, P.G. (in review). Comment on Fisheries Mismanagement. Mar. Pollut. Bull..

Corten, A., 1996. The widening gap between fisheries biology and fisheries management in the European Union. Fish. Res. 27, 1-15.

Costello, C., Rassweiler, A., Siegel, D., De Leo, G., Micheli, F., Rosenberg, A., 2010. The value of spatial information in MPA network design. PNAS 107, 18297-18299.

Cotter, J., 2010. Reforming the European Common Fisheries Policy: A case for capacity and effort controls. Fish Fish. 11, 210-219.

Cropper, M., Laibson, D., 1999. The implications of hyperbolic discounting for project evaluation, in: Portney, P.R., Weyant, J.P. (Eds.) Discounting and intergenerational equity. Resources for the future.

Cullis-Suzuki, S., Pauly, D., 2010. Failing the high seas: A global evaluation of regional fisheries management organisations. Mar. Pol. 34, 1036-1042.

Cury, P.M., Shin, Y.-J., Planque, B., Durant, J., Fromentin, J.-M., Kramer-Schadt, S., Stenseth, N.C., Travers, M., Grimm, V., 2008. Ecosystem oceanograpy for global change in fisheries. Trends Ecol. Evol. 23, 338-346.

Cushing, D.H., 1971. The dependence of recruitment on parent stock in different groups of fishes. ICES J. Mar. Sci. 33, 340-362.

Czybulka, D., Kersandt, P., 2000. Legal regulations, legal instruments and competent authorities with relevance for marine protected areas (MPAs) in the exclusive economic zone (EEZ) and the high seas of the OSPAR maritime area. German Federal Agency for Nature Conservation, BfN Skripten 22, Bonn.**:** 84pp.

Daan, N., 1997. TAC management in North Sea flatfish fisheries. J. Sea Res. 37, 321-341.

Damanaki, M., 2010. Taking science as our starting point. State of European Fish Stocks in 2010, September 2010. Borchette Centre, Brussels, 14 September 2010. http://europa.eu/rapid/pressReleasesAction.do?reference=SPEECH/10/431&format=HTML&aged=0&language=EN&guiLanguage=en

Daskalov, G.M., 2008. Overfishing affects more than fish populations: Trophic cascades and regime shifts in the Black Sea, in: Payne, A., Cotter, J.R., Potter, T. (Eds.) Advances in fisheries science: 50 years on from Beverton and Holt. Oxford, Blackwell Publishing.

Datta, S., Delius, G.W., Law, R., 2010. A jump-growth model for predator-prey dynamics: derivation and application to marine ecosystems. Bull. Math. Biol. 72, 1361-1382.

Daw, T., Gray, T., 2005. Fisheries science and sustainability in international policy: a study of failure in the European Union's Common Fisheries Policy. Mar. Policy. 29, 189-197.

De Stefano, V., Van der Heijden, P.G.M., 2007. Bluefin tuna fishing and ranching: A difficult management problem. New Medit. 2, 59-64.

Del Valle, I., Astorkiza, K., 2007. Institutional designs to face the dark side of total allowable catches. ICES J. Mar. Sci. 64, 851-857.

Deriso, R.B., 1980. Harvesting strategies and parameter estimation for an age-structured model. Can. J. Fish. Aquat. Sci. 37, 268-282.

Diamond, B., Beukers-Stewart, B.D., 2011. Fisheries discards in the North Sea: Waste of resources or a necessary evil? Res. Fish. Sci. 19, 231-245.

Doney, S.C., Fabry, V.J., Freely, R.A., Kleypas, J.A., 2009. Ocean acidification: the other CO_2 problem. Ann. Rev. Mar. Sci. 1, 169-192.

Druon, J.-N., 2010. Habitat mapping of the Atlantic bluefin tuna derived from satellite data: It's potential as a tool for the sustainable management of pelagic fisheries. Mar. Policy. 34, 293-297.

Dulvy, N.K., Sadovy, Y., Reynolds, J.D., 2003. Extinction vulnerability in marine populations. Fish Fish. 4, 25-64.

Duncan, S., Hepburn, C., Papachristodouloum, A., 2011. Optimal harvesting of fish stocks under a time-varying discount rate. J. Theor. Biol. 269, 166-173.

Eliasen, S., Sverdrup-Jensen, S., Holm, P., Johnsen, J.P., 2009. Nordic experience of fisheries management: Seen in relation to the reform of the EU Common Fisheries Policy. Nordic Council of Ministers. Copenhagen, TemaNord.

EU, 2010a. Agreed record of conclusions of fisheries consultations between the European Union and Norway for 2010. Brussels.

EU, 2010b. COUNCIL REGULATION (EU) No 23/2010 of 14 January 2010 fixing for 2010 the fishing opportunities for certain fish stocks and groups of fish stocks, applicable in EU waters and, for EU vessels, in waters where catch limitations are required and amending Regulations (EC) No 1359/2008, (EC) No 754/2009, (EC) No 1226/2009 and (EC) No 1287/2009. O. J.

FAO, 1996. Precautionary approach to fisheries. Part 2: Scientific papers. Prepared for the Technical Consultation on the Precautionary Approach to Capture Fisheries (Including Species Introductions). Lysekil, Sweden, 6–13 June 1995. (A scientific meeting organized by the Government of Sweden in cooperation with FAO). FAO Fisheries Technical Paper. No. 350, Part 2. Rome, FAO. 1996**:** 210p.

FAO, 2004. Management of tuna fishing capacity: conservation and socio-economics. FAO. Rome.

FAO., 2010a. "FAOSTAT Food Balance Sheets." from http://faostat.fao.org/site/354/default.aspx.

FAO, 2010b. The state of world fisheries and aquaculture, 2010. FAO Fisheries and Aquaculture Department. Rome.

Field, J.G., Moloney, C.L., du Buisson, L., Jarre, A., Stroemme, T., Lipinski, M.R., Kainge, P., 2008. Exploring the BOFFFF Hypothesis Using a Model of Southern African Deepwater Hake (*Merluccius paradoxus*), in: Tsukamoto, K.K., Takeuchi, T., Beard Jr., T.D., Kaiser, M.J. (Eds.) Fisheries for Global Welfare and Environment, 5th World Fisheries Congress. Memorial book of the 5th World Fisheries Congress 2008, TERRAPUB**:** 17-26.

Fleming, C.M., Alexander, R.R. 2003. Single-species versus multiple-species models: The economic implications. Ecol. Model. 170, 203-211.

Fonteneau, A., 2009. Atlantic bluefin tuna: 100 centuries of fluctuating fisheries." SCRS/2008/058. ICCAT Col. Vol. Sci. 63, 51-68.

Franchino, F., Rahming, A.J., 2003. Biased ministers, inefficiency and control in distributive policies: An application to the EU fisheries policy. EUP 4, 11-36.

Frank, V., 2007. The European Community and Marine Environmental Protection in the International Law of the Sea: Implementing Global Obligations at the Regional Level. Publications on Ocean Development. Brill, 504p.

Froese, R., 2011. Fishery reform slips through the net. Nature. 475, 7.

Froese, R., Kesner-Reyes, K., 2002. Impact of fishing on the abundance of marine species. ICES CM 2002/L:12.

Froese, R., Branch, T.A., Proelß, A., Quaas, M., Sainsbury, K., Zimmermann, C., 2011. Generic harvest control rules for European fisheries. Fish Fish. 12, 340-351.

Froese, R., Proelß, A., 2010. Rebuilding fish stocks no later than 2015: Will Europe meet the deadline? Fish Fish. 11, 194-202.

Froese, R., Stern-Pirlot, A., Kesner-Reyes, K., 2009. Out of new stocks in 2020: A comment on "Not all fisheries will be collapsed in 2048". Mar. Policy. 33, 180-181.

Fromentin, J.-M., 2006. Size limits regulation for tuna: Should we also consider the protection of large fish? ICCAT Col. Vol. Sci. 59, 590-592.

Fromentin, J.-M., 2009. The key importance of the underlying stock-recruitment assumption when evaluating the potential management regulations of Atlantic bluefin tuna. ICCAT Col. Vol. Sci. 64, 558-567.

Fromentin, J.-M., Kell, L.T., 2007. Consequences of variations in carrying capacity or migration for the perception of Atlantic bluefin tuna population dynamics. Can. J. Fish. Aquat. Sci. 67, 627-836.

Fromentin, J.-M., Ravier, C., 2005. The east Atlantic and Mediterranean bluefin tuna stock: Looking for sustainability in a context of large uncertainties and strong political pressures. Bull. Mar. Sci. 76, 353-361.

Gaines, S.D., White, C., Carr, M.H., Palumbi, S.R., 2010. Designing marine reserve networks for both conservation and fisheries management. PNAS 107, 18286-18293.

García, S.M., Zerbi, A., Aliaume, C., Do Chi, T., Laserre, G., 2003. The ecosystem approach to fisheries. issues, terminology, principles, institutional foundations, implementation and outlook. FAO Fisheries Technical Paper. no.443, Rome, FAO.2003.

Grafton, R.Q., Kompas, T., 2005. Uncertainty and the active adaptive management of marine reserves. Mar. Policy. 29, 471-479.

Grafton, R.Q., Kompas, T., Lindenmayer, D.B., 2005. Marine reserves with ecological uncertainty. Bull. Math. Biol. 67, 957-971.

Gray, N.J., 2010. Sea change: Exploring the international effort to promote marine protected areas. Conserv. Soc. 8, 331-338.

Gray, T., Hatchard, J., 2003. The 2002 reform of the Common Fisheries Policy's system of governance – Rhetoric or reality? . Mar. Policy. 27, 545-554.

Gray, T.S., 2005. Participatory fisheries governance – three central themes, in Gray, T.S., (Ed.), Participation in fisheries governance. Springer, Dordrecht, Netherlands**:** 343-356.

Greenpeace, 2007. Taking tuna out of the can: Rescue plan for the world's favourite fish, Greenpeace International.

Grift, R.E., Rijnsdorp, A.D., Barot, S., Heino, M., Dieckmann, U. 2003. Fisheries induced trends in reaction norms for maturation in North Sea plaice. Mar. Ecol. Prog. Ser. 257, 247-257.

Halpern, B.S., Walbridge, S., Selkoe, K.A., Kappel, C.V., Micheli, F., D'Agrosa, C., Bruno, J.F., Casey, K.S., Ebert, C., Fox, H.E., Fujita, R., Heinemann, D., Lenihan, H.S., Madin, E.M.P., Perry, M.T., Selig, E.R., Spalding, M., Steneck, R.S., Watson, R., 2008. A global map of human impact on marine ecosystems. Science. 319: 948-952.

Halpern, B.S., 2003. The impact of marine reserves: Do reserves work and does reserve size matter? Ecol. Appl. 13, 117-137.

Haltuch, M.A., Punt, A.E., Dorn, M.W., 2008. Evaluating alternative estimators of fishery management reference points. Fish. Res. 94, 290-303.

Hamon, K., Ulrich, C., Hoff, A., Kell, L.T., 2007. Evaluation of management strategies for the mixed North Sea roundfish fisheries with the FLR framework, in: Oxley, L., Kulasiri, D. (Eds.), MODSIM 2007 International Congress on Modelling and Simulation. Modelling and Simulation Society of Australia and New Zealand, December 2007.

Hannesson, R., 1998. Marine Reserves: What would they accomplish? Mar. Resour. Econ. 13, 159-170.

Hardin, G., 1968. The tragedy of the commons. Science. 162, 1243-1248.

Harrington, J.M., Myers, R.A., Rosenberg, A.A., 2005. Wasted fishery resources: discarded by-catch in the USA. Fish Fish. 6, 350-361.

Harris, J., Haward, M., Jabour, J., Woehler, E.J., 2007. A new approach to selecting marine protected areas (MPAs) in the Southern Ocean. Antarct. Sci. 19, 189-194.

Hauge, K.H., 2011. Uncertainty and hyper-precision in fisheries science and policy." Futures 43, 173-181.

Heal, G., 1997. Discounting and climate change. Clim. Change. 37, 335-343.

Heller, N.E., Zavaleta, E.S., 2009. Biodiversity management in the face of climate change: A review of 22 years of recommendations. Biol. Conserv. 142, 14-32.

Hiddink, J.G., Jennings, S., Kaiser, M.J., Queirós, A.M., Duplisea, D.E., Piet, G.J., 2006. Cumulative impacts of seabed trawl disturbance on benthic biomass, production, and species richness in different habitats. Can. J. Fish. Aquat. Sci. 63, 721-736.

Hilborn, R., 2004. Ecosystem-based fisheries management: the carrot or the stick? Mar. Ecol. Prog. Ser. 274, 275-278.

Hilborn, R., Mangel, M., 1997. Management of hake fisheries in Namibia, in: The ecological detective: Confronting models with data. New Jersey, Princeton University Press, 235-262.

Hilborn, R., Walters, C.J., 1992. Quantitative fisheries stock assessment: Choice, dynamics, and uncertainty. New York, Chapman & Hall.

Hoegh-Guldberg, O., Bruno, J.F., 2011. The impact of climate change on the world's marine ecosystems. Science. 328, 1523-1528.

Hoel, A.H., Kvalvik, I., 2006. The allocation of scarce natural resources: The case of fisheries. Mar. Policy. 30, 347-356.

Holden, M.J., 1994. The Common Fisheries Policy, Origin, Evaluation and Future. Oxford, Fishing News Books.

Holland, D.S., 2003. Integrating spatial management measures into traditional fishery management systems: the case of the Georges Bank multispecies groundfish fishery. ICES J. Mar. Sci. 60, 915-929.

Hollowed, A.B., Bax, N., Beamish, R., Collie, J., Fogarty, M., Livingston, P., Pope, J., Rice, J.C., 2000. Are multispecies models an improvement on single-species models for measuring fishing impacts on marine ecosystems? ICES J. Mar. Sci. 57, 707-719.

Holt, S., 2009. Sunken billions - But how many? Fish. Res. 97, 3-10.

Hooper, D.U., Chapin, F.S., Ewel, J.J., Hector, A., Inchausti, P., Lavorel, S., Lawton, J.H., Lodge, D.M., Loreau, M., Naeem, S., Schmid, B., Seta, H., Symstad, A.J., Vandermeer, J., Wardle, D.A., 2005. Effects of biodiversity on ecosystem functioning: A consensus of current knowledge. Ecol. Monogr. 75, 3-35.

Howell, K.L., 2010. A benthic classification system to aid in the implementation of marine protected area networks in the deep/high seas of the NE Atlantic. Biol. Conserv. 143, 1041-1056.

Hsieh, C-h., Reiss, C.S., Hunter, J.R., Beddington, J.R., May, R.M., Sugihara, G., 2006. Fishing elevates variability in the abundance of exploited species. Nature. 443, 859-962.

Hutchings, J.A., Reynolds, J.D., 2004. Marine fish population collapses: Consequences for recovery and extinction risk. Bioscience. 54, 297-309.

ICCAT, 1998. Recommendation by ICCAT to Establish a Rebuilding Program for Western Atlantic Bluefin Tuna [98-07].

ICCAT, 2006. Recommendation by ICCAT to Establish a Multi-Annual Recovery Plan for Bluefin Tuna in the Eastern Atlantic and Mediterranean [REC 06-05].

ICCAT, 2008. Report of the 2008 Atlantic Bluefin Tuna Stock Assessment Session.

ICCAT, 2010a. Report for Biennial Period 2008-2009, Part II**:** 116-136.

ICCAT, 2010b. Report of the 2010 Atlantic bluefin tuna stock assessment session. Madrid, Spain - September 6 to 12, 2010.

ICES, 2008. Report of the Study Group on Management Strategies (SGMAS), 17-21 November 2008. Lisbon, Portugal. ICES Document CM 2005/ACFM:09**,** 74pp.

ICES, 2009. Cod in Subarea IV (North Sea), Division VIId (Eastern Channel), and IIIa West Skagerrak. http://www.ices.dk/products/icesadvice/2009/ICES%20ADVICE%202009%20Book%206.pdf

Iles, T.C., 1994. A review of stock-recruitment relationships with reference to flatfish populations. Neth. J. Sea Res. 32, 399-420.

IPCC, 2007. Climate Change 2007: Synthesis Report. http://www.ipcc.ch/pdf/assessment-report/ar4/syr/ar4_syr.pdf

ISA, 2010. Non-living resources of the continental shelf beyond 200 nautical miles: Speculations on the implementation of Article 82 of the United Nations Convention on the Law of the Sea. Technical Study No. 5. International Seabed Authority, 69pp.

Issenberg, S., 2007. The sushi economy: Globalisation and the making of a modern delicacy. London, Penguin Books Ltd.

Jackson, J.B.C., Kirby, M.X., Berger, W.H., Bjorndal, K.A., Botsford, L.W., Bourque, B.J., Bradbury, R.H., Cooke, R., Erlandson, J., Estes, J.A., Hughes, T.P., Kidwell, S., Lange, C.B., Lenihan, H.S., Pandolfi, J.M., Peterson, C.H., Steneck, R.S., Tegner, M.J., Warner, R.R.. 2001. Historical Overfishing and the Recent Collapse of Coastal Ecosystems. Science. 293, 629-637.

Jakobsson, J., Stefánsson, G., 1999. Management of summer-spawning herring off Iceland. ICES J. Mar. Sci. 56, 827-833.

Jensen, O.P., Branch, T.A., Hilborn, R. (2012). Marine fisheries as ecological experiments. Theor. Ecol. 5, 3-22.

Johnsen, J.P., Eliasen, S., 2011. Solving complex fisheries management problems: What the EU can learn from the Nordic experiences of reduction of discards. Mar. Policy. 35, 130-139.

Kaiser, M.J., Attrill, M.J., Jennings, S., Thomas, D.N., Barnes, K.A., Brierley, A.S., Polunin, N.V.C., Raffaelli, D.G., Williams, P.J.l.B., 2005. Marine ecology: Processes, systems, and impacts. Oxford University Press.

Kar, T.K., Chakraborty, K., 2009. Marine reserves and its consequences as a fisheries management tool. WJMS. 5, 83-95.

Karagiannakos, A., 1996. Total Allowable Catch (TAC) and quota management system in the European Union. Mar. Policy. 20, 235-248.

Kaufman, L., Heneman, B., Barnes, J.T., Fujita, R., 2004. Transition from low to high data richness: An experiment in ecosystem-based fishery management from California. Bull. Mar. Sci. 74, 693-708.

Kell, L.T., Fromentin, J-M., 2009. A management strategy evaluation framework for Mediterranean Atlantic bluefun tuna. ICCAT Col. Vol. Sci. 64, 633-646.

Kelleher, G., 1999. Guidelines for marine protected areas. IUCN. Gland, Switzerland and Cambridge, UK. xxiv +107pp.

Keltz, S., Bailey, N., 2009. Fish and Nephrops Stocks Information: 2009. Fisheries Research Services.

Khalilian, S., Froese, R., Proelss, A., Requate, T., 2010. Designed for failure: A critique of the Common Fisheries Policy of the European Union. Mar. Policy. 34, 1178-1182.

Korman, S., 2011. International management of a high seas fishery: Political and property-rights solutions and the Atlantic bluefin. VA. J. Int. Law. 51, 697-748.

Koslow, J.A., Gowlett-Holmes, K., Lowry, J.K., O'Hara, T., Poore, G.C.B., Williams, A., 2001. Seamount benthic macrofauna off southern Tasmania: community structure and impacts of trawling. Mar. Ecol. Prog. Ser. 213, 111-125.

Kraak, S.B.M., Kelly, C.J., Codling, E.A., Rogan, E., 2010. On Scientists' discomfort in fisheries advisory science: the example of simulation-based fisheries management-strategy evaluations. Fish Fish. 11, 119-132.

Kristofersson, D., Rickertsen, K., 2009. Highgrading in quota-regulated fisheries: Evidence from the Icelandic cod fishery. Amer. J. Agr. Econ. 91, 335-346.

Kydland, F.E., Prescott, E.C., 1977. Rules rather than discretion: The inconsistency of optimal plans. J. Polit. Economy. 85, 473-492.

Kyriacou, A.P., 2009. Decision rules, membership and political centralization in the European Union. Europ. J. Law Econ. 27, 143-158.

Lauck, T., Clark, C.W., Mangel, M., Munro, G., 1998. Implementing the precautionary principle in fisheries management through marine reserves. Ecol. Appl. 8, S72-S78.

Law, R., 2007. Fisheries-induced evolution: present status and future directions. Mar. Ecol. Prog. Ser. 335, 271-277.

Law, R., Plank, M.J., Kolding, J., 2012. On balanced exploitation of marine ecosystems: results from dynamic size spectra. ICES J. Mar. Sci. 69, 602-614.

Le Quesne, W.J.F., Codling, E.A., 2009. Managing mobile species with MPAs: The effects of mobility, larval dispersal, and fishing mortality on closure size. ICES J. Mar. Sci. 66, 122-131.

Leal-Arcas, R., 2004. The EC in the WTO: The three-level game of decision-making. What multilateralism can learn from regionalism. European Integration online papers. 8. http://eiop.or.at/eiop/pdf/2004-014.pdf

Leigh, M., 1983. European integration and the common fisheries policy. London, Croom Helm.

Liermann, M., Hilborn, R., 2001. Depensation: evidence, models and implications. Fish Fish. 2, 33-58.

Lindenmayer, D.B., Fischer, J., Felton, A., Montague-Drake, R., Manning, A.D., Simberloff, D., Youngentob, K., Saunders, D., Wilson, D., Felton, A.M., Blackmore, C., Lowe, A., Bond, S., Munro, N., Elliott, C.P., 2007. The complementarity of single-species and ecosystem-orientated research in conservation research. Oikos. 116, 1220-1226.

Link, J.S., Bundy, A., Overholtz, W.J., Shackell, N., Manderson, J., Duplisea, D., Hare, J., Koen-Alonso, M. Friedland, K.D., 2011. Ecosystem-based fisheries management in the Northwest Atlantic. Fish Fish. 12, 152-170.

Lotze, H.K., Lenihan, H.S., Bourque, B.J., Bradbury, R.H., Cooke, R.G., Kay, M.C., Kidwell, S.M., Kirby, M.X., Peterson, C.H., Jackson, J.B.C., 2006. Depletion, degradation, and recovery potential of estuaries and coastal seas. Science. 312, 1806-1809.

Lotze, H.K., Coll, M., Magera, A.M., Ward-Paige, C., Airoldi, L., 2011. Recovery of marine animal populations and ecosystems. Trends Ecol. Evol. 26, 595-605.

Lotze, H.K., Worm, B., 2008. Historical baselines for large marine animals. Trends Ecol. Evol. 24, 254-262.

Lubchenco, J., Palumbi, S.R., Gaines, S.D., Andelman, S., 2003. Plugging a hole in the ocean: The emerging science of marine reserves. Ecol. Appl. 13, S3-S7.

Mace, P.M., 2001. A new role for MSY in single-species and ecosystem approaches to fisheries stock assessment and management. Fish Fish. 2, 2-32.

Mace, P.M., Doonan, I., 1988. A generalised bioeconomic simulation model for fish dynamics. New Zealand Fishery Assessment Research Document no. 99/4. Wellington, New Zealand.

MacKenzie, B.R., Mosegaard, H., Rosenberg, A.A., 2008. Impending collapse of bluefin tuna in the northeast Atlantic and Mediterranean. Conserv. Lett. 2, 34.

MacKenzie, B.R., Myers, R.A., 2007. The development of the northern European fishery for north Atlantic bluefin tuna *Thunnus thynnus* during 1900-1950. Fish. Res. 87, 229-239.

Mackinson, S., Wilson, D.C., Galiay, P., Deas, B., 2011. Engaging stakeholders in fisheries and marine research. Mar. Policy. 35, 18-24.

Maguire, J-J., 2001. Fisheries science and management in the North Atlantic, in: Pitcher, T., Sumaila, R., Pauly, D. (Eds.), Fisheries impacts of North Atlantic ecosystems: Evaluations and policy exploration. Fisheries Centre Research Reports. 9, 36-48.

Margules, C.R., Pressey, R.L., 2000. Systematic conservation planning. Nature. 405, 243-253.

Mathworks, 2010. Matlab version 7.1. 2010a. The Mathworks Inc.

McAllister, M.K., Babcock, E.A., 2002. Stock assessment approaches and their data requirements for dealing with mixing of western and eastern north Atlantic bluefin tuna: A Bayesian perspective. ICCAT Col. Vol. Sci. 54, 377-389.

McAllister, M., Babcock, E., Apostolaki, P., Pikitch, E.. 2004. Outlining a statistical modelling framework to support the development of a new area-based management regime for Atlantic bluefin tuna. ICCAT Col. Vol. Sci. 56, 1134-1143.

McGoldrick, D., 1997. International relations law of the European Union. London, New York, Longman.

McKane, A.J., Newman, T.J., 2005. Predator-prey cycles from resonant amplification of demographic stochasticity. Phys. Rev. Lett. 94, 218102.

McLeod, K.L., Lubchenco, J., Palumbi, S.R., Rosenberg, A.A., 2005. Scientific Consensus Statement on Marine Ecosystem-Based Management., Signed by 221 academic scientists and policy experts with relevant expertise and published by the Communication Partnership for Science and the Sea at http://compassonline.org/?q=EBM.

Megrey, B.A., Lee, Y.-W., Macklin, S.A., 2005. Comparative analysis of statistical tools to identify recruitment - environment relationships and forecast recruitment strength. ICES J. Mar. Sci. 62, 1256-1269.

Myers, R.A. Worm, B., 2003. Rapid worldwide depletion of predatory fish communities. Nature. 423, 280-283.

Myers, R.A., Worm, B., 2005. Extinction, survival or recovery of large predatory fishes. Phil. Trans. R. Soc. B. 360, 13-20.

NAFO, 2002. Report of the Working Group of Technical Experts on the Precautionary Approach (PA), 20-21 June 2002. NAFO FC Doc., No. 02/12, Serial No. N4704.

NEAFC, 2010. Information on the protection of biodiversity and mitigating impact of fisheries in the North East Atlantic. http://www.neafc.org/system/files/biodiversity_protection_report_nagoya52%20_and_54_19oct2010.pdf

Needle, C.L., 2002. Recruitment models: Diagnosis and prognosis. Rev. Fish Biol. Fisher. 11, 95-111.

NIA, 2001. The Common Fisheries Policy: Issues, Impact and Future of the CFP. Northern Ireland Assembly. Research Paper 37/01:22p. http://www.niassembly.gov.uk/io/research/3701.pdf

O'Leary, B.C., Brown, R.L., Johnson, D.E., von Nordheim, H., Ardron, J., Packeiser, T., Roberts, C.M., 2012. The first network of marine protected areas (MPAs) in the high seas: the process, the challenges, and where next. Mar. Policy. 36, 598-605.

O'Leary, B.C., Roberts, C.M., 2011. Fishery reform: ban pollitical haggling. Nature. 475: 454.

O'Leary, B.C., Smart, J.C.R., Neale, F.C., Hawkins, J.P., Newman, S., Milman, A.C., Roberts, C.M., 2011. Fisheries Mismanagement. Mar. Pollut. Bull. 62, 2642-2648.

OSPAR, 2003a. Criteria for the identification of species and habitats in need of protection and their method of application (The Texel-Faial Criteria) (Reference number: 2003-13).

OSPAR, 2003b. Guidelines for the identification and selection of marine protected areas in the OSPAR maritime area (Reference number: 2003-17).

OSPAR, 2005. Rules of procedure of the OSPAR Commission. (Reference number: 2005-17).

OSPAR, 2007. 2006 Report on the Status of the OSPAR Network of Marine Protected Areas.

OSPAR, 2010a. 2009/2010 Status Report on the OSPAR Network of Marine Protected Areas.

OSPAR, 2010b. Background Document on Milne Seamount. Publication Number: 524/2010:17p. http://www.ospar.org/documents/dbase/publications/p00524_BD_Milne_Seamount_Complex.pdf

OSPAR, 2010c. Report of the Meeting of the Ad Hoc Working Group on the Charlie Gibbs Fracture Zone. WG-Charlie 10/6/1-E, 15 February 2010.

Ostrom, E., 2008. The challenge of common-pool resources. VENV 50, 8-20.

Ottolenghi, F., 2008. Capture based aquaculture of bluefin tuna, in: Lovatelli, A., Holthus, P.F. (Eds.), Capture-based aquaculture: Global overview. Rome: UN FAO, FAO Fisheries Technical Paper No 508**:** 169-182.

Ottolenghi, F., Silvestri, C., Giordano, P., Lovatelli, A., New, M.B., 2004. Capture-based aquaculture: The fattening of eels, groupers, tunas, and yellowtails. Rome, Food and Agriculture Organisation of the United Nations.

Owen, D., 2006. The powers of the OSPAR Commission and coastal State parties to the OSPAR Convention to manage marine protected areas in the seabed beyond 200nm from the baseline. http://www.ngo.grida.no/wwfneap/Projects/Reports/WWF_Owen_Jurisdiction.pdf

Patterson, K., Résimont, M., 2007. Change and stability in landings: The responses of fisheries to scientific advice and TACs. ICES J. Mar. Sci. 64, 714-717.

Patterson, K.R., Cook, R.M., Darby, C.D., Gavaris, S., Mesnil, B., Punt, A.E., Restrepo, V.R., Skagen, D.W., Stefánsson, G., Smith, M., 2000. Validating three methods for making probability statements in fisheries forecasts. International Council for the Exploration of the Sea Theme Session on Medium-Term Forecasts in Decision-Making, ICES C.M.2000 V:06.

Pauly, D., 1980. On the interrelationships between natural mortality, growth parameters, and mean environmental temperature in 175 fish stocks. ICES J. Mar. Sci. 39, 175-192.

Pauly, D., Alder, J., Bennett, E., Christensen, V., Tyedmers, P., Watson, R., 2003. The future for fisheries. Science. 302, 1359-1361.

Pauly, D., Christensen, V., Guénette, S., Pitcher, T.J., Sumaila, U.R., Walters, C.J., Watson, R., Zelle, D., 2002. Towards sustainability in world fisheries. Nature. 418, 689-695.

Pauly, D., Maclean, J., 2002. In a perfect ocean: The state of fisheries and ecosystems in the North Atlantic Ocean. Washington : London, Island Press.

Penas, E., 2007. The fishery conservation policy of the European Union after 2002: Towards long-term sustainability. ICES J. Mar. Sci. 64, 559-595.

Perry, A.L., Low, P.J., Ellis, J.R., Reynolds, J.D., 2005. Climate change and distribution shifts in marine fishes. Science. 308, 1912-1915.

Pierce, G.J., Dyson, J., Kelly, E., Eggleton, J.D., Whomersley, P., Young, I.A.G., Santos, M.B., Wang, J., Spencer, N.J., 2002. Results of a short study on by-catches and discards in pelagic fisheries in Scotland (UK). Aquat. Living. Resour. 15, 327-334.

Pimm, S.L., Russell, G.J., Gittleman, J.L., Brooks, T.M., 1995. The future of biodiversity. Science. 269, 347-350.

Pinnegar, J.K., Blanchard, J.L., Mackinson, S., Scott, R.D., Duplisea, D.E., 2005. Aggregation and removal of weak-links in food-web models: System stability and recover from disturbance. Ecol. Modell. 184, 229-248.

Pita, C., Pierce, G.J., Theodossiou, I., 2010. Stakeholders' participation in the fisheries management decision-making process: Fishers' perceptions of participation. Mar. Policy. 34, 1093-1102.

Pitchford, J.W., Codling, E.A., Psarra, D., 2007. Uncertainty and sustainability in fisheries and the benefit of marine protected areas. Ecol. Modell. 207, 286-292.

Plank, M.J., Law, R., (in press). Ecological drivers of stability and instability in marine ecosystems. Theor. Ecol.

Poos, J.J., Bogaards, J.A., Quirijns, F.J., Gillis, D.M., Rijnsdorp, A., 2010. Individual quotas, fishing effort allocation, and over-quota discarding in mixed fisheries. ICES J. Mar. Sci. 67, 323-333.

Pörtner, H.O., Knust, R., 2007. Climate change affects marine fishes through the oxygen limitation of thermal tolerance. Science. 315, 95-97.

Poudel, D., Scandal, L.K., Kvamsdal, S.F., Steinshamn, S.I., 2011. Fisheries management under irreversible investment: Does stochasticity matter? http://papers.ssrn.com/sol3/papers.cfm?abstract_id=1957601

Press, W.H., 1992. Numerical recipes in C: the art of scientific computing. Cambridge; New York, Cambridge University Press.

Princen, S., 2010. Venue shifts and policy change in EU fisheries policy. Mar. Policy. 34, 36-41.

Punt, A.E., Smith, A.D.M., 2001. The gospel of maximum sustainable yield in fisheries management: Birth, crucifixion and reincarnation, in: Reynolds, J.D., Mace, G.M., Redford, K.H., Robinson, J.G. (Eds.), Conservation of exploited species. Cambridge, Cambridge University Press, 43-66.

R, 2008. R: A language and environment for statistical computing. Vienna, Austria, R Foundation for Statistical Computing.

Rankin, J., 2010. Problems of Scale. European Voice. http://ec.europa.eu/commission_2010-2014/damanaki/about/mandate/european_voice_290710.pdf

Reeb, C.A., 2010. Genetic discontinuity of big fish in a small sea. PNAS 107, 2377-2378.

Reiss, H., Greenstreet, S.P.R., Robinson, L., Ehrich, S., Jørgensen, L.L., Piet, G.J., Wolff, W.J., 2010. Unsuitability of TAC management within an ecosystem approach to fisheries: An ecological perspective. J. Sea Res. 63, 85-92.

Rice, J., 2008. Can we manage ecosystems in a sustainable way? J. Sea Res. 60, 8-20.

Rice, J., Cooper, J.A., 2003. Management of flatfish fisheries - what factors matter? J. Sea Res. 50, 227-243.

Ricker, W.E., 1975. Computation and interpretation of biological statistics of fish populations. Fisheries Research Board of Canada. Bulletin No.191.

Rijnsdorp, A.D., Daan, N., Dekker, W., Poos, J.J., Van Densen, W.L.T., 2007. Sustainable use of flatfish resources: Addressing the credibility crisis in mixed fisheries management. J. Sea Res. 57, 114-125.

Roberts, C.M., 1995. Effects of fishing on the ecosystem structure of coral reefs. Conserv. Biol. 9, 988-995.

Roberts, C.M., 1997. Ecological advice for the global fisheries crisis. Trends Ecol. Evol. 12, 35-38.

Roberts, C.M., 2002. Deep Impact: The rising toll of fishing in the deep sea. Trends Ecol. Evol. 17, 242-245.

Roberts, C.M., 2007. The unnatural history of the sea: The past and future of humanity and fishing. Island Press.

Roberts, C.M., Hawkins, J.P., 1999. Extinction risk in the sea. Trends Ecol. Evol. 14, 241-246.

Roberts, C.M., Hawkins, J.P., Gell, F.R., 2005. The role of marine reserves in achieving sustainable fisheries. Phil. Trans. R. Soc. B. 360, 123-132.

Robinson, L.A., Frid, C.L.J., 2008. Historical marine ecology: Examining the role of fisheries in changes in North Sea benthos. Ambio. 37, 362-372.

Rochet, M-J., Benoît , E., 2012. Fishing destabilizes the biomass flow in the marine size spectrum. Proc. R. Soc. B. 279, 284-292.

Rooker, J.R., Bremer, J.R.A., Block, B.A., Dewar, H., de Metrio, G., Corriero, A., Kraus, R.T., Prince, E.D., Rodriguez-Marin, E., Secor, D.H., 2007. Life history and stock structure of Atlantic bluefin tuna (*Thunnus thynnus*). Res. Fish. Sci. 15, 265-310.

Rosenberg, A., 2003. Managing to the margins: the overexploitation of fisheries. Front. Ecol. Environ. 1, 102-106.

Rossiter, T., Stead, S., 2003. Days at sea: from the fishers' mouths. Mar. Policy. 27, 281-288.

Safina, C., 1993. Bluefin tuna in the west Atlantic: negligent management and the making of an endangered species. Conserv. Biol. 7, 229-234.

Safina, C., Klinger, D.H., 2008. Collapse of bluefin tuna in the western Atlantic. Conserv. Biol. 22, 243-246.

Salpin, C., Germani, V., 2010. Marine protected areas beyond areas of national jurisdication: What's mine is mine and what you think is yours is also mine. RECIEL. 19, 174-184.

Salz, P., Macfadyen, G., 2007. Regional Dependency on Fisheries. Brussels, European Parliament Directorate-General for Internal Policies of the Union. IP/B/PECH/ST/IC/206-198. http://www.europarl.europa.eu/activities/expert/eStudies.do?language=EN

Sanchirico, J.N., Malvadkar, U., Hastings, A., Wilen, J.E., 2006. When are no-take zones an economically optimal fishery management strategy. Ecol. Appl. 16, 1643-1659.

Sastre, P., Lobo, J.M., 2009. Taxonomist survey biases and the unveiling of biodiversity patterns. Biol. Conserv. 142, 462-467.

Schlacher, T.A., Rowden, A.A., Dower, J.F., Consalvey, M., 2010. Seamount science scales undersea mountains: New research and outlook. Mar. Ecol. 31, 1-13.

Schnute, J., 1985. A general theory for the analysis of catch and effort data. Can. J. Fish. Aquat. Sci. 42, 414-429.

Scott, B.E., Marteinsdottir, G., Begg, G.A., Wright, P.J., Sigurd Kjesbu, O., 2006. Effects of population size/age structure, condition and temporal dynamics of spawning on reproductive output in Atlantic cod (*Gadus morhua*). Ecol. Modell. 191, 383-415.

SCRS, 2002. ICCAT workshop on bluefin mixing (Madrid, Spain, September 3-7, 2001). ICCAT Col. Vol. Sci. 54, 261-352.

SCRS, 2010. Extension of the 2009 SCRS meeting to consider the status of Atlantic bluefin tuna populations with respect to CITES biological listing criteria. Convention on International Trade in Endangered Species of Wild Fauna and Flora: Fifteenth meeting of the Conference of the Parties. Doha (Qatar), 13-25 March 2010**:** 36p.

Serchuk, F.M., Rivard, D., Casey, J., Mayo, R., 1997. Report of the Ad hoc Working Group of the NAFO Scientific Council on the Precautionary Approach. NAFO SCS Doc., No. 97/12, Serial No. N 2911.

Sethi, S., Thompson, G., 2000. Optimal control theory: Applications to management science and economics. New York, Springer.

Sharpe, D.M.T., Hendry, A.P., 2009. Life history change in commercially exploited fish stocks: An analysis of trends across studies. Evol. Appl. 2, 260-275.

Shertzer, K.W., Prager, M.H., 2007. Delay in fishery management: Diminished yield, longer rebuilding and increased probability of stock collapse. ICES J. Mar. Sci. 64, 149-159.

Shogren, J., Settle, C., 2004. Hyperbolic discounting and time inconsistency in a native-exotic conflict. Resour. Energy Econ. 26, 255-274.

Sissenwine, M., Mace, P.M., 2001. Governance for responsible fisheries: An ecosystem approach. Reykjavik Conference on Responsible Fisheries in the Marine Ecosystem. Reykjavik, Iceland.

Sovacool, B., 2009. A game of cat and fish: How to restore the balance in sustainable fisheries management. Ocean Dev. & Int'l. L.40, 97-125.

Sparholt, H., Bertelsen, M., Lassen, H., 2007. A meta-analysis of the status of ICES fish stocks during the past half century. ICES J. Mar. Sci. 64, 707-713.

Standal, D., Utne, I.B., 2011. The hard choices of sustainability. Mar. Policy. 35, 519-527.

STECF, 2009. The 2009 annual economic report on the European fishing fleet. Scientific, Technical and Economic Committee for Fisheries (STECF), JRC Scientific and Technical Reports. Report EUR 24069 EN.

Steneck, R.S., Graham, M.H., Bourque, B.J., Corbett, D., Erlandson, J.M., Estes, J.A., Tegner, M.J., 2002. Kelp forest ecosystems: biodiversity, stability, resilience and future. Environ. Conserv. 29, 436-459.

Stewart, G.B., Kaiser, M.J., Côte, I.M., Halpern, B.S., Bayliss, H.R., Pullin, A.S., 2009. Temperate marine reserves: Global ecological efforts and guidelines for future networks. Conserv. Lett. 2. 243-253.

Stokstad, E., 2010. Trade trumps science for marine species at international meeting. Science. 328, 26-27.

Sumaila, U.R., Huang, L., 2012. Managing bluefin tuna in the Mediterranean Sea. Mar. Policy. 36, 502-511.

Sumaila, U.R., Khan, A.S., Dyck, A.J., Watson, R., Munro, G., Tydemers, P., Pauly, D., 2010. A bottom-up re-estimation of global fisheries subsidies. J. Bioecon. 12, 201-225.

Sumaila, U.R., Marsden, A.D., Watson, R., Pauly, D., 2007. A global ex-vessel fish price database: Construction and applications. J. Bioecon. 9, 39-51.

Surís-Regueiro, J.C., Varela-Lafuente, M.M., Garza-Gil, M.D., 2011. Evolution and perspectives of the Fisheries Structural Policy in the European Union. Ocean Coast. Manage. 54, 593-600.

Sutinen, J.G., Soboil, M., 2001. The performance of fisheries management systems and the ecosystem challenge. Reykjavik Conference on Responsible Fisheries in the Marine Ecosystem, Reykjavik, Iceland, 1-4 October 2001.

Symes, D., 1997. Fisheries management: In search of good governance. Fish. Res. 32, 107-114.

Symes, D., 2007. Fisheries management and institutional reform: A European perspective. ICES J. Mar. Sci. 64, 779-785.

Taylor, N., McAllister, M., Block, B.A., Lawson, G., 2009. A multi stock tag integrated age structured assessment model for the assessment of Atlantic bluefin tuna. ICCAT Col. Vol. Sci. 64, 513-531.

Turner, S.J., Thrush, S.F., Hewitt, J.E., Cummings, V.J., Funnell, G., 1999. Fishing impacts and the degradation or loss of habitat structure. Fisheries Manag. Ecol. 6, 401-420.

UN, 2002. Report of the World Summit on Sustainable Development, Johannesburg, South Africa, 26 August - 4 September 2002. A/CONF.199/20. United Nations, New York, USA. http://daccess-dds-ny.un.org/doc/UNDOC/GEN/N02/636/93/PDF/N0263693.pdf?OpenElement

UNEP, 2007. Deep-sea biodiversity and ecosystems: A scoping report on their socio-economy, management and governance. http://www.unep-wcmc.org/resources/publications/UNEP_WCMC_bio_series/28/Deep-Sea%20Biodiversity%20and%20Ecosystems_small.pdf

Villasante, S., 2010. Global assessment of the European Union fishing fleet: An update. Mar. Policy. 34, 663-670.

Villasante, S., Sumaila, U.R., 2010. Estimating the effects of technological efficiency on the European fishing fleet. Mar. Policy. 34, 720-722.

Voinov, A., Farley, J., 2007. Reconciling sustainability, systems theory and discounting. Ecol. Econ. 63, 104-113.

von Bertalanffy, L., 1938. A qualitative theory of organic growth. Hum. Biol. 10, 181-213.

Waller, R., Watling, L., Auster, P., Shank, T., 2007. Anthropogenic impacts on the Corner Rise Seamounts, north-west Atlantic Ocean. J. Mar. Biol. Assoc. UK. 87, 1075-1076.

Walli, A., Teo, S.L.H., Boustany, A.M., Farwell, C.J., Williams, T.D., Dewar, H., Prince, E., Block, B.A., 2009. Seasonal movements, aggregations and diving behavior of Atlantic bluefin tuna (*Thunnus thynnus*) revealed with archival tags. PLoS ONE. 4, 1-18.

Walters, C.J., 1998. Designing fisheries management systems that do not depend on accurate stock assessment, in Pitcher, T.J., Hart, P.J.B., Pauly, D. (Eds.), Reinventing fisheries management. London, Chapman and Hall, 279-288.

Webster, D.G., 2011. The irony and the exclusivity of Atlantic bluefin tuna management. Mar. Policy. 35, 249-251.

West, C.D., Dytham, C., Righton, D., Pitchford, J.W., 2009. Preventing overexploitation of migratory fish stocks: The efficacy of marine protected areas in a stochastic environment. ICES J. Mar. Sci. 66, 1919-1930.

Worm, B., Barbier, E.B., Beaumont, N., Duffy, E., Folke, C., Halpern, B.S., Jackson, J.B.C., Lotze, H.K., Micheli, F., Palumbi, S.R., Sala, E., Selkoe, K.A., Stachowicz, J.J., Watson, R., 2006. Impacts of biodiversity loss on ocean ecosystem services. Science. 314, 787-790.

Worm, B., Hilborn, R., Baum, J.K., Branch, T.A., Collie, J.S., Costello, C., Fogarty, M.J., Fulton, E.A., Hutchings, J.A., Jennings, S., Jensen, O.P., Lotze, H.K., Mace, P.M., McClanahan, T.R., Minto, C., Palumbi, S.R., Parma, A.M., Ricard, D., Rosenberg, A.A., Watson, R., Zeller, D., 2009. Rebuilding global fisheries. Science. 325, 578-585.

Worm, B., Sandow, M., Oschlies, A., Lotze, H.K., Myers, R.A., 2005. Global patterns of predator diversity in the open oceans. Science. 309, 1365-1369.

Wright, R., 2004. A short history of progress. Edinburgh, Canongate Books Ltd.