Investigating the biological and socio-economic impacts of marine protected area network design in Europe

Submitted by

Kristian Metcalfe

January 2013

to the Durrell Institute of Conservation and Ecology,

School of Anthropology and Conservation,

University of Kent as a thesis for the degree of

Doctor of Philosophy in Biodiversity Management

DICE

University of Kent

SUPERVISORS

Dr Robert J Smith - Durrell Institute of Conservation and Ecology (DICE)

Dr Sandrine Vaz - Institut Français de Recherche pour l'exploitation de la Mer (IFREMER)

Professor Stuart R Harrop - Durrell Institute of Conservation and Ecology (DICE)

ABSTRACT

Marine ecosystems are under increasing pressure from a diverse range of threats. Many national governments have responded to these threats by establishing marine protected area (MPA) networks. One such approach for designing MPA networks is systematic conservation planning, which is now considered the most effective system for designing protected area networks. However, the main exception to this trend is Europe, where the designation of MPAs is still largely based on expert opinion, despite growing awareness that these existing methods are not the most effective. Therefore, there is a need to demonstrate how systematic conservation planning can be used to inform MPA design in European waters and show how this approach can fit within existing marine conservation policy and practice. This thesis brings together a range of biological, legal and socio-economic data to address these issues and is comprised of four main chapters:

After the introductory chapter, this thesis begins with a review of how existing approaches for guiding the selection of MPAs in Europe compare to conservation planning best practice (**Chapter 2**). Here I show that whilst existing legislation has widespread political support and has underpinned the rapid expansion of MPA networks, it fails to incorporate three key elements from systematic conservation planning which are designed to identify MPA networks that achieve conservation goals, minimise impacts on stakeholders, and facilitate implementation. These include the extent to which current legislation fails to: (i) translate broad policy goals into quantitative targets; (ii) incorporate socio-economic data; and (iii) requires a social assessment.

In **Chapter 3** I investigate the species-area relationship (SAR) based approach that has been used to set conservation targets for marine habitats in the UK. Here I use data from the English Channel to show this approach is strongly influenced by changes in: (i) the number of samples used to generate estimates of species richness for each habitat; (ii) the different estimators used to calculate species richness; and (iii) the resolution of the habitat classification. However, whilst each of these tested factors had an influence on targets, this work found that the number of samples had the greatest impact.

In **Chapter 4** I examine the impact of using MPA size constraints when designing MPA networks in English waters, and show that increasing the size of MPAs would result in MPA networks that are only slightly larger but more costly to stakeholders. The findings also suggest that increasing this minimum size constraint produces MPA networks comprised of fewer MPAs that are more widely distributed throughout the planning region, thus reducing potential connectivity for a range of species.

Finally, in **Chapter 5** I use an ecosystem model of the eastern English Channel to investigate the potential trade-offs associated with different spatially explicit MPA management strategies. In particular, I show that broader classes of spatial management based on zoning fleet access and gear restrictions can have conservation and fisheries benefits, which is important given that this approach is less politically contentious than strict no-take MPA networks. However, I also demonstrate that if MPA networks are to ensure the sustainable use of fisheries they should be comprised of at least 60% no-take zones and that a 100% no-take MPA network would produce substantial increases in exploited ecosystem biomass and fisheries catches. Equally importantly, I show that exploited catches recovered six times as quickly in 100% no-take MPA networks when compared to 100% limited-take MPA networks.

Collectively, these chapters demonstrate the value of adopting a systematic approach to MPA network design in Europe, as it: (i) provides a flexible and transparent platform for exploring different designs and management strategies; and, (ii) can be combined with spatial prioritisation and decision support tools to help identify and manage priority areas that meet regional and national obligations, minimise impacts on stakeholders, and fit within existing policy frameworks.

ACKNOWLEDGEMENTS

I would like to thank my supervisors Bob Smith, Stuart Harrop and Sandrine Vaz for their invaluable help and support. I have genuinely and thoroughly enjoyed working with you all. I would also like to take this opportunity to personally thank Bob for providing me with many other opportunities outside of my PhD that have helped me develop both academically and personally.

I must also thank Greg Vaughan of Balanced Seas for providing me with support and guidance when ArcGIS disagreed with me and Sue Wells also of Balanced Seas and Natural England for useful discussions relating to my research. I would also like to thank Catherine McClellan and Juliette Delavenne for useful discussions relating to our work on the CHARM project.

I have also received an invaluable amount of support in the form of the weekly 'School of Anthropology and Conservation Social', so a big thanks to Diogo Veríssimo and all those at DICE. A special thank you must also go to the Unicorn Inn and those who always come for a beer (you shall remain nameless). I would also like to thank Simon Tollington, Simon Raby, Andy Curtis, Matthew Neighbour, Joe Burden, and Giles Darymple, who have unknowingly been a big support and kept me entertained from start to finish.

Finally, and most importantly I would like to take this opportunity to thank my family and loved ones. Mum, Dad, Ross, Brett, Pat, John, Paddy, and Kate none of this would have been possible without you – so thank you for all your help, support and patience.

AUTHOR'S DECLARATION

The research for this thesis was supported by funding from the European Union under the Interreg IVA France (Channel) England cross-border European cooperation programme, which was co-financed by the European Regional Development Fund as part of the Channel Integrated Approach for Marine Resource Management (CHARM) Phase III project (Nº 4037).

All chapters in this thesis were written by K. Metcalfe and are formatted in the style of manuscripts for publication, and are denoted with an (*) where published. Comments and editing was provided by R. J. Smith, S. R. Harrop and S. Vaz. However, this work would not have been possible without the additional support, data and contributions of others detailed in the acknowledgements at the end of each chapter.

CONTENTS

TABLES

FIGURES

CHAPTER 1

GENERAL INTRODUCTION

1.1 MARINE ECOSYSTEM THREATS: AN OVERVIEW

Coastal and marine ecosystems are amongst the most heavily degraded in the world, with marine biodiversity under increasing pressure from a diverse range of threats that vary both in their intensity and spatial distribution (Lubchenco *et al.*, 2003; Halpern *et al.*, 2008). The over-exploitation of marine resources, pollution, habitat destruction, invasive species and global climate change has played a central role in driving unanticipated, unprecedented and complex changes in the chemistry (Islam & Tanaka 2004; Brierley & Kingsford 2009), physical structure (Turner *et al.*, 1999; Coleman & Williams 2002), and biological and ecological functioning (Hutchings 2000; Jackson *et al.*, 2001; Harley *et al.*, 2006) of oceans worldwide. The impacts of which are numerous and diverse, and often involve complicated interactions with unpredictable effects (Pinnegar *et al.*, 2000; Shears & Ross 2010).

Over the last decade there has however been a rapid increase in our understanding and knowledge of human impacts on marine ecosystems (Roberts 2003), where changes described have ranged from direct impacts at local scales (Hughes *et al.*, 2003; Carpenter *et al.*, 2008), to indirect but far reaching effects operating at regional and global scales (Pauly *et al.*, 2005; Worm *et al.*, 2006). Growing evidence also suggests that advances in technology have opened up previously unknown or unexploited deeper offshore waters that served as refuges to exploitation and its associated impacts. Nonetheless, despite increasing awareness new and cumulative impacts to marine ecosystems are increasing markedly, exacerbating existing environmental problems (Roberts 2003).

1.2 GLOBAL MARINE PROTECTION TARGETS

Consequently, increased concerns about rapid declines in the environmental status of many of the world's oceans (Halpern *et al.*, 2008), inadequate regulation of user groups (Carr *et al.*, 2003), the failure of existing management regimes (Beddington *et al.*, 2007), and impacts of biodiversity loss on ecosystem services (Worm *et al.*, 2006) have triggered calls for more effective approaches to protect marine ecosystems (Allison *et al.*, 1998; Lubchenco *et al.*, 2003). In response, the establishment of marine protected areas (hereafter referred to as MPAs) are now strongly emphasised in environmental policy throughout the world (Spalding *et al.,* 2008; Wood *et al.,* 2008; Wood 2011), and this has resulted in their inclusion in several global marine protection targets and international obligations (**Table 1.1**). The most widely known of which is the goal of protecting and effectively managing 10% of the global ocean area in MPAs (CBD 2010b).

1.3 MARINE PROTECTED AREAS

The definition of a MPA varies both in the literature and in practice due to differing perceptions of their role and objectives; such as the activities (extractive and non-extractive) regulated both within and outside their boundaries, and the effectiveness with which those regulations are implemented (Agardy *et al.,* 2003; Wood 2011). However, the most authoritative definition has been provided by the Convention on Biological Diversity (CBD), which states that a MPA is: *"…any defined area within or adjacent to the marine environment, together with its overlying waters and associated flora, fauna and historical and cultural features, which has been reserved by legislation or other effective means, including custom, with the effect that its marine and/or coastal biodiversity enjoys a higher level of protection than its surroundings"* (CBD 2004b). The CBD also state that a MPA network is: *"…a portfolio of biologically connected protected areas that is fully representative of the range of target ecosystems, species, and processes including in marine areas beyond national jurisdiction"* (CBD 2009).

The term MPA is therefore associated with a highly diverse range of protected areas. Some aim to protect and conserve the functioning and integrity of marine ecosystems, some are intended as resource management tools, whilst others can be designed to address both these objectives (Jentoft *et al.,* 2011). Consequently, MPAs range from small highly protected inshore sites to huge sections of the open ocean where certain activities are prohibited and others carefully monitored, and can have varying levels of protection; ranging from no-take areas that restrict all activities to partially protected areas that allow selective extraction of resources (Wood 2007).

Table 1.1 Global marine protection targets and international commitments.

Convention (Source)	Year Adopted	Deadline	Target (%)	Target pertains to:	Original text, goals, targets and recommendations:
World Summit on Sustainable Development, Johannesburg, South Africa (United Nations 2002)	2002	2012	-	Global Ocean	Section IV, paragraph 32(a): "Maintain the productivity and biodiversity of important and vulnerable marine and coastal areas, including in areas within and beyond national jurisdiction" Section IV, paragraph 32(c): "the establishment of marine protected areas consistent with international law and based on scientific information, including representative networks by 2012"
5th World Parks Congress, Durban, South Africa (IUCN 2003).	2003	2012	20-30	Global Ocean	Recommendation 5.22: "Establish by 2012 a global system of effectively managed, representative networks of marine and coastal protected areas" and that "these networks should be extensive and include strictly protected areas that amount to 20 - 30% of each habitat"
7th Conference of the Parties to the Convention on Biological Diversity, Kuala Lumpur, Malaysia (CBD 2004b; CBD 2004c; CBD 2004d).	2004	2021	10	Areas under National Jurisdiction	Decision VII/5: (Operational objective 3.1): "To establish and strengthen national and regional systems of marine and coastal protected areas integrated into a global network and as a contribution to globally agreed goals" Decision VII/28 (Goal 1.1): "By 2010, terrestrially and 2012 in the marine area, a global network of comprehensive, representative and effectively managed national and regional protected area system is established" Decision VII/30 (Goal 1.1) "At least 10% of each of the world's ecological

					regions effectively conserved” Decision VII/30 (Goal 1.2 Target): “Areas of particular importance to biodiversity protected”
8th Conference of the Parties to the Convention on Biological Diversity, Curitiba, Brazil (CBD 2006).	2006	2012	10 [a]	Global Ocean	A suggested activity of the Parties under this target was to: “By 2006, establish suitable time bound and measurable national and regional level protected area targets and indicators” Decision VIII/15: “at least 10% of each of the world’s ecological regions [including marine and coastal] be effectively conserved [by 2012]”
10th Conference of the Parties to the Convention on Biological Diversity, Nagoya Japan (CBD 2010b; CBD 2011).	2010	2020 [b]	10	Global Ocean	Decision X/2 (Target 11): “By 2020, at least 10% of coastal and marine areas 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”

[a] The 10% target is not strictly about MPAs, as “effective conservation” was more broadly defined to include “other means of area based protection, for which management plans exist” (CBD 2005). [b] Due to the slow progress towards achieving the marine protection target first adopted in 2006, the 10% target remained unchanged although the deadline was extended from 2012 to 2020 at the 10th Conference of Parties in 2010.

1.3.1 Current status of global marine protected area networks

The number of MPAs has increased from 4,116 in 2003 to 5,878 in 2010, covering 4.21 million km^2 (1.17%) of the total ocean area (**Table 1.2**). However, despite increasing efforts there are still doubts about the ability to meet global marine protection targets (Wood *et al.,* 2008), as marine conservation still lags several decades behind terrestrial conservation in the establishment of protected areas (Spalding *et al.,* 2008; Fox *et al.,* 2012a). There is also a significant bias towards the location of MPAs, with almost all sites located entirely within areas of national jurisdiction (with the exception of 38 sites in Antarctica), covering approximately 4.12 million km^2 (2.86%) of waters within 200 nautical mile Exclusive Economic Zones (**Table 1.2**). Moreover, a large proportion of these sites have been designated within territorial waters (< 12 nautical miles) to conserve temperate rocky nearshore environments and coral reefs in tropical marine ecosystems (Spalding *et al.,* 2008). Thus, the focus of conservation efforts is often directed at continental shelf areas where MPA coverage is 1.27 million km^2 (4.32%). In contrast, 3.01 million km^2 (0.91%) of off-shelf areas are protected (**Table 1.2**), leaving vast areas of the open ocean unprotected, despite their biogeographic, ecological and conservation importance.

Table 1.2 Summary of recent growth in the number and coverage of marine protected areas globally.

MPA Statistics	**2003**[a]	**2006**[b]	**2008**[c]	**2010**[d]
Number of MPAs	4116	4435	5045	5878
Global total (million km^2)	1.64 (0.45%)	2.35 (0.65%)	2.59 (0.72%)	4.21 (1.17%)
Within EEZs (million km^2)	1.64 (1.14%)	2.35 (1.63%)	2.59 (1.80%)	4.12 (2.86%)
On continental shelf (million km^2)			1.20 (4.09%)	1.27 (4.32%)
Off-shelf (million km^2)			1.39 (0.42%)	3.01 (0.91%)

Source: [a] Chape *et al.,* (2003); [b] Wood *et al.,* (2008); [c] Spalding *et al.,* (2008); and [d] Torpova *et al.,* (2010).

The size of MPAs is also highly variable, ranging from less than a hectare (0.01 km^2) to more than 100,000 km^2 (Fox *et al.,* 2012a), with 46% (2700) of sites covering less than 1 km^2 (Torpova *et al.,* 2010). In addition, almost 2.5 million km^2 of global ocean area (60% of the entire global coverage) is protected by 11 MPAs which are larger than 100,000 km^2 (Torpova *et al.,* 2010). Consequently, the total global coverage is comprised of a relatively small number of very large MPAs, combined with a large number of very small sites. Therefore, the current distribution, extent, sizing and spacing of MPAs globally is considered vastly inadequate, with only half of the world's MPAs forming part of a coherent network (Laurel & Bradbury 2006; Spalding *et al.,* 2008; Wood *et al.,* 2008).

Although guidelines exist on scaling up MPAs (IUCN-WCPA 2008), the establishment of MPA networks can frequently take more time than envisaged, as they are often hampered by a range of biological and socio-political factors including: divergent interests of stakeholders in marine resource governance; legal difficulties in defining boundaries and protecting important areas located in international waters; conflicting interests between resource users; and the scarcity of data, particularly for offshore waters (Agardy 1994; Sumaila *et al.,* 2000; Leathwick *et al.,* 2008; Fox *et al.,* 2012a). Furthermore, experience from around the world has shown the lack of perceived benefits associated with fishery closures, especially no-take MPAs, are a common thread among objections from fisheries groups (Hutchings 2000; Roberts & Hawkins 2000; Gell & Roberts 2003).

1.3.2 Biological impact of marine protected area networks

Although much needs to be done to increase their global coverage and effectiveness, MPAs are expanding globally (**Table 1.2**), and are increasingly advocated as an important tool for ecosystem-based management (Halpern *et al.,* 2010). This is because they have the potential to address a broad array of management goals (Pollnac *et al.,* 2010). There is also growing consensus that MPAs, especially 'no-take' MPAs, can provide a variety of ecological, economic and social benefits; and these benefits have been the subject of numerous scientific studies and reviews over the last two decades (e.g. Babcock *et al.,* 1999; Roberts & Hawkins 2000; Roberts *et al.,* 2001; Halpern & Warner 2002; Gell & Roberts 2003; Halpern 2003; Shears & Babcock 2003; Russ *et al.,* 2004; Pillans *et al.,* 2005; Claudet *et al.,* 2006; Harmelin-Vivien *et al.,* 2008; Russ *et al.,* 2008).

In terms of ecosystem integrity, MPAs can generate a range of potential benefits. Within their boundaries they can protect species and habitats (including nursery and spawning grounds) from damaging activities and exploitation (Turner *et al.,* 1999); restore habitat complexity and structure, essential for prey protection and in the role of recruitment (Botsford *et al.,* 1997; Jones *et al.,* 2004); and create more natural and desirable population structures (characterised by age, gender or individual size), which can result in increased breeding success and recruitment to exploited areas as larger older individuals are often more highly fecund (Bohnsack 1998; Jennings 2000; Birkeland & Dayton 2005). Beyond their boundaries they can enhance surrounding areas through processes known as ‘spill-over’ and ‘export’ (Gell & Roberts 2003; Roberts *et al.,* 2005). The former suggests that as populations within a MPA increase, adult and juvenile species will migrate or spill-over into surrounding areas (Gell & Roberts 2003). The latter assumes that when species reach maturity and spawn, their eggs, larvae or other propagules will be transported in the water column to surrounding areas, supporting and enhancing populations outside the boundaries of protected areas (Gell & Roberts 2003; Higgins *et al.,* 2008). Evidence supporting the contribution of spill-over to exploited areas has been demonstrated by several studies that have indicated: a gradient of biomass and/or species richness that decreased from higher to less protected areas (Harmelin *et al.,* 1995; Garcia-Charton *et al.,* 2004; Goni *et al.,* 2006; Harmelin-Vivien *et al.,* 2008); increased catches per unit effort and increased population sizes in adjacent areas (Gell & Roberts 2003); and harvests of larger and more highly valued species (Bhat 2003).

In addition to recovering stocks and increasing the size, abundance and catches of commercially valuable species (Murawski *et al.,* 2000; Pillans *et al.,* 2005; Claudet *et al.,* 2006; Hoskin *et al.,* 2011) there is also evidence that MPAs provide a number of indirect effects for non-target species and habitats through trophic cascades (Pinnegar *et al.,* 2000; Shears & Babcock 2003; Babcock *et al.,* 2010). In contrast to direct effects that are often detectable over a relatively short time frame, indirect effects such as those resulting from trophic interactions tend to accrue more slowly, and may take decades (Edgar *et al.,* 2009; Babcock *et al.,* 2010). However, the few changes that have been observed in populations of non-target species in MPAs are thought to result from indirect effects that develop after the restoration of populations of higher trophic level species (Babcock *et al.,* 1999; Shears & Babcock 2003; Behrens & Lafferty 2004). In tropical marine ecosystems the recovery of herbivorous fish populations have been shown to lead to a decrease in macroalgal biomass, resulting in enhanced recruitment of corals due to increased space and reduced competition (Mumby *et al.,* 2006; Mumby *et al.,* 2007). In temperate reef ecosystems the recovery of lobsters and large fish have led to higher predation and decline of sea urchin populations (Babcock *et al.,* 1999) and the recovery of kelp forests as a result of a reduction in the density of grazing species (Shears & Babcock 2003). However, the absence of sufficient baseline data can often make it difficult to know whether these changes represent full or partial recovery (Pinnegar & Engelhard 2008).

Nonetheless, in a review of 89 MPAs biological measures such as population density, biomass, species diversity and size were markedly higher within protected area boundaries compared to reference sites; i.e. the same site before designation or adjacent areas outside the boundaries of the MPAs (Halpern 2003). Furthermore, this research and similar work has demonstrated that such increases are often independent of protected area size, indicating that even small MPAs can contribute towards the conservation of marine biodiversity (Halpern 2003; Guarderas *et al.,* 2011; Hoskin *et al.,* 2011). However, studies that have quantified the rate at which recovery may take place have found that there are many factors affecting recovery of populations in MPAs, including: initial population size, relationships with source locations, size of MPA, annual variations in individual recruitment events, and the degree to which existing levels of fishing have affected populations (Babcock *et al.,* 2010).

1.3.3 Uncertainty surrounding marine protected area networks

Despite the wealth of knowledge concerning the benefits of MPAs some scientists and stakeholders remain sceptical, arguing that the potential benefits of excluding activities is an easy concept for non-specialists to grasp, making MPAs an overly alluring alternative to the current array of management tools (Kaiser 2005). They state that: (i) we still lack experience with MPAs implemented at large scales, particularly in temperate waters and settings with industrial fisheries (Roberts *et al.,* 2005); (ii) they are often implemented without a firm understanding of conservation science, failing to address both the ecological and socio-economic components underpinning marine protection (Agardy *et al.,* 2003); (iii) MPAs often place the welfare of marine resource users above the well-being of communities that are dependent on access to marine resources (West *et al.,* 2006; Mascia *et al.,* 2010); and, (iv) reducing fishing effort may be equally effective for achieving conservation and management objectives (Allison *et al.,* 1998).

Many stakeholders also claim that scientific evidence used to support their designation is based on studies that focus on: (i) projects that demonstrate a positive outcome of MPA implementation (Roberts *et al.,* 2001; Halpern & Warner 2002; Gell & Roberts 2003; Halpern 2003); (ii) habitat specific species associated with coral and temperate rocky reefs, which are relatively small in scale and easier to protect (Halpern 2003; Claudet *et al.,* 2006); (iii) less mobile or sedentary species that are more likely to benefit from the exclusion of human activities (Murawski *et al.,* 2000); and (iv) species which are commercially valuable, disregarding species and habitats that play an important role in ecosystem functioning (Jouvenel *et al.,* 2004; Pillans *et al.,* 2005; Claudet *et al.,* 2006). Thus, there are still some doubts as to whether these scientific conclusions are valid for many temperate and commercially important species that are widespread across a variety of habitats, exhibit entirely different life history characteristics and which are highly mobile and move considerable distances each year (Hilborn *et al.,* 2004; Kaiser 2005).

In terms of impacts on fisheries, evidence suggests that fishing effort is often highly aggregated and that large areas of the seabed often remain un-fished, while other areas receive intensive fishing pressure (Rijnsdorp *et al.,* 1998; Kaiser 2005). Excluding access could therefore displace current activities, increasing the impacts associated with over-exploitation and thereby causing wider ecological damage to areas that have been previously undisturbed (Kaiser 2005; Stefansson & Rosenberg 2005). Moreover, areas impacted by fishing can often be slow to recover as many important habitat forming species are slow colonisers (Turner *et al.,* 1999). Therefore, displacement of fishing effort could leave a significantly larger portion of the seabed in an altered state before recolonisation occurs in newly protected areas (Dinmore *et al.*, 2003). In addition, research suggests that if fishers are displaced from favoured fishing grounds into areas where catch rates of legal sized target species are lower, the use of existing quota management controls are likely to result in considerable increases in bycatch as fishers work harder to land the same quota, and could even risk displacing fishing effort onto more vulnerable habitats and life stages (Horwood *et al.*, 1998; Kaiser 2005). The establishment of MPAs could also induce a shift of fishing effort towards other fisheries sectors and target species, creating new conflicts between user groups (Murawski *et al.*, 2000; Sanchirico *et al.*, 2002; Hilborn *et al.*, 2004).

Finally, many sceptics argue that the ineffective nature of some MPAs demonstrates that they are not the only solution (McClanahan 1999; Jameson *et al.*, 2002), and that mitigating threats to marine ecosystems will require adopting a suite of strategies, which should include incentives to encourage conservation and sustainability, and build awareness of the value of biodiversity (Leslie 2005). They also argue that MPAs should be placed into a broader management framework for the sustainable use of marine resources, which incorporates watershed management, marine spatial planning, shipping regulations and fishery controls such as quotas and gear restrictions (Allison *et al.*, 1998; Kaiser 2005). Additional evidence about the need to implement MPAs in conjunction with other management tools is based on the fact that even the most well designed and managed protected areas cannot protect habitats, species and ecosystems from the activities outside of their boundaries (Agardy *et al.*, 2011). Therefore, without adequate management in adjacent areas the effectiveness of MPAs will be severely compromised (Agardy 1994; Allison *et al.*, 1998). This is because uncontrolled pollution and unsustainable exploitation of marine resources outside MPA boundaries can adversely affect species and ecosystem functioning inside protected areas (Keller *et al.*, 2009).

1.3.4 Socio-economic impact of marine protected area networks

Given that MPAs have essentially been designed to govern human activities in a defined area, their implementation is commonly seen as politically and socially contentious (Charles & Wilson 2009; Mascia & Claus 2009). This is because they often restrict and control access to the economic wealth associated with the exploitation of natural resources. However, whilst research indicates that the impacts and distribution of benefits on various groups will differ (Mangi & Hattam 2009), the socio-economic impact of MPAs are often poorly studied or even acknowledged (Badalamenti *et al.,* 2000; Sanchirico *et al.,* 2002; Christie *et al.,* 2003). Moreover, very few peer-reviewed studies have quantified the social impacts (Fox *et al.*, 2012a).

Research efforts have largely focused on the impacts of MPAs on discrete activities such as fisheries (Klein *et al.,* 2008a; Klein *et al.,* 2008b; Scholz *et al.,* 2010); tourism (Agardy 1993; Davis & Tisdell 1996; Hargreaves-Allen *et al.,* 2011); and recreation (Lynch *et al.,* 2004). Though research shows that protection of natural resource bases such as breeding, nursery and recruitment habitats provide the most important economic revenues to be derived from establishing MPAs (Harmelin *et al.,* 1995; Russ *et al.,* 2004; Higgins *et al.,* 2008; Hoskin *et al.,* 2011). In addition, several studies have reported that MPAs have had a considerable impact on the local and regional economy, as they can provide economically valuable activities, create new jobs (diversification of livelihoods), and increase revenue in the form of tourist taxes and expenditure from non-consumptive recreation and tourism (Farrow 1996; Badalamenti *et al.,* 2000; Sanchirico *et al.,* 2002; Lloret *et al.*, 2008).

Nonetheless, some sceptics argue that the increasing effort to enhance marine biodiversity through the implementation of MPAs will negatively affect the livelihoods and social well-being of communities who are already poor and marginalised, and therefore most dependent on access to marine resources (Christie *et al.,* 2003; Mascia & Claus 2009; Mascia *et al.,* 2010). Moreover, research has shown that these effects are likely to be compounded with increasing size of no-take MPAs (Mangi & Hattam 2009). This is because fisheries may face significant upfront costs following MPA establishment, as the displacement of fishing effort, or exclusion of gear types has a direct influence on operating costs, especially if they are required to travel further from traditional fishing grounds, as this will increase the costs associated with the increased time required to meet their quota (Hilborn *et al.,* 2004; Smith *et al.,* 2010a).

In addition to the substantial amount of scientific uncertainty surrounding the designation of MPAs, their implementation is often fraught with socio-economic problems. This is because they are unlikely to be a cheaper alternative to existing management measures given the costs associated with consultation, planning, implementation, administration, management, enforcement (Kaiser 2005; McCrea-Strub *et al.,* 2011), and in some cases compensation (Roberts & Hawkins 2000). However, such costs are often not incorporated accurately when proposing areas for protection (Torpova *et al.,* 2010). From a socio-economic perspective, MPAs are regarded as an investment of public resources (Sanchirico 2000; Sanchirico *et al.,* 2002), and preliminary investigations into understanding their true cost has revealed that total cost is often correlated with the size of MPA and the duration of the establishment phase (Balmford *et al.,* 2004; McCrea-Strub *et al.,* 2011). These studies also suggest that whilst the total establishment cost is expected to be higher for larger MPAs when considered per unit area, smaller MPAs may be more expensive to establish, reflecting economies of scale (McCrea-Strub *et al.,* 2011), and that annual running costs were higher for MPAs that were smaller, closer to coasts and in developed countries (Balmford *et al.,* 2004).

1.4 SYSTEMATIC CONSERVATION PLANNING

Given the increase in our understanding of human impacts on coastal and marine ecosystems a primary focus of conservation efforts to date has been the establishment of MPAs (**Table 1.1**; **Table 1.2**). However, over the last few decades there has been growing awareness that existing approaches have not been the most effective, and that the extent to which MPAs protect biodiversity depends in part on the selection of areas that maximise the representation and long term persistence of biodiversity (Margules & Pressey 2000). It is also widely recognised that conservation planners need to account for opportunity costs and potential biodiversity loss when designing MPA networks (Arkema *et al.*, 2006; Ban & Klein 2009). In this context, this has led to the widespread adoption of systematic conservation planning (Margules & Pressey 2000), which emerged in an attempt to redress the biodiversity losses incurred by previous *ad hoc* allocations of protected areas (Pressey *et al.*, 1993; Pressey 1994). This approach is based around a clear and transparent framework (**Table 1.3**) that essentially combines a short-term conservation assessment, which is the process of identifying priority areas (Stage 1 – 4; **Table 1.3**), with a long-term implementation strategy (Stage 5 – 6; **Table 1.3**) that is used to achieve conservation action (Knight *et al.*, 2006a; Knight *et al.*, 2006b).

Table 1.3 Systematic conservation planning framework.

Key stages in systematic conservation planning[a]
1. *Compile data on the biodiversity of the planning region:* ▪ Review existing data and decide on which data sets are sufficiently consistent to serve as surrogates for biodiversity across the planning region. ▪ If time and/or money allows, collect new data to augment or replace some existing data sets. ▪ Collect information on the localities of species considered to be rare and/or threatened in the region, which are likely to be missed or under-represented in conservation areas selected only on the basis of habitat classes.
2. *Identify conservation goals for the planning region:* ▪ Set quantitative targets for species, vegetation types or other biodiversity features. ▪ Set quantitative targets for minimum size, connectivity, ecological processes or other design criteria. ▪ Identify qualitative targets or preferences (for example, new conservation areas should have minimal previous disturbance or not impede on economic output such as fisheries).
3. *Review existing conservation areas:* ▪ Measure the extent to which the quantitative targets for representation and design have been achieved by existing conservation areas. ▪ Identify the threats to under-represented features such as species or habitat types, and the threats posed to areas that are important to meeting design targets.
4. *Select additional conservation areas:* ▪ Regard established conservation areas as ‘constraints’ or focal points for the design of an expanded system. ▪ Identify preliminary sets of new conservation areas for consideration as additions to established areas. This can be achieved through reserve selection algorithms or decision- support tools to allow practitioners and stakeholders to design expanded systems that achieve regional goals subject to constraints such as existing reserves, acquisition budgets or limits on feasible opportunity costs for other activities.
5. *Implement conservation actions:* ▪ Decide on the most appropriate or feasible form of management to be applied to the individual conservation areas.

- If one or more selected areas prove to be unexpectedly degraded or difficult to protect, return to stage 4 and identify alternatives.
- Decide on the relative timing of conservation management when resources are insufficient to implement the whole system in the short term.

6. *Maintain the required value of conservation areas:*
   - Set conservation goals at the level of individual conservation areas. Ideally these goals will acknowledge the particular values of the area in the context of the whole system.
   - Implement management actions and zonings in and around each conservation area to achieve these goals.
   - Monitor key indicators that will reflect the success of management actions or zonings in achieving goals, modifying management as required.

---

[a] Margules & Sarkar (2007) have produced a more detailed systematic conservation planning framework in which the original six stages have been divided into thirteen stages.

However, even though systematic conservation planning is now widely considered the most effective approach for designing protected area networks (Kareiva & Marvier 2012), this has not resulted in its widespread uptake by practitioners (Prendergast *et al.,* 1999; Knight *et al.,* 2006a). Since its origin in the early 1980s the use of this approach has largely been restricted to Australia, South Africa and North America (Pressey 1999; Groves *et al.,* 2002; Balmford 2003; Pressey *et al.,* 2007) where it has helped shape environmental policy and conservation planning on the ground (e.g. Cowling & Pressey 2003; Cowling *et al.,* 2003a; Cowling *et al.,* 2003b; Leslie *et al.,* 2003; Fernandes *et al.,* 2005; Rouget *et al.,* 2006; Smith *et al.,* 2006; Smith *et al.,* 2008). This approach has therefore generated a significant amount of literature, often referred to as 'best practice' on how protected area networks can be designed to reduce their impact on stakeholders, increase the likelihood of implementation, and ensure the long term persistence of biodiversity (Knight *et al.,* 2006a; Knight *et al.,* 2006b; Knight *et al.,* 2008). It has also provided conservation practitioners with well tested tools (e.g. area selection algorithms) and principles (e.g. replication, representation, and complementarity) that can be used to generate data to support and inform conservation planning efforts (Cowling *et al.,* 2004; Knight *et al.,* 2006b).

1.5 THESIS AIMS AND OBJECTIVES

By setting such time-specific targets, there is an increasing emphasis on national governments to establish MPA networks to achieve a broad range of marine conservation and management objectives. However, work still needs to be done to increase the global coverage and effectiveness of MPAs. Such trends are reflected in Europe, where there is growing interest in designating MPAs in the waters of Member States as part of fulfilling international obligations and regional commitments to the European Birds and Habitats Directives (EC 1979; EC 1992), and the Convention for the Protection of the Marine Environment of the North East Atlantic (OSPAR 2003b). However, whilst no one-size-fits-all approach exists for establishing MPAs networks, research over the last 20 years has provided a greater understanding of how different variables can potentially influence MPA design. This knowledge has resulted in ecological 'rules of thumb' for designing MPA networks (e.g. Airame *et al.,* 2003; Roberts *et al.,* 2003a; IUCN-WCPA 2008; Roberts *et al.,* 2010).

Nevertheless, whilst the field of MPA design has evolved significantly over the last two decades (Gaines *et al.,* 2010a) the designation of MPAs in Europe is still largely based on expert opinion. Moreover, recent research has shown there is a growing need to adopt more flexible and transparent approaches to MPA network design in Europe (Fenberg *et al.,* 2012; Giakoumi *et al.,* 2012), only then can science inform and influence policy more effectively (Torpova *et al.,* 2010; Fox *et al.,* 2012a). Consequently, there is a need to refine existing approaches and demonstrate how systematic conservation planning can be used to inform MPA design in European waters, and show how this approach can fit with existing policy frameworks (Rees *et al.,* 2013). Therefore, in this thesis I bring together a range of biological, legal and socio-economic data to address these issues.

Specific objectives contributing to this thesis's aims included:

- to review existing approaches used to guide the identification and selection of MPAs in Europe

- to demonstrate the value of systematic conservation planning and spatial prioritisation (decision support) tools to inform and support MPA network design in Europe

- to investigate the impacts of data quality on the species-area relationship approach used to develop habitat targets in conservation planning

- to investigate the biological and socio-economic impacts of using MPA size constraints when designing MPA networks

- to explore the potential role of spatial marine zoning to achieve conservation and sustainable fisheries management objectives in temperate marine ecosystems

- to determine whether no-take MPAs justify the cost of their implementation, or whether there are alternative forms of management that could achieve similar results

1.6 THESIS OUTLINE

This thesis is structured as follows:

This thesis begins with a review of how existing approaches used to guide the selection and designation of MPAs in Europe compare to best practice in contemporary conservation planning (**Chapter 2**). This review focuses on three key elements which are designed to identify MPA networks that achieve conservation goals, minimise impacts on stakeholders and increase the likelihood of implementation.

Chapter 3 focuses on the species-area relationship (SAR) based approach that is increasingly used to set conservation targets for habitat types when designing protected area networks. This approach has subsequently been adopted to develop targets for marine habitats in the UK, thus this chapter uses data from the Eastern English Channel to investigate how the SAR-based approach is influenced by underlying data quality.

Chapter 4 uses spatial prioritisation software to investigate the impact of using MPA size constraints when designing MPA networks in English waters, and focuses on how increasing the minimum size of MPAs influences: (i) the spatial characteristics of MPA networks; (ii) stakeholders, in terms of opportunity costs; and, (iii) connectivity for a range of species dispersal distances.

Chapter 5 uses two of the most widely adopted software tools in marine conservation planning 'Marxan' and 'Ecopath with Ecosim' to investigate the potential trade-offs associated with different spatially explicit MPA management strategies. This research focuses on the Eastern English Channel, and explores: (i) the potential role of spatial marine zoning to achieve conservation and sustainable fisheries management objectives; and, (ii) whether no-take MPAs justify the cost of their implementation, or whether MPA networks that have multiple zones with different management restrictions can achieve similar results.

Chapter 6 offers a discussion of the thesis's key findings and conclusions, and discusses useful avenues along which future research might proceed to better inform global and European marine conservation planning efforts.

CHAPTER 2

MARINE CONSERVATION SCIENCE AND GOVERNANCE IN NORTH-WEST EUROPE: CONSERVATION PLANNING AND INTERNATIONAL LAW AND POLICY *

Kristian Metcalfe[a], Thomas Roberts[a], Robert J. Smith[a] and Stuart R. Harrop[a]

[a] Durrell Institute of Conservation and Ecology (DICE), University of Kent, Canterbury, Kent, CT2 7NR, UK

* **Metcalfe, K.**, Roberts, T., Smith, R.J., and Harrop, S.R. (2013) Marine conservation science and governance in North-West Europe: conservation planning and international law and policy. *Marine Policy* 39(3), 289-295.

2.1 ABSTRACT

Member States of the European Union are increasingly designating marine protected areas (MPAs) to meet globally agreed marine protection targets and regional commitments. A number of studies have examined the impact of the associated European policy on the representation of species and habitats but we lack a comprehensive review of their combined impact on marine conservation in Europe. Here we use a systematic conservation planning framework to conduct such a review and compare the existing legislation to three elements of best practice, which are designed to identify MPA networks that achieve conservation goals whilst increasing the likelihood of implementation. In particular, we investigate the extent to which legislation: (i) translates broad policy goals into explicit targets; (ii) incorporates socio-economic data into the planning process; and (iii) requires a social assessment. Whilst this legislation has widespread political support and has underpinned the rapid expansion of MPA networks, we show it largely fails to incorporate these key components from systematic conservation planning. Therefore, if European approaches to marine conservation are to fulfil their goal of halting marine biodiversity loss, it is essential they link existing policy frameworks with transparent strategies that account for local conditions and support implementation.

Keywords: *Birds and Habitats Directive, CBD, Marine protected areas, NATURA 2000, OSPAR, Systematic conservation planning*

2.2 INTRODUCTION

There is international agreement on the need for increased protection of the world's oceans because of rapid declines in the health of many marine ecosystems (Lubchenco *et al.,* 2003). However, protected area (PA) coverage in the marine realm is relatively low, with only 1.17% of the ocean's surface designated as marine protected areas (MPAs), in contrast to 12.7% of terrestrial areas (CBD 2010a; Fox *et al.,* 2012b). In response, many governments have agreed to establish or expand existing MPA networks within their marine jurisdictions to meet globally agreed marine protection targets (Wood *et al.,* 2008; Wood 2011), such as the Convention on Biological Diversity's (CBD) '*Aichi Target*', which recommends that by 2020 10% of marine and coastal areas should be covered by MPAs (CBD 2011; Harrop 2011). This interest in establishing MPAs is also reflected in the European Union (EU), where MPAs are increasingly seen as important spatial management tools to address a broad array of management goals, such as biodiversity conservation and sustainable fisheries (Smith *et al.,* 2009).

Implementing a network of MPAs in Europe however, is likely to be challenging because approaches that govern marine conservation are often developed at both the European and national level (Haslett *et al.,* 2010). Consequently, recent work has called for research to address knowledge gaps about the factors influencing the success of European MPAs (Fenberg *et al.,* 2012). Thus, whilst many commentators have examined the impact of European law and policy on the representation of species and habitats (Dimitrakopoulos *et al.,* 2004; Jackson *et al.,* 2004; Maiorano *et al.,* 2007; Sundblad *et al.,* 2011), and there is a growing body of evidence on the effectiveness of MPAs in Europe (Fenberg *et al.,* 2012), we lack a comprehensive review of the combined impacts of marine conservation policy in Europe. Here, we address this gap by: (i) reviewing the extensive body of marine conservation planning legislation in Northern Europe, defined as the North East Atlantic (**Figure 2.1**); and, (ii) identifying problems with the existing approaches used to guide the selection and designation of MPAs. As part of this we highlight several key components of best practice from conservation planning science and propose how existing measures should be adapted to include such elements.

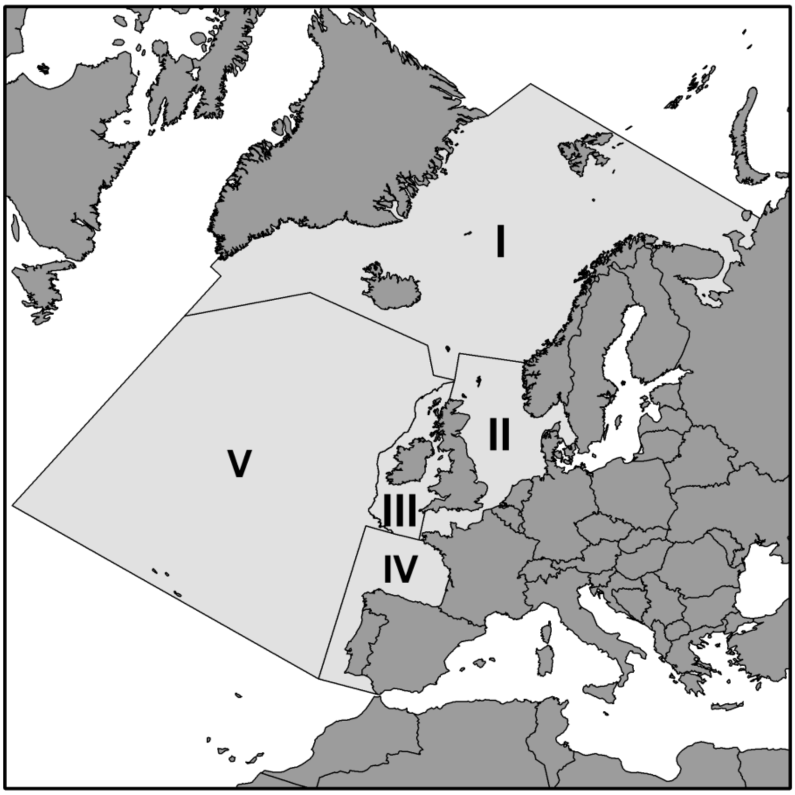

Figure 2.1 North East Atlantic as defined by the OSPAR Commission. Regions are defined as follows: (I) Arctic Waters; (II) Greater North Sea; (III) Celtic Seas; (IV) Bay of Biscay and Iberian Coast; and, (V) Wider Atlantic.

2.3 CURRENT CONSENSUS ON BEST PRACTICE IN CONSERVATION PLANNING

It is generally agreed in the scientific literature that the best approach for designing PA networks is systematic conservation planning (Margules & Pressey 2000). This approach is designed to identify priority areas for conservation that ensure the representation and persistence of biodiversity, whilst minimising impacts on stakeholders and increasing the likelihood of implementation (Knight *et al.,* 2006a; Knight *et al.,* 2006b). Systematic conservation planning is a process that combines a short-term conservation assessment, which identifies priority areas for conservation management, together with a long-term implementation framework that is used to achieve conservation action (Knight *et al.,* 2006a). This approach has been widely used in both the terrestrial and marine realms and this is partly because it avoids being overly prescriptive. However, there are three key aspects that underpin the success and flexibility of this approach and these are described subsequently.

2.3.1 Compile a list of broad-goals and set quantitative targets

Systematic conservation planning involves translating the broad-goals of the planning process into explicit and measurable objectives. This generally involves: (i) compiling a list of conservation features, such as important species, habitats and ecological processes, based on legislation or expert opinion, and (ii) setting quantitative targets for the minimum amount of each feature intended for protection (Pressey *et al.*, 2003; Carwardine *et al.*, 2009). There has been substantial debate about target-based conservation planning but there are two broad reasons why it is generally seen as best practice (Carwardine *et al.*, 2009). First, it allows policy makers to measure how well existing PA networks meet these targets and makes it less likely that conservation features with high economic value are under-represented (Pressey *et al.*, 2003). Second, it provides a clear purpose for conservation decisions, lending them accountability and scientific defensibility and so makes them less open to direct or unconscious political interference (Cowling *et al.*, 2003b; Pressey *et al.*, 2003). This transparency helps build stakeholder support and also provides a platform for discussing trade-offs between different groups.

2.3.2 Incorporate socio-economic data

Another advantage of setting targets is that it allows the incorporation of socio-economic data into the planning process without compromising conservation goals, as the process is based on meeting targets for every feature, even when there is no alternative but to select costly areas. In contrast, priority setting without targets creates an incentive to avoid areas that are deemed too costly to protect, regardless of their conservation value (Margules & Pressey 2000). Including socio-economic data facilitates the development of conservation plans that: (i) minimise impacts on stakeholders, and so reduce conflict between conservationists and resource users (Klein *et al.*, 2008a; Klein *et al.*, 2008b); (ii) are more cost effective to implement and manage (Naidoo *et al.*, 2006; Carwardine *et al.*, 2008); (iii) can influence policy by highlighting trade-offs between achieving higher levels of a feature target and the increase in cost to obtain it (Naidoo *et al.,* 2006), and; (iv) account for conservation opportunity and constraint data and so increase the likelihood of implementation (Nhancale & Smith 2011). There are a number of types of conservation costs that can be included in the planning process, such as: acquisition, management, opportunity, transaction and damage (Naidoo *et al.,* 2006), although opportunity costs (the foregone revenues to stakeholders) are commonly used to influence the location of MPAs (Ban & Klein 2009).

2.3.3 Conduct a social assessment

Much of the early literature on systematic conservation planning focused on analysing biological data, but it is now widely accepted that it is vital to also conduct a social assessment (Knight *et al.,* 2006a), which involves incorporating socio-economic, social and policy-based information in the planning process (Cowling *et al.,* 2010). Thus, in order to facilitate the translation of priority areas and goals into conservation action it is essential to undertake a well-resourced social assessment that gathers the relevant non-biological data (Cowling & Wilhelm-Rechmann 2007). This must involve identifying and working with the relevant stakeholders and implementing agencies to develop a better understanding of impacts, such as the opportunities and constraints associated with each type of conservation intervention (Knight *et al.,* 2006b; Cowling & Wilhelm-Rechmann 2007). This information can then be used to inform the conservation assessment, by setting targets that reflect both biological, social, and economic requirements and adjusting costs to preferentially select areas where stakeholder support is most likely (Knight *et al.,* 2006b; Cowling & Wilhelm-Rechmann 2007; Jones 2012). However, it should be recognised that the designation of some priority areas will never have full stakeholder support. Thus, this information should also be used to minimise conflict and inform the implementation strategy by identifying how priority areas should be managed in ways that foster support and fit within existing policy frameworks (Knight *et al.,* 2006b; Jones 2012).

2.4 INTERNATIONAL AND EUROPEAN MARINE CONSERVATION POLICY

There are a number of 'peripheral' legal obligations and non-binding provisions that influence biodiversity conservation in Europe (EC 2002b; EC 2008b), which include the following: Convention on Wetlands of International Importance (Ramsar); Convention on the Conservation of European Wildlife and Natural Habitats (Bern); Convention on the Conservation of Migratory Species of Wild Animals (Bonn); World Summit on Sustainable Development (WSSD); the Protected Areas Programme of the World Conservation Union (IUCN 2000); and the Marine Strategy Framework Directive (EC 2008b). However, the main policy instruments that govern the conservation of marine biodiversity and the selection and designation of MPAs in Northern Europe are: (i) the Convention on Biological Diversity; (ii) the European Birds and Habitats Directives; and (iii) the Convention for the Protection of the Marine Environment of the North East Atlantic, which are summarised and compared to best practice.

2.4.1 Convention on Biological Diversity

2.4.1.1 Marine policy relevance

The EU's Member States are Contracting Parties to the Convention on Biological Diversity (CBD), which states in Article 8(a) that: "*each contracting party shall as far as possible and as appropriate establish a system of protected areas or areas where special measures need to be taken to conserve biological diversity"* (CBD 1992). Such PAs are defined in Article 2 as: "*a geographically defined area which is designated or regulated and managed to achieve specific conservation objectives"* (CBD 1992). The establishment of a representative global network of MPAs was initially proposed at the 7th Conference of the Parties (COP) to the CBD (Wood 2011) where it was agreed that the goal of the Programme of Work on Protected Areas, and on Marine and Coastal Biological Diversity should be: "*the establishment and maintenance of marine and coastal protected areas that are effectively managed, ecologically based and contribute to a global network of marine and coastal protected areas, building upon national and regional systems, including a range of levels of protection, where human activities are managed, particularly through national legislation, regional programmes and policies, traditional and cultural practices* *and international agreements, to maintain the structure and functioning of the full range of marine and coastal ecosystems"* (CBD 2004b; CBD 2004c), echoing commitments made at the WSSD and 5th World Parks Congress (United Nations 2002; IUCN 2003).

This goal was further reinforced with the formulation of the 20 time-bound Aichi targets that were negotiated within the CBD's new Strategic Plan for Biodiversity at the 10th COP (CBD 2010b). In the context of MPAs, Aichi Target 11 urges that: "*by 2020*, at least *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"* (CBD 2010b).

2.4.1.2 Critique

In terms of developing a list of broad-goals and translating these into targets, the Programme of Work on Protected Areas states Contracting Parties should aim to achieve 10% protection of their coastal and marine areas by 2020, and this should involve developing feature-specific targets that reflect their national and regional priorities (CBD 2004b). These targets act as an important foundation (Metcalfe *et al.*, 2013a) but this programme has neither prescribed any subsidiary legal requirements to the generality of Article 8(a), nor established any explicit goals or targets defining what these systems should aim to achieve (Gaston *et al.*, 2008). Moreover, whilst the Aichi Targets address marine protection in substantially more detail, they have been criticised because these targets: (i) do not resolve how Contracting Parties and regions such as the EU will work together to achieve these goals; and (ii) are not legally binding since the CBD merely 'urges' Contracting Parties to fulfil them (Harrop 2011; Harrop & Pritchard 2011).

With regards to incorporating socio-economic data, the Programme of Work on Protected Areas states that Contracting Parties should: *"use relevant socio-economic data required to develop effective planning processes"* to substantially improve site-based protected area planning and management (CBD 2004c). However, this is only a 'suggested' activity as Contracting Parties are only: *"encouraged to pay due regard to the social, economic and environmental costs and benefits of various options"* (CBD 2004c). Thus, there are no clear requirements to incorporate these data into the planning process. In fact a greater emphasis is placed on collecting data on: (i) the socio-economic value of marine ecosystems, and the cost of their continuing decline; and (ii) the establishment and maintenance cost of managing protected areas (CBD 2004b; CBD 2004c).

In contrast, the Programme of Work on Protected Areas does clearly state that to improve site-based PA planning and management that "*all protected areas"* should be developed using: *"participatory and science-based site planning processes that incorporate clear biodiversity objectives, targets, management strategies and monitoring programmes, drawing upon existing methodologies and a long-term management plan with active stakeholder involvement"* (CBD 2004c). Whilst this implies some aspects of best practice from conservation planning science, and clearly highlights that PA design and management should involve collaboration with relevant stakeholders, once again there is no requirement to incorporate this into national policies that govern PA selection and designation (CBD 2004c).

Nonetheless, despite its voluntary nature the EU has declared its commitment to integrate the CBD's Strategic Plan for Biodiversity and its time-bound Aichi targets into: *"all relevant EU sectors and policies and to implement them, including through the future EU Biodiversity Strategy"* (EC 2010). However, the EU Biodiversity Strategy, a policy document developed to support these objective, only refers to MPAs as a tool for supporting sustainable fisheries, and makes no explicit reference to achieving Aichi Target 11 (EC 2011).

2.4.2 European Birds and Habitats Directives

2.4.2.1 Marine policy relevance

The European Birds and Habitats Directives are two of the EU's principal and most comprehensive instruments of conservation strategy that are legally binding on Member States. The Birds

Directive (*Council Directive 79/409/EEC)*, though primarily concerned with avian conservation, requires the designation of Special Protection Areas (SPAs) to: *"maintain endangered, vulnerable, and migratory species of conservation concern across their natural range"* (EC 1979). The principal goal of the Habitats Directive *(Council Directive 92/43/EEC)* is the conservation of natural habitats and of wild fauna and flora (EC 1992), and requires the designation of Special Areas of Conservation (SACs) defined as the most appropriate areas to: "*maintain or restore, natural habitats, plant and animal species of conservation concern to a favourable conservation status across their natural range"* (EC 1992).

The selection of SACs is described in Annex III of the Habitats Directive and is based exclusively on scientific criteria, such as: (i) the degree of representativity, ecological quality and area for habitat types; and (ii) the size, density of populations, and the degree of their isolation for species (EC 1992; EC 2002a; EC 2007). In contrast, there are no agreed EU criteria for the selection and designation of SPAs, although, many countries use the criteria based on the Ramsar 1% flyway population (Evans 2012). In combination these sites form the Natura 2000 network, which is described as an ecologically coherent community wide-network of PAs covering terrestrial and marine ecosystems (EC 1992; EC 2002a), and each EU state must contribute to Natura 2000 "*in proportion to the representation within its territory of the natural habitat types and the habitats of the species detailed in the Directive's Annexes"* (EC 1992).

2.4.2.2 Critique

Although the EU Birds and Habitats Directives contain a list of conservation features that are considered appropriate subjects for conservation interventions[1], and were established with extensive national and political input, they have not been re-evaluated since 2007 (Evans 2012).

[1] The EC Birds Directive requires the designation of SPAs for rare and/or threatened species (192 species or sub-species as listed in Annex I of the Directive), together with sites which that are important for regularly occurring migratory species. The EC Habitats Directive includes measures for the strict protection of species listed in Annex IV, and requires the designation of SACs that will make a contribution to conserving the 189 habitat types and 788 species identified in Annexes I and II of the Directive (as amended in the consolidated version 1.1 of the Habitats Directive in 2007).

This has resulted in several problems, as it fails to take into account: (i) how species and habitat conservation status has changed with the expansion of the EU (Cogalniceanu & Cogalniceanu 2010); (ii) new data on the importance of marine species and habitats (EC 2007; Apostolopoulou & Pantis 2009; van Haastrecht & Toonen 2011); and (iii) changing risks from climate change and other factors (Harrop 1999; Giakoumi *et al.,* 2012). In addition, there is no formal agreement or coordinated attempt to establish which other biodiversity features should be represented (Dimitrakopoulos *et al.,* 2004; Smith *et al.,* 2009), particularly areas that are important for marine ecosystem functioning such as spawning and aggregation sites. This may neglect areas that support key ecological processes that are difficult to define spatially such as migratory routes (De Santo & Jones 2007), arguably failing to implement Bonn Convention obligations. In addition, the Habitats Directives has been criticised as ill-suited for marine conservation because it was originally designed for terrestrial use and then initially only applied to inshore areas (De Santo & Jones 2007; Evans 2012).

In addition, compared to the detail that prescribes what should be protected, there has been little consideration of targets specifying how much of each feature should be conserved. Thus, existing approaches for designating SPAs and SACs are at the discretion of Member States, and have varied substantially as a result (EC 2002a; EC 2007; Gaston *et al.,* 2008). Selection has almost always focused on the properties of individual sites, such as the presence of target species and habitats (Ioja *et al.,* 2010). The only mention of targets in the Habitats Directive relates only to whether a nominated site should be proposed as a SAC, so that sites containing 60% of a feature should automatically be proposed, whereas sites containing 20% of a feature need further assessment before being considered for proposal (EC 1997). However, these figures have often been misunderstood to mean that between 20% and 60% of a species population or habitat area should be protected (EC 2007; Evans 2012). Thus, although the Directives oblige Member States to ensure each site achieves ‘favourable conservation status’, they provide no guidance on how much of each feature should be conserved in a PA network. This makes it difficult to determine: (i) how close the marine component of Natura 2000 is to being complete; (ii) what protection shortfalls need to be resolved through conservation planning; and (iii) how well this network will perform in the future (Dimitrakopoulos *et al.,* 2004; Maiorano *et al.,* 2007; Gaston *et al.,* 2008).

Article 2 of the Habitats Directive does, however, state that conservation measures shall take account of: "*economic, social and cultural requirements and regional and local characteristics"* (EC 1992), although their inclusion is often limited because European guidelines require that Member States should 'only' employ scientific and ecological criteria in the selection and designation of sites (EC 2007; van Haastrecht & Toonen 2011). The guidelines do require Member States to identify how different stakeholders interact with the species and habitats targeted for protection. However, this is primarily concerned with environmental impact assessments and identifying the negative impacts of activities, rather than documenting where stakeholders may support conservation (EC 2007). Furthermore, the level of stakeholder participation is often restricted to what has been described as 'consultative' (Borrini-Feyerabend 1999), so whilst stakeholders are encouraged to be involved in implementation and management, they lack powers to influence where a site is designated or how specific features are protected (Jones 1999; Jones 2012). This is in line with other approaches to conservation planning in Europe which specifies that socio-economic data and stakeholder involvement should not guide the selection of PAs (EC 2002a).

This has given rise to problems in some Member States, such as: (i) disagreements about the scope of stakeholders influence over designated areas; (ii) increased conflict at various stages of the planning and implementation process, particularly the designation of site boundaries; and (iii) a lack of local acceptance, and confusion surrounding the protection statuses (i.e. overlap among national, EU and IUCN statuses) of existing and new PAs (Dimitrakopoulos *et al.,* 2004; Apostolopoulou & Pantis 2009; Roberts & Jones 2009; Ioja *et al.,* 2010; Grodzinska-Jurczak & Cent 2011). Therefore, given limited conservation resources, the present approach to identifying PAs has often generated unwanted economic impacts and increased social tensions rather than foster support for conservation (Apostolopoulou & Pantis 2009; Grodzinska-Jurczak & Cent 2011).

2.4.3 Convention for the Protection of the Marine Environment of the North East Atlantic

2.4.3.1 Marine policy relevance

The Convention for the Protection of the Marine Environment of the North East Atlantic (OSPAR) is designed to regulate marine activities across a number of EU Member States marine jurisdictions (**Figure 2.1**). This includes territorial waters, exclusive economic zones (EEZs) and areas beyond national jurisdiction (OSPAR 1992), and can be interpreted as legally binding for the Governments of the 15 Contracting Parties and EU Member States through the effect of the EU being a direct signatory (De Santo & Jones 2007).

The convention's primary emphasis was on anti-dumping and pollution measures (OSPAR 1992), but it now includes explicit references to marine conservation planning, which include obligations in Article 2(1)(a) to: "*conserve marine ecosystems and, when practicable, restore marine areas"* (OSPAR 1992). In addition, OSPAR has issued several relevant binding and non-binding provisions with regard to MPAs through its Biological Diversity and Ecosystems Strategy, which are directed at: (i) "*conserving species, habitats and ecological processes which have been adversely affected by human activities"*; and (ii) "*protection of areas that best represent the range of species habitats and ecological processes"* in the OSPAR maritime area (OSPAR 2010b). Furthermore, to complement existing European measures OSPAR has developed a number of strategies for Contracting Parties to implement a joint network of "*well-managed"* MPAs, that together with the Natura 2000 network is "*ecologically coherent"* (OSPAR 2003a; OSPAR 2003b; OSPAR 2010b). In addition, though some differences exist in their text and geographical scope, OSPAR also operates a joint programme of work on MPAs with HELCOM, which is the Convention on the Protection of the Marine Environment of the Baltic Sea Area (OSPAR 2003b; Ardron 2008).

2.4.3.2 Critique

In order to address gaps in existing European measures, OSPAR has developed a list of conservation features in need of protection[2] (OSPAR 2008c) but this has primarily focused on offshore habitats and species, as existing efforts have generally been directed at protecting inshore territorial waters (De Santo & Jones 2007; van Haastrecht & Toonen 2011). This list seeks to complement, but not duplicate work under other international and European agreements (OSPAR 2008a), and forms part of the criteria in the guidelines used to reinforce the identification and selection of OSPAR MPAs (OSPAR 2003a; OSPAR 2006). However, even though OSPAR provides a framework for identifying suitable sites, there are no explicit or legally binding targets for what this network should aim to achieve (OSPAR 2006). Although, OSPAR does encourage Contracting Parties to develop a network that is consistent with existing international obligations, such as the CBD target that: *"at least 10% of each of the world's marine and coastal ecological regions"* should be conserved (CBD 2006; OSPAR 2010b). OSPAR also recommends that Contracting Parties should determine the proportion of each biodiversity feature to be included within this joint network using the best available data (OSPAR 2006), which is likely to be difficult given that: (i) there is no formal guidance on how to develop quantitative targets; (ii) data on many of the listed species, if available, are often mapped at too coarse a spatial and temporal scale; and (iii) there has been no coordinated attempt by EU Member States to develop a research agenda to address these data gaps (OSPAR 2006; OSPAR 2007; OSPAR 2008c).

In addition, given that EU Member States have different capacity levels and priorities (Cogalniceanu & Cogalniceanu 2010), they have often interpreted the Convention's broad-goals differently. This is because Member States have their own framework for the organisation of environmental policy (Haslett *et al.*, 2010), and so consequently, targets for features may be influenced by social and political acceptability. Such trends are already evident in European terrestrial conservation strategies, where protected areas are commonly placed at high elevations

[2]OSPAR has produced three documents since it was first ratified that identifies threatened and or declining species and habitats in the OSPAR maritime area that should be represented in MPAs; the latest version includes 16 habitats and 42 species (comprised of 5 invertebrates, 9 birds, 22 fish, 2 reptiles and 4 mammals).

and in areas of low population density and economic potential (Oldfield *et al.*, 2004; Maiorano *et al.*, 2007). Furthermore, given that the OSPAR selection criteria do not account for what is already conserved under Natura 2000, it is unlikely that the ecological goals of this 'joint network' will be met (OSPAR 2003a). This is especially because the majority of existing OSPAR MPAs are SACs and SPAs, so that 144 of the 159 OSPAR MPAs overlap with these existing Natura 2000 sites. Thus, the current network is failing to fulfil its goal of conserving offshore areas, as most Natura 2000 sites are located in inshore territorial waters[3] or are simply extensions of terrestrial sites (OSPAR 2007; OSPAR 2010a; Giakoumi *et al.*, 2012).

In addition, even though OSPAR explicitly states that conservation measures should consider: *"social and economic implications"* (OSPAR 1992), the guidelines for the identification and selection of MPAs make no reference of how to account for socio-economic data when identifying MPAs (OSPAR 2006). Though, in contrast to other European measures, OSPAR has developed guidance on how to incorporate relevant stakeholders, experts and organisations into the planning process (OSPAR 2008b). However, this guidance was only developed to ensure that Contracting Parties are aware of: (i) approaches to communicating with different types of stakeholders; and (ii) the benefits and challenges of stakeholder participation. Moreover, it also states that the selection and designation of sites is often a lengthy process and that stakeholder engagement should be assessed on a case by case basis (OSPAR 2008b). This further emphasises, as with other European measures, that stakeholder consultation about the nature of designated or proposed sites is often disregarded at the value of other stages in the planning process (Dimitrakopoulos *et al.*, 2010).

[3] In 2010 the OSPAR MPA network consisted of 159 sites (144 of which overlap with existing Natura 2000 sites) collectively covering 147,322 km^2, corresponding to 1.06% of the OSPAR maritime area. As the vast majority of sites have been designated in territorial waters overall coverage of coastal waters by OSPAR MPAs is 13.5%. In contrast, coverage of offshore areas i.e. exclusive economic zones is 0.57%. In addition, no MPA has yet been established in areas beyond national jurisdiction, which comprises 40% of the OSPAR maritime area.

2.5 DISCUSSION

2.5.1 Successes in current European law and policy

Whilst developing PAs in Europe has proven difficult, the European legislation described herein has significant political buy-in and widespread support (Grodzinska-Jurczak & Cent 2011). This is highlighted by the rapid expansion of PA networks such as Natura 2000 (EC 2011), which currently contains more than 26,000 sites covering 17.5% of the EU territory (Evans 2012). The EU also has the clear expertise and legal authority to effectively implement a network of transnational MPAs, which is demonstrated by the Habitats Directive being the first international instrument to address the protection of all habitats within the region (De Santo & Jones 2007).

Moreover, this European legislation has provided the first coherent framework for conservation planning at a national level in a number of countries, so there would probably be far less interest in designating PAs in Europe without such obligations (Gaston *et al.*, 2008). It is also likely that EU legislation has resulted in far better representation of important biodiversity features than could have been achieved by individual Member States acting alone. In addition, the broad goals identified in European legislation mean there is a great deal of scope for Member States to tailor their actions to local conditions. For example, the Marine and Coastal Access Act was developed by the UK government in response to their OSPAR commitments (MCAA 2009; JNCC & Natural England 2010). This Act resulted in the Marine Conservation Zone (MCZ) project which, to our knowledge, is the first such attempt in Europe to adopt principles of best practice from conservation planning. Thus, the initial recommendations for priority areas that form the basis of the UKs first comprehensive MPA network are based on achieving explicit quantitative targets, and involved significant stakeholder participation (Jones 2009; JNCC & Natural England 2010; Jones 2012).

However, it should be noted that the UK Government were not required to adopt this approach and voluntarily used aspects of best practice to underpin the MCZ project. Thus, current legislation makes it more likely that Member States will adopt less systematic approaches and so produce

MPA networks that fail to conserve marine biodiversity adequately or reduce negative impacts on stakeholders (Stewart *et al.*, 2003; Rabaut *et al.*, 2009).

2.5.2 Adopting key components of best practice from conservation planning

European legislation is currently failing to benefit from the lessons learnt in systematic conservation planning but there are opportunities for its application. This is because current measures adopt some aspects of best practice, such as compiling a list of important species and habitats of conservation concern. However, these aspects are not used as part of a coherent framework and are generally not applied in a transparent manner. Thus, there is an obvious need for change but any suggested amendments must account for the current legislative frameworks. This is why we think that amending the OSPAR legislation is most appropriate because it focuses on developing a network of MPAs, which is in contrast to the site-by-site approach of the Birds and Habitats Directives (Gaston *et al.*, 2008). In addition, such a role would be possible given that OSPAR's text and actions are legally binding on Member States through the effect of the EU being a signatory (De Santo & Jones 2007).

2.5.3 Adopting a more coordinated approach to conservation planning in Europe

One of the key issues with existing approaches to marine conservation planning in Europe is the lack of quantitative targets or framework to develop them. This has inevitably led to a lack of consistency between individual Member States and a failure to measure progress and adapt strategies based on changes in data and socio-economic conditions. Moreover, recent research has shown that if Europe was to adopt a target-based approach then Member States would require less money if they adopted a coordinated approach, rather than identifying priorities in isolation (Kark *et al.*, 2009). Therefore, a more transparent and coordinated strategy within Europe would allow the development of more sophisticated planning that accounts for socio-economic data, resulting in increased representation of biodiversity and cost-efficiency (Bladt *et al.*, 2009; Kark *et al.*, 2009). Such a target-based approach could be particularly important in the EU, as it would allow better consideration of the trade-offs involved in exploiting and developing 'shared' marine resources and conserving biodiversity.

2.6 CONCLUSION

Marine conservation planning in Europe is often seen as a balancing act between socio-economic and political interests and the need to improve the status of the marine environment (van Haastrecht & Toonen 2011). Despite this trade-off, existing approaches have resulted in the rapid expansion of PA networks across Member States, underlining the EU's ability to implement a network of transnational MPAs. However, existing legislation neglects several key components of best practice from conservation planning, which is likely to prevent the achievement of the EU's broad conservation goals. Moreover, given that every Member State is committed to developing MPA networks, policy makers and practitioners should see these shortcomings as critically important. This is because a failure to adopt best practice will result in wasted resources, increased stakeholder conflict and lost opportunities (Agardy *et al.*, 2011). Therefore, if European approaches to marine conservation are to fulfil their original goals, it is essential that they link existing EU objectives with implementation strategies that account for local conditions and facilitate appropriate conservation action.

2.7 ACKNOWLEDGEMENTS

We would like to thank Ingrid Chudleigh of Natural England and Sue Wells of Balanced Seas and Natural England for informative discussions and useful advice, and are also grateful to two anonymous referees for constructive comments that improved earlier versions of this manuscript.

CHAPTER 3

INVESTIGATING THE IMPACTS OF DATA QUALITY ON THE SETTING OF CONSERVATION PLANNING TARGETS USING THE SPECIES-AREA RELATIONSHIP *

Kristian Metcalfe[a], Juliette Delavenne[b], Clément Garcia[c], Aurélie Foveau[d], Jean-Claude Dauvin[e], Roger Coggan[c], Sandrine Vaz[b], Stuart R. Harrop[a], and Robert J. Smith[a]

[a] Durrell Institute of Conservation and Ecology (DICE), University of Kent, Canterbury, Kent, CT2 7NR, UK

[b] Institut Français de Recherche pour l'exploitation de la Mer (IFREMER), Laboratoire Ressources Halieutiques, 150 quai Gambetta, BP699, 62321 Boulogne-sur-Mer, France

[c] The Centre for Environment, Fisheries and Aquaculture Science (CEFAS), Pakefield Road, Lowestoft, NR33 0HT, UK

[d] Institut Français de Recherche pour l'exploitation de la Mer (IFREMER), Laboratoire Environnement Littoral et Ressources Aquacoles Finistère-Bretagne Nord, CRESCO, 38 Rue du Port Blanc, BP 80108, 35801 Dinard, France

[e] Université de Caen Basse Normandie, Laboratoire Morphodynamique Continentale et Côtière, UMR CNRS 6143 M2C, 2-4 rue des Tilleuls, F-14000 Caen, France

* **Metcalfe, K.**, Delavenne, J., Garcia, C., Foveau, A., Dauvin, J. C., Coggan, R., Vaz, S., Harrop, S. R. & Smith, R. J. (2013) Impacts of data quality on the setting of conservation planning targets using the species-area relationship. *Diversity and Distributions*, 19(1), 1-13.

3.1 ABSTRACT

The species-area relationship (SAR) is increasingly being used to set conservation targets for habitat types when designing protected area networks. This approach is transparent and scientifically defensible but there has been little research on how it is affected by data quality. Here we used a macrobenthic dataset from the eastern English Channel containing 1314 sampling points, and assigned each point to its associated habitat type based on a detailed habitat map. We then used the SAR based approach for setting targets and tested whether this was influenced by changes in: (i) the number of sampling points used to generate estimates of total species richness for each habitat type; (ii) the non-parametric estimator used to calculate species richness; and, (iii) the level of habitat classification employed. We then compared our results with targets from a similar national-level study that is currently being used to identify Marine Conservation Zones in the UK. We found that habitats targets were strongly influenced by all of the tested factors. However, sample size had the greatest impact, with specific habitat targets increasing by up to 45% when sample size increased from 50 to 300. We also found that results based on the Bootstrap estimator of species richness, which is the most widely used for setting targets, were more influenced by sample size than the other tested estimators. Finally, we found that targets were higher when using broader habitat classification levels or a larger study region. However, this is also likely to be a sample size effect because these larger habitat areas generally contained more sampling points. Therefore, whilst setting habitat targets using best-available data should play a key role in conservation planning, further research is needed to develop methods that better account for sampling effort.

Keywords: *English Channel, Habitat Targets, Marine Conservation Zones, Marine Protected Areas, Species-area relationship, Systematic Conservation Planning*

3.2 INTRODUCTION

Marine and coastal ecosystems are under increasing pressure from a diverse range of threats including the over-exploitation of natural resources (particularly over-fishing), pollution, and climate change (Lubchenco *et al.,* 2003). One response to these threats is to develop marine protected areas (MPAs), which are seen as increasingly important spatial management tools for conserving marine biodiversity (Wood *et al.,* 2008), maintaining large scale ecological processes (Roberts *et al.,* 2005) and supporting the sustainable use of marine resources (Spalding *et al.,* 2008). A widely used approach for helping to ensure that new MPAs achieve these goals is systematic conservation planning, which seeks to identify representative and viable networks of MPAs that also minimise costs (Margules & Pressey 2000). Thus, systematic conservation planning can be used to design MPA networks that reduce impact on stakeholders (Smith *et al.,* 2009), increase the likelihood of implementation, and help ensure long-term biodiversity persistence (Knight *et al.,* 2006b).

A key step in systematic conservation planning involves producing a list of important species, habitats and ecological processes, known collectively as "conservation features", and then setting quantitative targets for the minimum amount of each feature intended for conservation (Knight *et al.,* 2006b; Carwardine *et al.,* 2009). These targets can then be used by several conservation planning software packages (e.g. Marxan, C-Plan and Zonation) to help identify priority areas for protection (Ball *et al.,* 2009). Setting such targets provide a clear basis for conservation decisions, lending them accountability and defensibility, and ensures that the conservation planning process is more transparent, open to stakeholder involvement and less likely to be affected by political interference (Cowling *et al.,* 2003b). Approaches to target setting depend on the type of conservation feature of interest (Noss 1987). Targets for species are often set using relatively well established techniques based on population viability estimates (Rondinini *et al.,* 2006; Justus *et al.,* 2008; Rondinini & Chiozza 2010). In contrast, target-setting approaches for coarse-filter conservation features, such as habitat and vegetation types, are frequently based on expert opinion (e.g. Cowling *et al.,* 2003a; Pressey *et al.,* 2003; Smith *et al.,* 2006) or policy-driven targets such as those specified in the Convention on Biological Diversity (CBD), which currently recommends that 10% of coastal and marine areas under national jurisdiction should be protected by 2020 (CBD 2011). However, both expert-based and policy-driven targets have been widely criticised for a lack ecological credibility (see review by Carwardine *et al.,* 2009), so there is a real need for data-driven and scientifically defensible approaches for setting habitat targets.

In response to this problem, researchers developed an approach based on using field survey data to model the species-area relationship (SAR) for each important habitat type, which is then used to estimate the proportion of habitat area required to represent a user-specified percentage of species, and can be multiplied by the extent of the habitat type to produce a target area (Desmet & Cowling 2004; Reyers *et al.,* 2007). This methodology was subsequently adopted by the South Africa National Biodiversity Institute (SANBI) to calculate targets for each vegetation type listed in the national vegetation classification system (Rouget *et al.,* 2004). These targets were then used to help identify priority conservation areas (Rouget *et al.,* 2006; Smith *et al.,* 2008; Gallo *et al.,* 2009) and conduct threatened vegetation type assessments as part of South Africa’s first National Spatial Biodiversity Assessment (Nel *et al.,* 2007; Reyers *et al.,* 2007), helping to ensure a level of consistency between projects and regions.

The success of this approach means that SAR-based targets are beginning to be developed elsewhere. In particular, they have been used to set national marine habitat targets as part of four regional projects funded by the UK Government, which seek to establish a network of Marine Conservation Zones (MCZs) in English territorial waters (JNCC & Natural England 2010; Rondinini 2011a). With increasing adoption, it is important that conservation planners and practitioners have confidence in this approach to target setting, as targets have a large influence on the final extent of any protected area (PA) network (Vimal *et al.,* 2011; Delavenne *et al.,* 2012) and any subsequent socio-economic impacts (Chittaro *et al.,* 2010; Mascia *et al.,* 2010; McCrea-Strub *et al.,* 2011). However, despite their growing use, there is still uncertainty about how this target setting process is affected by data constraints, as the SAR is known to be influenced by biogeographic patterns, model parameters, model type, and data quality (Chiarucci *et al.*, 2003; Walther & Moore 2005; Hortal *et al.*, 2006). Here we investigate these issues using a macrobenthic dataset from the eastern English Channel, examining how targets are affected by the number of sampling points used to model the SAR, the choice of estimator used to calculate total species richness in each habitat type, and the level of habitat classification employed. We then compare these results developed at a regional level with those developed for the MCZ project at a national level, and calculate how using these different sets of targets would influence the extent of any resulting MPA network in the English Channel.

3.3 METHODS

3.3.1 Study area

This study was carried out in the English Channel (**Figure 3.1**), a cold-temperate epicontinental sea separating the south coast of the United Kingdom from the North coast of France (Delavenne *et al.*, 2012). The English Channel constitutes a bio-geographical transition zone between the warm temperate Atlantic oceanic system, and the boreal North and Baltic Sea continental systems of northern Europe, encompassing a wide range of ecological conditions (Coggan & Diesing 2011; Delavenne *et al.*, 2012). The study region focused on the eastern English Channel (EEC), which is delimited by the Dover Strait to the east and Cotentin Peninsula to the west and is a key area for tourism, shipping, energy production and aggregate extraction (Carpentier *et al.*, 2009). In addition, it supports an important commercial fishery, as well as key nursery, spawning areas and migratory routes linked to specific environmental characteristics (Martin *et al.*, 2009).

There are several ongoing MPA designation projects in this section of the English Channel. Both France and the UK have implemented MPAs as part of their EU Birds and Habitats Directive commitments and France is currently developing a MPA network in the “Three Estuaries region” (Bay of Somme, Authie, and Canche; **Figure 3.1**). In addition, the EEC is the focus of the Balanced Seas project (http://www.balancedseas.org/), which is one of four regional MCZ projects which seeks to identify and recommend MPAs for the inshore and offshore waters of south-east

England (JNCC & Natural England 2010). Balanced Seas uses habitat targets based on the SAR that were developed at a national level from biodiversity data collected in English waters (JNCC & Natural England 2010).

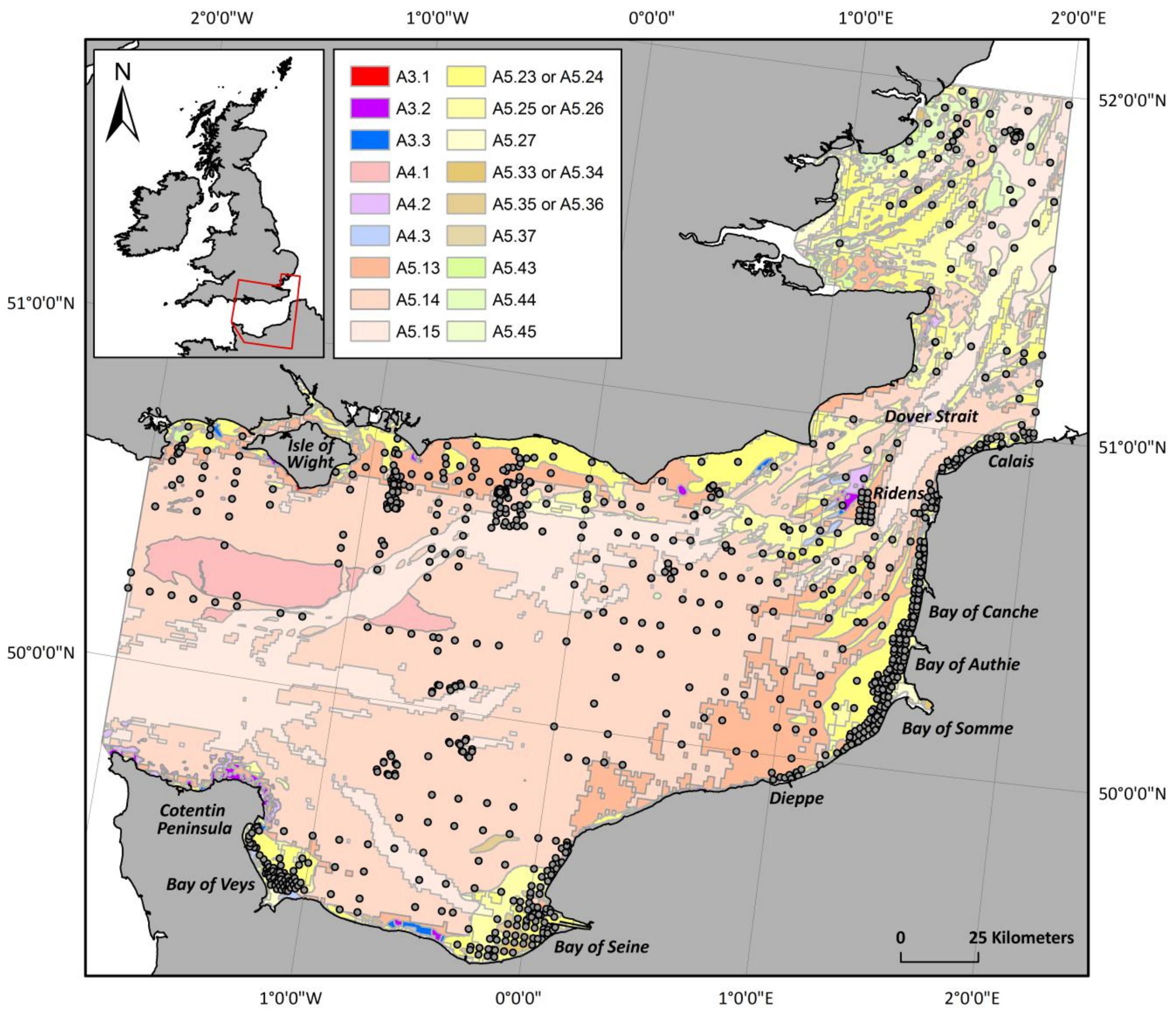

Figure 3.1 EUNIS level 3 and 4 habitat map for the eastern English Channel showing the location of the 1314 sampling points. See **Table S3.1** for a key to EUNIS habitat codes, levels and descriptions.

3.3.2 Habitat map

We used a broad-scale habitat map in this analysis, which is based on the European Nature Information System (EUNIS) habitat classification hierarchy developed by the European Environment Agency (EEA 2006; Coggan & Diesing 2011). **Figure 3.1** shows the distribution of each EUNIS habitat class that was modelled using physical and environmental data such as depth, substratum and energy levels; with rock and sediment habitats modelled to level 3 and 4 in the EUNIS hierarchy respectively (Coggan & Diesing 2011). The EUNIS level 3 habitats are broken down into three habitat types and coded as follows: infralittoral rock (A3.x), circalittoral rock (A4.x), and sublittoral coarse sediment (A5.x), which was further divided into its finer-scale EUNIS level 4 habitats (A5.xx).

3.3.3 Biodiversity survey data

Given the importance of macrobenthic diversity in the EEC (Vaz *et al.*, 2007; Carpentier *et al.*, 2009), the increasing emphasis on their conservation (Sanvicente-Anorve *et al.*, 2002; Vincent *et al.*, 2004) and the large amount of benthic sampling that has taken place in the EEC (e.g. Desroy *et al.*, 2003; Dauvin *et al.*, 2004; Carpentier *et al.*, 2009), we developed targets using presence/absence data from macrobenthic surveys carried out between 1985 – 2007, providing data from 1314 sampling points (**Figure 3.1**). These surveys used a range of sampling protocols and gear sizes (0.1m^2 to 0.5m^2), with samples predominantly collected using a Hamon grab, with the exception of 16 stations in the Ridens that used a van Veen grab. The sampling strategy in the study area was predominantly regularly spaced; however, there was more intensive sampling in surveys from the east of the Isle of Wight, in the Ridens and in coastal areas such as between Dieppe and Calais, the Bay of Veys, and the Bay of Seine (**Figure 3.1**).

3.3.4 Calculating habitat targets

We calculated habitat targets following the SAR based approach developed by Desmet & Cowling (2004), which treats the SAR as a power function. While reservations about using this particular approach in conservation planning have been expressed in the literature (see Smith 2010 for a detailed review) we employed it in our study because: (i) we specifically sought to investigate the uncertainties around this existing approach; and (ii) the power function has been shown to perform well for macrobenthic datasets containing between 42 and 1300 samples (Azovsky 2011).

This approach involves transforming the power function (**Equation 3.1**) to estimate the proportion of habitat area required to represent a given percentages of species (**Equation 3.2**):

Equation 3.1 $$S = cA^{Z}$$

Equation 3.2 $$log\,A = \, log\,S/z$$

Here *S*' and *A*' denote the proportion of species and habitat area respectively (Desmet & Cowling 2004; Rondinini & Chiozza 2010), and *z* describes the slope of the power function, which is the rate of species accumulation with increase in area (Lomolino 2000; Tjorve & Tjorve 2008). The constant c is a scaling factor that relates to the size (area) of an individual sampling unit and can be ignored when comparing proportions or percentages of species and area (Desmet & Cowling 2004; Rondinini & Chiozza 2010). Thus, it is possible to calculate habitat targets by: (i) determining the *z*-value of the SAR for a given habitat; (ii) using the *z*-value to calculate the percentage of area required to represent a given percentage of species, and (iii) multiplying this percentage value by the total habitat area.

We calculated habitat specific *z*-values using the formula for calculating the slope of a straight line (**Equation 3.3**), because a SAR modelled with a power function appears as a straight line with slope z on a log-log plot (Desmet & Cowling 2004).

Equation 3.3 $$z = (y_2 \, - \, y_1)/(x_2 \, - \, x_1)$$

This relationship is expressed by; y_2 = log(total number of species in a habitat class); y_1 = log(average number of species per sampling point); x_2 = log(total area of habitat class); and x_1 = log(average area of sampling points). Three of these variables (y_1 , x_2 , x_1) are derived from habitat specific inventory data (Desmet & Cowling 2004; Rondinini & Chiozza 2010), so all that is needed to calculate *z*-values is to estimate the total number of species (y_2) in a given habitat type (Desmet & Cowling 2004).

The habitat map shows the distribution of each EUNIS level 3 habitat type and sub-divides the sedimentary habitat types further into finer-scale EUNIS level 4 types (**Figure 3.1**). Thus, we assigned sampling points on rocky habitats to their associated level 3 habitat types and sampling points on sedimentary habitats to both their associated parent level 3 habitat types, and their constituent level 4 habitat types (**see Figure S3.1** and **Table S3.1** in Supporting Information for more information regarding EUNIS level 3 parent habitats for level 4 habitat types in the EEC). We then calculated targets for each of these level 3 and level 4 habitats by using *EstimateS* software (Colwell 2009) to generate estimates of total species richness (y_2) and determine habitat specific *z*-values for each of these habitat types.

Although there is no consensus as to which estimator provides the best predictions when estimating total species richness for a habitat type (or region) from field survey data (Brose 2002; Herzog *et al.,* 2002; Chiarucci *et al.,* 2003; Walther & Moore 2005), there is general agreement that the Bootstrap estimator is the most conservative (Colwell & Coddington 1994; Chiarucci *et al.,* 2001; Chiarucci *et al.,* 2003; Hortal *et al.,* 2006). A prediction of total species richness based on this estimator should be considered as a minimum estimate (Desmet & Cowling 2004; Rondinini 2011a), which is why this estimator was subsequently applied by the SANBI and MCZ projects to develop national targets for both terrestrial and marine habitats.

To assess the effect that choice of species-richness estimator has on the calculation of conservation targets, we compared targets derived using the Bootstrap estimator to those derived using several alternative non-parametric estimators of species richness – ICE, Chao2, Jackknife1, and Jackknife2. While these alternative estimators were investigated by both Desmet and Cowling (2004) and Rondinini (2011a) these authors did not explicitly test their effect on target setting (see Colwell & Coddington 1994; Gotelli & Colwell 2001; Hortal *et al.,* 2006; Colwell 2009 for more details on these estimators and their performance). Our comparison involved calculating each richness estimate based on the mean of 1000 estimates that used 1000 randomisations of sample accumulation order without replacement (Colwell 2009). We then used these results to: (i) calculate the proportion of habitat area required to represent 80% of species, hereafter referred to simply as "targets", for each habitat type with > 5 sampling points – we chose to calculate targets based on representing 80% of species because this was used by the Balanced Seas and the other regional MCZ projects (JNCC & Natural England 2010); (ii) estimate the number of sampling points required to produce a stable target for each habitat type, and each richness estimator, where a target was defined as stable if it exhibited a standard deviation of < 5% (as used by Desmet & Cowling 2004); (iii) assess how the targets developed in this study compare with those from the MCZ project in the EEC; and (iv) assess how sensitive each of the estimators was to sample size effects by using successively larger numbers of accumulated sampling points, which involved dividing the percentage target for each habitat type based on 100, 200, and 300 sampling points by the percentage target based on 50 sampling points (we then took the mean of each of these habitat results for each estimator to show how relative target size changed with sample size).

Finally, we investigated the effects of using different levels of habitat classification on the extent of the MPA network needed to meet the targets. This involved multiplying each habitat target by the extent of its occurrence in the planning region to provide an area target in km^2 and then summing these area targets from EUNIS level 4 habitats belonging to the same "parent" level 3 type, so that the combined level 4 result could be compared with the level 3 result.

3.4 RESULTS

Based on using stable results for the Bootstrap estimator, the total number of species estimated to occur in each habitat class ranged between 240 and 1665 for the six EUNIS level 3 habitats, whilst estimates for the ten EUNIS level 4 habitats ranged between 160 and 1470 (**Table 3.1**). Habitat specific *z*-values ranged between 0.098 for deep sea mixed sediments and 0.162 for sublittoral sand (**Table 3.1**). Percentage targets ranged from 10.27% for deep sea mixed sediments to 25.28% for sublittoral sand (**Table 3.1**), so that eight of the EUNIS level 4 habitats and four of the EUNIS level 3 habitats had targets of greater than 10% (**Table 3.1**). Based on the available data for each habitat investigated, this would translate into approximately 18.41% of the EEC for the finer-scale EUNIS mixed level 3 and 4 habitat classification (**Figure 3.1**), compared to 20.27% for the coarse-scale EUNIS level 3 habitat classification (**Figure. S3.1**).

We found that both species richness estimates (**Table S3.2**) and resulting targets, varied between different estimators, with the difference in targets for a given habitat ranging between 1.58% for infralittoral coarse sediment, and 7.66% for low-energy circalittoral rock (**Table 3.2**). In addition, there were clear differences in the number of sampling points required to reach stable target estimates across estimators, with the Bootstrap estimator producing twelve stable target estimates, compared to five for the Jackknife1 estimator (**Table 3.2**). Moreover, the Bootstrap estimator generally required the smallest number of sampling points to reach stable estimates compared to the other estimators. For example, for a relatively well sampled habitat such as sublittoral sand with a total of 469 sampling points, the Bootstrap estimator required 276 sampling points to reach stability compared to 409 for Chao2 (**Table S3.3**).

When we evaluated how targets calculated with the Bootstrap estimator varied with successively larger numbers of accumulated samples, we found that estimates of both species richness and targets increased with sampling effort (**Table 3.3**). For example, we found that for four relatively well sampled habitats (sublittoral coarse sediment, infralittoral coarse sediment, circalittoral coarse sediment, and sublittoral sand) targets increased by 39%, 30%, 39%, and 45% respectively when the number of sampling points increased from 50 to 300 (**Table 3.3**), with the mean relative target increasing by 41% across all habitats (**Figure 3.2**). In addition, the standard Bootstrap approach produced targets that were most influenced by sample size, as the mean relative increase in targets for the other estimators ranged from 26% for ICE to 33% for Jackknife1 when the number of sampling points increased from 50 to 300 (**Figure 3.2**).

Table 3.1 Habitat specific inventory data, total number of species estimated to occur in each habitat type (values calculated using Bootstrap estimator and rounded to nearest whole number), *z*-values and the proportion (%) of target habitat area for each EUNIS level 3 and 4 habitat type.

EUNIS Code	EUNIS Level	EUNIS Habitat Description	Area (km^2) of habitat	Number of sampling points	Average area (m^2) of samples	Average number of species per sample	Total number of observed species	Bootstrap Estimator (y_2)	Number of stations to reach stable estimate	*z*-value	Target (%)
A3.3	3	Low-energy infralittoral rock	116	11	0.5	10	60	74	-	0.104	11.68
A4.3	3	Low-energy circalittoral rock	108	5	0.5	38	142	178	-	0.080	6.25
A5.1[†]	3	Subtidal coarse sediment	29889	725	0.26	53	1520	1665	65	0.135	19.23
A5.13	4	Infralittoral coarse sediment	4092	263	0.2	46	971	1079	67	0.133	18.65
A5.14	4	Circalittoral coarse sediment	18934	373	0.31	59	1326	1470	53	0.129	17.84
A5.15	4	Deep circalittoral coarse sediment	6863	89	0.25	49	825	950	52	0.123	16.38
A5.2 [a]	3	Subtidal Sand	7633	469	0.45	18	714	823	276	0.162	25.28
A5.23 or A5.24	4	Infralittoral fine sand or muddy sand	3701	288	0.45	18	590	684	208	0.159	24.65
A5.25 or A5.26	4	Circalittoral fine sand or muddy sand	3046	165	0.45	18	454	539	133	0.150	22.63
A5.27	4	Deep circalittoral sand	886	16	0.28	14	128	160	15	0.111	13.48
A5.3 [a]	3	Subtidal mud	335	28	0.48	21	198	240	27	0.120	15.49
A5.33 or A5.34	4	Infralittoral sandy mud or fine mud	196	17	0.49	18	139	170	-	0.113	13.97
A5.35 or A5.36	4	Circalittoral sandy mud or fine mud	134	11	0.46	26	131	158	-	0.093	8.98
A5.4 [a]	3	Subtidal mixed sediments	900	64	0.26	25	333	393	44	0.130	16.88
A5.44	4	Circalittoral mixed sediments	477	50	0.3	25	245	287	38	0.115	14.41
A5.45	4	Deep mixed sediments	198	14	0.11	25	164	202	13	0.098	10.27

[a] Species Richness estimates and corresponding z-values for these EUNIS level 3 habitats are obtained from their combined EUNIS level 4 habitat and survey data; A5.1 = (A5.13, A5.14, A5.15); A5.2 = (A5.23 or A5.24, A5.25 or A5.26, A5.27); A5.3 = (A5.33 or A5.34, A5.35 or A5.36); and A5.4 = (A5.44, A5.45).

Table 3.2 Proportion (%) of target habitat area for each of the EUNIS level 3 and 4 habitat types, based on five estimators of species richness. Shaded targets were determined not to be stable as the standard deviation of the richness estimate was > 5% of the estimate.

EUNIS Code	EUNIS Level	EUNIS Habitat Description	Number of sampling points	Non-parametric estimators					Mean Target	Target Range
				ICE	Chao2	Jackknife1	Jackknife2	Bootstrap		
A3.3	3	Low-energy infralittoral rock	11	17.53	14.96	14.28	16.31	11.68	14.95	5.85
A4.3	3	Low-energy circalittoral rock	5	13.91	12.07	8.89	11.17	6.25	10.46	7.66
A5.1	3	Subtidal coarse sediment	725	19.94	20.45	20.18	21.05	19.23	20.17	1.82
A5.13	4	Infralittoral coarse sediment	263	19.34	19.16	19.66	20.23	18.65	19.41	1.58
A5.14	4	Circalittoral coarse sediment	373	18.71	18.97	18.90	19.79	17.84	18.84	1.95
A5.15	4	Deep circalittoral coarse sediment	89	17.83	17.54	17.78	18.79	16.38	17.66	2.41
A5.2	3	Subtidal Sand	469	27.04	26.97	26.65	27.83	25.28	26.75	2.55
A5.23 or A5.24	4	Infralittoral fine sand or muddy sand	288	26.57	26.09	26.10	27.22	24.65	26.13	2.57
A5.25 or A5.26	4	Circalittoral fine sand or muddy sand	165	26.22	26.45	24.54	26.39	22.63	25.25	3.82
A5.27	4	Deep circalittoral sand	16	18.56	17.20	15.90	17.99	13.48	16.63	5.08
A5.3	3	Subtidal mud	28	20.70	20.24	17.96	20.27	15.49	18.93	5.21
A5.33 or A5.34	4	Infralittoral sandy mud or fine mud	17	19.15	19.15	16.66	19.15	13.97	17.62	5.18
A5.35 or A5.36	4	Circalittoral sandy mud or fine mud	11	13.61	14.84	11.56	13.98	8.98	12.59	5.86
A5.4	3	Subtidal mixed sediments	64	19.87	19.87	18.86	20.63	16.88	19.22	3.75
A5.44	4	Circalittoral mixed sediments	50	17.33	18.29	16.48	18.48	14.41	17.00	4.07
A5.45	4	Deep mixed sediments	14	16.14	14.83	12.72	15.01	10.27	13.79	5.87

Table 3.3 Species richness estimates and targets (values calculated using the Bootstrap estimator and rounded to nearest whole number) for each EUNIS level 3 and 4 habitat with increasing sample size.

			Number of sampling points used to generate estimates of species richness													
EUNIS Code	**EUNIS Habitat Description**	**Number of observed species**	**5**	*% Target*	**10**	*% Target*	**20**	*% Target*	**50**	*% Target*	**100**	*% Target*	**200**	*% Target*	**300**	*% Target*
A3.3	Low-energy infralittoral rock	60	**46**	*5.98*	**71**	*11.16*	-	-	-	-	-	-	-	-	-	-
A4.3	Low-energy circalittoral rock	142	**178**	*6.25*	-	-	-	-	-	-	-	-	-	-	-	-
A5.1	Sublittoral coarse sediment	1520	**252**	*2.61*	**394**	*5.88*	**563**	*9.03*	**823**	*12.59*	**1039**	*14.81*	**1257**	*16.61*	**1384**	17.52
A5.13	Infralittoral coarse sediment	971	**210**	*3.05*	**324**	*6.63*	**460**	*10.02*	**672**	*13.87*	**848**	*16.24*	**1019**	*18.08*	-	-
A5.14	Circalittoral coarse sediment	1326	**274**	*2.71*	**419**	*5.92*	**589**	*8.99*	**845**	*12.47*	**1052**	*14.61*	**1271**	*16.44*	**1400**	*17.38*
A5.15	Deep circalittoral coarse sediment	825	**232**	*3.18*	**365**	*6.92*	**527**	*10.46*	**787**	*14.49*	-	-	-	-	-	-
A5.2	Sublittoral sand	714	**87**	*3.56*	**138**	*7.57*	**210**	*11.77*	**334**	*16.54*	**460**	*19.75*	**611**	*22.51*	**709**	23.91
A5.23 or A5.24	Infralittoral fine sand or muddy sand	590	**87**	*3.94*	**139**	*8.27*	**208**	*12.47*	**335**	*17.51*	**460**	*20.76*	**604**	*23.46*	-	-
A5.25 or A5.26	Circalittoral fine sand or muddy sand	454	**88**	*4.15*	**136**	*8.23*	**200**	*12.27*	**312**	*17.02*	**430**	*20.36*	-	-	-	-
A5.27	Deep circalittoral sand	128	**73**	*5.20*	**120**	*10.31*	-	-	-	-	-	-	-	-	-	-
A5.3	Sublittoral mud	198	**91**	*4.51*	**139**	*9.04*	**202**	13.44	-	-	-	-	-	-	-	-
A5.33 or A5.34	Infralittoral sandy mud or fine mud	139	**82**	*5.42*	**127**	*10.41*	-	-	-	-	-	-	-	-	-	-
A5.35 or A5.36	Circalittoral sandy mud or fine mud	131	**104**	*4.34*	**151**	*8.44*	-	-	-	-	-	-	-	-	-	-
A5.4	Sublittoral mixed sediments	333	**106**	*3.36*	**162**	*7.26*	**233**	*11.13*	**354**	*15.74*	-	-	-	-	-	-
A5.44	Circalittoral mixed sediments	245	**99**	*3.22*	**143**	*6.65*	**197**	*10.12*	**287**	*14.41*	-	-	-	-	-	-
A5.45	Deep mixed sediments	164	**107**	*3.80*	**167**	*8.17*	-	-	-	-	-	-	-	-	-	-

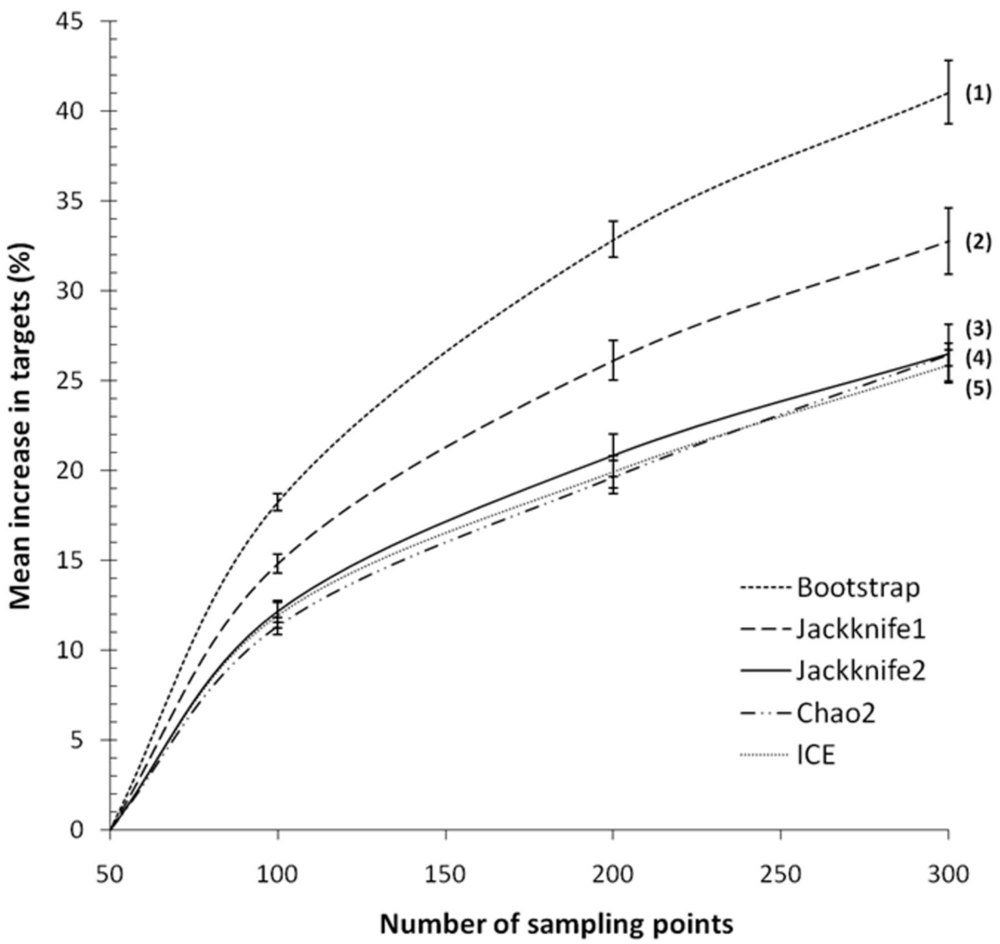

Figure 3.2 Mean increase in targets (including standard errors) based on increasing sample size across all habitats for the: (1) Bootstrap; (2) Jackknife1; (3) Jackknife2; (4) Chao2; and (5) ICE estimators, relative to an estimate based on 50 sampling points.

The level of habitat classification also impacted the targets, with species richness estimates, habitat specific *z*-values and targets being higher when developed for parent EUNIS level 3 habitats than for their finer-scale EUNIS level 4 constituents (**Table 3.1**). For example, the area of each parent EUNIS level 3 habitat needed to meet targets was 8.4% higher for sublittoral coarse sediments and 41.4% higher for sublittoral mixed sediments when compared to the combined target area of their finer-scale EUNIS level 4 constituents (**Figure 3.3**).

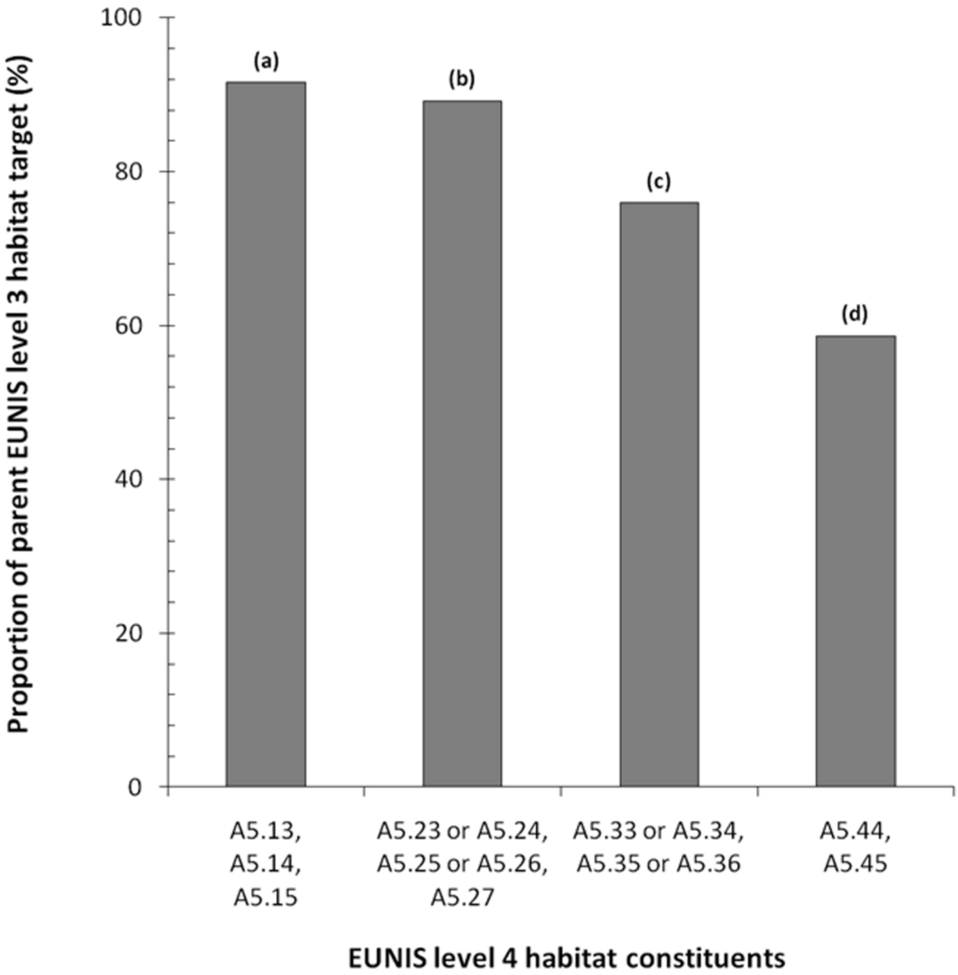

Figure 3.3 The proportion of target habitat area for combined fine-scale EUNIS level 4 habitat constituents compared to their coarse-scale EUNIS level 3 parent habitats: (a) A5.1; (b) A5.2; (c) A5.3; and (d) A5.4.

Finally, our regional EEC targets developed in this study were lower than the national MCZ targets developed for EUNIS level 3 habitats, with our targets ranging between 15.49% - 25.28% compared to 29.80% - 32.40% recommended by the MCZ Ecological Network Guidance, producing large differences in the area of habitat needed to meet these targets (**Table 3.4**).

Table 3.4 Habitat specific *z*-values and targets for four broad-scale EUNIS level 3 habitats developed for the eastern English Channel (EEC) in this study, and as provided by the Marine Conservation Zone (MCZ) Ecological Network Guidance in the UK (JNCC & Natural England 2010).

EUNIS Code	EUNIS Habitat Description	Area (km^2) of habitat in EEC	Number of EEC sampling points	EEC habitat *z*-values	EEC Target (%)	Number of MCZ sampling points	MCZ habitat *z*-values [a]	MCZ Target (%)	Difference in habitat area (km^2)
A5.1	Sublittoral coarse sediment	29889	725	0.14	19.23	8532	0.19	32.40	3936.38
A5.2	Sublittoral sand	7633	469	0.16	25.28	9065	0.18	29.90	352.64
A5.3	Sublittoral mud	335	28	0.12	15.49	2064	0.17	29.80	47.94
A5.4	Sublittoral mixed sediments	900	64	0.13	16.88	1922	0.18	31.90	135.18

[a] MCZ habitat specific *z*-values based on estimates of the average area of samples (x_1) being 0.5m ^{2} (see Rondinini 2011a)

3.5 DISCUSSION

The SAR is increasingly being used to set targets for habitat types in systematic conservation planning (Smith 2010), and has been specifically advocated for use in marine conservation planning (Neigel 2003; Smith *et al.,* 2009). Nonetheless, SAR based targets have to be part of a broader set of PA design parameters because they relate only to the minimum representation of biodiversity, i.e. ensuring the presence of a species regardless of its abundance, rather than ensuring its persistence (Smith 2010). Moreover, the approach provides no information about where PAs should be located within a particular habitat type (Desmet & Cowling 2004; Justus *et al.*, 2008; Chittaro *et al.,* 2010; Rondinini & Chiozza 2010). However, SAR-based target setting is likely to remain an important element of terrestrial and marine PA network design. This paper is the first to investigate several key issues that may affect the robustness of targets set using this approach. In this section we discuss the factors that affect the target setting process and their implication for conservation planning, with particular reference to the EEC. Finally, we examine the role of international policy-based targets in conservation planning and how they should be used in conjunction with the SAR-based approach.

3.5.1 Effects of sample size, species-richness estimators and habitat classification level

The value of the SAR-based approach depends entirely on producing accurate habitat specific *z*-values which, in turn, requires accurate estimates of total species richness within each habitat type. However, species richness estimates may be sensitive to the type of estimator used (**Table S3.2**) and the amount and quality of biological survey data employed, rather than reflecting true differences in species accumulation rates (Colwell *et al.,* 2004; Walther & Moore 2005; Hortal *et al.,* 2006; Rondinini & Chiozza 2010). Our results show that the rate of species accumulation with increase in area (expressed as the *z*-value) for each habitat type was quite similar across estimators (**Table S3.4**) which is consistent with other studies that have investigated their behaviour (Borges *et al.,* 2009). However, we show that sample size in particular can have a large influence on targets, so that increasing the number of sampling points often produced substantially higher targets (**Figure 3.2; Table 3.3**). The number of sampling points needed to produce a stable result also varied with estimator type, with the Bootstrap estimator generally requiring the fewest number to reach stability (**Table 3.2**) which is consistent with the results obtained for the MCZ project (Rondinini 2011a). This estimator is the most widely used for setting habitat targets (e.g. Desmet & Cowling 2004; Rondinini 2011a) and our stability results provide further support for this use (**Table 3.2**). However, we also found this estimator produced targets that were most influenced by changes in sample size (**Figure 3.2**), which raises doubts about the robustness of the targets produced using the standard Bootstrap-based approach.

We also investigated the extent to which using different habitat classification levels affects targets because SAR-based targets provide no information about where PAs should be located within a given habitat type. Thus, it is generally better to use the most detailed habitat classification available because this ensures each finer-scale habitat type is represented, but separating broad-scale parent habitat types into finer-scale sub-classes also involves dividing up the sampling points upon which the targets are based, and so we would expect these smaller sample sizes to produce lower targets. Our results confirmed this pattern, so that the area of each parent EUNIS level 3 habitat needed to meet the targets was always higher than the combined area of the constituent EUNIS level 4 habitat types (**Figure 3.3**). In some cases, dividing up the data into level 4 types led to sample sizes that were too small to produce stable results (**Table 3.2**), but even results for sublittoral coarse sediment and sublittoral sand habitats, which were relatively well sampled, showed that using the finer-scale level 4 instead of level 3 habitat classification reduced the total area needed to meet the targets (**Figure 3.3**). However, it is likely that this result was also directly influenced by using a more detailed habitat classification system. This is because habitats types that are subdivided into finer classes are more biologically homogenous, so the target area needed to represent a specified proportion of species may become lower (Whittaker *et al.*, 2001).

This means that conservation planners need to be careful when calculating and interpreting SAR-based targets, yet there is currently little guidance available in relation to sample size and choice of richness estimator. Perhaps the most important advice is to use only stable results and, based on this, Desmet and Cowling (2004) suggested a minimum sample size of 30. However, we found that this stability threshold is estimator-dependent and that it was possible to produce a stable result with a sample size as low as 14 (**Table 3.2**). Previous studies also implicitly recommend using the Bootstrap-based approach because it generally produces the most conservative targets (Desmet & Cowling 2004; Rondinini 2011a) but our results indicate that this estimator is the least likely to produce robust results. One way to overcome such problems would be to ensure that conservation planners adopt a standardised sampling strategy before collecting data because, as sampling becomes more exhaustive, this tends to produce more accurate estimates. This is because estimators will generally converge towards the same estimate of species richness (Colwell & Coddington 1994; Borges *et al.*, 2009) thereby providing a more reliable basis for setting targets. However, this will not always be possible, so we also need research on how to achieve post-hoc sampling parity between habitats, as simply using an equal number of samples per habitat type may over-sample habitats with a small extent of occurrence.

3.5.2 Applying SAR based targets in conservation planning

There is often a near-linear relationship between habitat targets and the extent of the resulting PA networks identified (Rodrigues & Gaston 2001; Warman *et al.*, 2004; Carpentier *et al.*, 2009; Delavenne *et al.*, 2012). Thus, setting unjustifiably high targets produces unnecessary impacts on the lives and activities of stakeholders (Chittaro *et al.*, 2010; Mascia *et al.*, 2010) and increases the costs associated with developing and managing the resulting PA systems (Naidoo *et al.*, 2006; McCrea-Strub *et al.*, 2011). We found that the national targets estimated for the MCZ projects (and applied by Balanced Seas) were between 18% and 92% higher than those estimated by this study for the four EUNIS level 3 habitats (**Table 3.4**), which implies an MPA network that would be 56.7% larger than if the MCZ targets were applied to the whole EEC. This is obviously a large discrepancy and so it is important to understand the differences in results and the level of uncertainty associated with each, especially as both studies used the same approach and the same richness estimator. The main source of difference appears to be in the sample size because the targets developed for the Balanced Seas project were based on national level data and the number of sampling points for each habitat type was between 2 and 3 orders of magnitude higher than for this study (**Table 3.4**). In addition, these national MCZ targets were based on all species recorded within the Marine Recorder database (Rondinini 2011a; Rondinini 2011b), whereas this study only used species obtained from macrobenthic surveys, and these different sets of species may show different biogeographical patterns.

This further supports the need for approaches that adjust percentage targets for sampling effort to produce results that account for total and per-habitat differences in sampling effort. It also emphasises that systematic conservation planning has to be seen as an adaptive process that accounts for improvements in data quality over time (Margules & Pressey 2000). The MCZ projects have followed this adaptive approach and gradually improved the quality of their ecological, socio-economic and resource-use data during the length of their project, as the UK Government recognised that this approach was the best compromise between accuracy and urgency. However, these MCZ networks are likely to be further modified, as part of a regular review process, and to form only part of marine spatial planning policy in the UK, so we would recommend that additional research on target setting is undertaken to inform these future developments. This research could also investigate the appropriateness of the current form of the SAR underpinning this approach (i.e. the power function) as previous work has shown that alternative functional forms, or mixes of these forms, are sometimes more appropriate (Stiles & Scheiner 2007; Guilhaumon *et al.,* 2008; Guilhaumon *et al.,* 2010; Smith 2010).

3.5.3 Policy driven and SAR based targets

The most widely known example of a conservation target defined by socio-political feasibility is the 10% target for world protected area coverage (IUCN 1993). This figure was subsequently adopted by the CBD in 2004 whereby 10% of ‘each of the world’s ecological regions’ was to be ‘effectively’ conserved by 2010 (CBD 2004d). However, at the 10th Conference of the Parties (COP) the proportion of terrestrial land area targeted for conservation was increased to 17%, whilst the proportion of the earth’s oceans targeted for conservation remained at 10% (CBD 2010a;

Harrop & Pritchard 2011). The use of such policy-based conservation targets has been heavily criticised in recent years with some scientists suggesting that they are ecologically irrelevant, undermine the goal of biodiversity protection, foster the assumption that every habitat type needs to be equally protected, and create the false expectation that such targets are sufficient for biodiversity representation and persistence (see review by Carwardine *et al.,* 2009). Our results suggest that the application of the 10% policy-driven habitat target would fail to represent the majority of species in the EEC adequately (**Table 3.1**), and are consistent with results from other studies (Desmet & Cowling 2004; JNCC & Natural England 2010; Rondinini 2011a).

However, there are two reasons why these policy-driven targets play a valuable role. First, they are generally time-bound and encourage governments to increase the extent of their MPA systems. Thus, the 10% targets should be seen in the context that only 1.17% of the total ocean area and 5.9% of territorial seas are currently designated as MPAs (CBD 2010a). Second, there are many occasions where there is insufficient data to develop SAR-based targets and so lower, policy-based targets can be used in the interim period. For example, we could not set targets for four of the EUNIS level 3 and two of the EUNIS level 4 habitat types in the EEC because of a lack of data. Therefore, our results suggest that policy-based targets can play a role as long as: (i) conservation practitioners are aware that they should be used as interim measure whilst SAR-based targets are being developed; and (ii) policy-based targets are low enough to ensure that no habitat type is over-represented in any eventual MPA system.

3.6 CONCLUSION

The SAR-based approach to setting habitat targets was developed to achieve two related goals. First, it provides a transparent and objective method for converting judgements of minimum species representation into a quantitative target. Second, it provides an approach for distinguishing between different habitat types and so tailors targets to account for differences in patterns of species richness and turnover. Our analysis shows that this approach can achieve these goals, but that issues relating to sample size (which are largely related to survey effort) and estimator choice have the potential to confound real differences between habitat types. Therefore, if this existing approach is to be applied to conservation decisions there is a need for substantial research on producing target estimates that account for sample size and data collection to address any issues of under-sampling. In the meantime, conservation practitioners should make use of best-available data and techniques to set habitat targets. However, they should be aware that time-bound policy targets can still offer a valid baseline whilst waiting for tailored targets to be developed.

3.7 ACKNOWLEDGEMENTS

We would like to acknowledge the following organisations and sources for providing the datasets used in this study: Cefas, Wimmereux Marine Station, Ifremer and the Marine Aggregate Levy Sustainability Fund (MALSF) GIS Database for providing access to the South Coast and Thames Regional Environmental Characterisation surveys. We are also grateful to Zoe Davies, Bruno Nhancale and Diogo Veríssimo for comments on earlier versions of this work, and to Simon Ferrier, Adam B. Smith and two anonymous referees for constructive comments that greatly improved this work.

CHAPTER 4

INVESTIGATING THE IMPACTS OF MINIMUM SIZE CONSTRAINTS WHEN ADOPTING A SYSTEMATIC CONSERVATION PLANNING APPROACH TO MARINE PROTECTED AREA NETWORK DESIGN

Kristian Metcalfe[a], Greg Vaughan[b], Sandrine Vaz[c] and Robert J. Smith[a]

[a] Durrell Institute of Conservation and Ecology (DICE), University of Kent, Canterbury, Kent, CT2 7NR, UK

[b] Balanced Seas, Durrell Institute of Conservation and Ecology (DICE), University of Kent, Canterbury, Kent, CT2 7NR, UK

[c] Institut Français de Recherche pour l'exploitation de la Mer (IFREMER), Laboratoire Ressources Halieutiques, 150 quai Gambetta, BP699, 62321 Boulogne-sur-Mer, France

4.1 ABSTRACT

Size and spacing guidelines based on species' movement and dispersal characteristics are increasingly used to inform the MPA network design process. However, the effects of using MPA size constraints in conservation planning are often poorly understood, despite having potentially major impacts on: (i) the overall size and spatial characteristics of MPA networks; (ii) stakeholders; and, (iii) connectivity. Therefore, to address this research gap we included MPA size constraints in a systematic conservation planning assessment using data from English waters. This involved: (i) using Marxan to identify networks of MPAs that met conservation feature targets, whilst minimising impacts on important areas for fisheries; and, (ii) using MinPatch to modify these networks to investigate the effects of setting different MPA size constraints when identifying priority areas for conservation in English waters. We found that increasing the minimum size of MPAs resulted in MPA networks that are: (i) comprised of a smaller number of MPAs; (ii) slightly larger and less fragmented; and, (iii) more costly to fisheries. For example, increasing MPA size constraints by a factor of 10 in inshore and offshore waters decreased the median number of MPAs in a network by 21%, but increased the median area of MPA network portfolios by 0.44% and cost to fisheries by 10.15%. We also show that increasing MPA size constraints resulted in reduced connectivity in the planning region for species that disperse 10 – 100 km. However, the impact was greater for species that disperse ≤ 10 km, because increasing MPA size constraints resulted in MPAs that were spaced further apart. Thus, these results highlight the importance of testing the impact of applying MPA size constraints before making recommendations about their adoption in MPA network design.

Keywords: *Connectivity, Europe, Marine Conservation Zones, Marxan, MinPatch, Systematic conservation planning, Viability*

4.2 INTRODUCTION

Marine protected areas (MPAs) are considered the cornerstone of most marine conservation strategies because effectively managed and well-designed MPA networks have consistently been shown to deliver ecological (Lester & Halpern 2008; McCook *et al.,* 2010), economic (Roberts *et al.,* 2001; Russ *et al.,* 2004) and social benefits (Cinner *et al.,* 2005). However, a number of studies have shown that MPA network design must be underpinned by six ecological principles to achieve these benefits, which are: representation, replication, adequacy, viability, connectivity and protection (Airame *et al.,* 2003; Roberts *et al.,* 2003a; Roberts *et al.,* 2003b; IUCN-WCPA 2008). A commonly used approach to help achieve these principles involves developing a list of important conservation features (species and habitats), setting targets for how much of each feature should be conserved and using spatial prioritisation software to identify where new MPAs should be located (Klein *et al.,* 2008b). This approach is named systematic conservation planning and implicitly accounts for MPA network representation, replication and adequacy by: (i) representing a full range of habitats; (ii) including replicates of each habitat, and; (iii) protecting a sufficient amount of each habitat to adequately conserve a range of associated species, communities and physical characteristics (Margules & Pressey 2000).

The systematic conservation planning approach also allows conservation practitioners to set targets related to viability, connectivity and protection but this is much less common. The development of Marxan with Zones conservation planning software has overcome this problem of accounting for protection, by allowing management type targets to be included in the analysis (Watts *et al.,* 2009; Wilson *et al.,* 2010). However, viability and connectivity are still rarely accounted for in systematic MPA design and this is problematic because MPAs need: (i) to be large enough to encompass the typical movements of species and support viable populations that are self-sustaining and maintain the integrity of the features throughout natural cycles of variation (Airame *et al.,* 2003; Roberts *et al.,* 2003a), and: (ii) to be spaced sufficiently apart to maintain large-scale ecological processes by maximising connectivity between individual MPAs, a process which is largely driven by the movement and dispersal of eggs, larvae, juveniles or adults (Palumbi 2003; Laurel & Bradbury 2006; Almany *et al.,* 2009).

Fortunately, this limitation with MPA network design is beginning to change and recent projects have adopted viability and connectivity goals, which are based on size and spacing targets. These targets are designed to protect species with a broad range of movement and dispersal characteristics and so maintain important ecological processes (Palumbi 2004). For example, setting the minimum size of an MPA based on the maximum home range of the target species and setting the maximum spacing that balances the different dispersal requirements would benefit the widest range of species (Moffitt *et al.,* 2010). Developing these MPA size and spacing constraints is not easy, as most marine organisms are not completely sedentary and vary greatly in their movement ability, both in the planktonic larval stage, and as adults (Grantham *et al.,* 2003; Palumbi 2003; Shanks *et al.,* 2003). Thus, most MPA projects typically based 'size and spacing' on studies of home range, tagging, genetics and dispersal (**Table 4.1**). In addition, researchers can also simulate population dynamics to model whether a proposed MPA network would support persistent populations of species with certain movement characteristics (e.g. Botsford *et al.,* 2001; Kaplan *et al.,* 2009; Moffitt *et al.,* 2009).

Until recently, however, accounting for these viability and connectivity targets has also been limited by the spatial prioritisation software. Software packages such as Marxan and Zonation do not allow users to set size and spacing constraints and this has had two important negative effects. First, it has not been possible to account for minimum MPA size and spacing targets in the spatial prioritisation analysis, forcing planners to modify the software outputs and produce less efficient results (Moilanen *et al.,* 2009). Second, it has prevented vital research on the impacts of applying these constraints on the size and socio-economic impacts of the resultant MPA networks. The recent development of MinPatch (Smith *et al.,* 2010b) helps overcome this problem, as this software package is designed to manipulate outputs from Marxan (Ball *et al.,* 2009) to ensure each MPA meets a minimum size threshold. Thus, it can be used to test the impact of setting different

MPA size constraints, which is important given that recommendations on the minimum size varies widely (**Table 4.1**). Here we address this research gap by using software to investigate the effects of setting different MPA size constraints when identifying priority areas for conservation in English waters. More specifically, we: (i) use MinPatch to investigate how MPA size constraints affects the size, spatial characteristics and socio-economic impacts of MPA networks, and (ii) measure how these size constraints influence the connectivity of these networks.

Table 4.1 Marine protected area size and spacing guidelines.

Study	**MPA Size**	**MPA Spacing**
GBRMPA (2002)	20 km in dimension, with the exception of coastal regions where the minimum size was 10 km due to fine scale patterns of diversity in coastal areas.	-
Halpern & Warner (2003)	10 – 100 km^2 should adequately protect and maintain the density and biodiversity of a large proportion of benthically associated organisms.	-
Palumbi (2003)	10 – 20 km in dimension to accommodate larval dispersal distances of species that show genetic isolation by distance.	25 – 150 km apart to reflect dispersal variation among different taxa.
Shanks *et al.*, (2003)	4 – 6 km in dimension to retain the propagules of short distance dispersers.	10 – 20 km apart to promote connectivity among protected areas for species with a pelagic larval phase.
CDFG (2008)	10 – 20 km in dimension to best protect adult populations based on neighbourhood sizes and movement patterns.	50 – 100 km apart to facilitate dispersal of important bottom dwelling fish and invertebrate groups.

Roberts *et al.*, (2010)	5 km in their minimum dimension, and that the average size in a network should be 10 – 20 km in dimension in territorial waters (< 12 nautical miles). Although in offshore waters (12 – 200 nautical miles) between 30 – 60 km in their minimum dimension to protect commercially valuable species that inhabit continental shelves and move relatively large distances.	40 – 80 km apart in order to assure sufficient ecological connectivity for sites supporting similar habitats.

4.3 METHODS

4.3.1 Study area

We defined the study area as the UK marine area for England, covering 243,263 km^2 of English territorial waters and offshore waters. Currently there are 542 European Marine Sites (EMSs) in these waters that are either wholly marine or contain a marine component, covering 17,018 km^2. These EMSs comprise of: 55 Special Protection Areas, 79 Special Areas of Conservation, 361 Sites of Special Scientific Interest, and 47 Ramsar sites. However, these sites alone do not fulfil the UKs requirement to designate an ecologically coherent and representative network of MPAs as part of obligations to the: (i) Convention for the Protection of the Marine Environment of the North-East Atlantic (OSPAR 2003b); (ii) European Marine Strategy Framework Directive (EC 2008b); and, (iii) Marine and Coastal Access Act (MCAA 2009). To address this, the UK Government set up four regional projects (JNCC & Natural England 2010) to identify a network of Marine Conservation Zones (MCZs) that meet a range of design criteria including minimum MPA size (**Table 4.1**). Therefore, in this paper we use the species and habitat distribution maps and targets developed as part of the MCZ project.

4.3.2 Producing the conservation assessment data

4.3.2.1 Conservation feature data

To represent both broad and fine-scale biodiversity patterns we collected data on two types of conservation feature (see **Table S4.1** in Supporting Information) as defined by the MCZ Ecological

Network Guidance (ENG). The first was based on a seabed habitat map produced by combining UKSeaMap (McBreen *et al.,* 2011) with intertidal habitat data for inshore waters (DEFRA 2010), and Mapping European Seabed Habitat (MESH) data for any remaining gaps in coverage (Coltman *et al.,* 2008). This combined map represented 30 broad-scale habitats (**Table S4.1**; **Figure 4.1**) that were based on the European Nature Information System (EUNIS) level 3 habitat classification (EEA 2006). The second was based on point distribution data for 26 species of conservation interest with low or limited mobility (**Table S4.1**; **Figure 4.1**), which are to be protected within MPAs in each MCZ project region (JNCC & Natural England 2010). These data were downloaded from the UKs National Biodiversity Network Gateway (NBN 2010), and were restricted to records from 1980 – 2009 to more accurately reflect the current distribution of each species (n = 429).

4.3.2.2 Conservation feature targets

Targets for 17 of the 30 habitats were based on the ENG, which were developed using the species-area relationship based approach (JNCC & Natural England 2010; Metcalfe *et al.,* 2013a). Targets therefore reflect the minimum amount of habitat area required to represent 80% of species known to occur in each habitat type (**Table S4.1**). Limited data meant that eight habitat types did not have targets specified in the ENG, so we set these targets by calculating the mean of the targets for habitat types belonging to the same EUNIS Level 3 habitat class. However, none of the six deep-sea habitats had targets so, given that they are only found at two locations, we set their targets as being the same as the habitat type with the highest target listed in the ENG (**Table S4.1**). Species targets were also based on the ENG and developed to ensure that each regional project area contained a minimum of 3 replicates of each species, with the exception of those with < 3 records whose targets were based on the total number of records (**Table S4.1**).

4.3.2.3 Defining the planning region and planning units

To address recommendations that MPA size should generally be smaller in inshore waters than those in offshore areas (**Table 4.1**), we divided the planning region into 5 km^2 hexagon planning units inshore (based on the UK territorial limit 0 – 12 nautical miles) and 25 km^2 hexagons offshore

(> 12 nautical miles) using the Repeating Shape Extension in ArcView (Jenness 2005). We used hexagons because they produce more efficient and less fragmented portfolios (Nhancale & Smith 2011). The boundaries of the hexagons were then overlaid with a theme of existing EMSs using the Union function in ArcView (ESRI 2002), which resulted in protected areas that were divided up into hexagons or segments of hexagons at their boundaries. We then removed planning units that were less than 1 hectare (0.01 km^2; n = 579) to reduce eventual processing time in MinPatch, and calculated the amount of each conservation feature in the remaining planning units (n = 22,553) using the Conservation Land-Use Zoning (CLUZ) ArcView Extension (Smith 2004a).

4.3.2.4 Developing planning unit cost data

Given that opportunity costs are commonly used to influence the location of MPAs and so minimise their socio-economic impacts (Ban & Klein 2009), we assigned each planning unit a cost based on the spatial distribution of fishing effort for UK vessels > 15 m in length (classified according to six gear types: dredges, hooks and lines, nets, seines, traps, and trawls), which was derived from vessel monitoring system (VMS) data from 2007 (Lee *et al.,* 2010). To reflect the relative value of areas to fisheries we combined the estimated fishing effort reported as the time spent fishing (in hours) per unit area (0.05^{o}) for each gear to produce a single fishing effort layer using the ‘Raster Calculator’ function in ArcView (ESRI 2002). The ‘Summarise Zones’ function in ArcView was then used to calculate the cost for all planning units based on the mean number of hours fished, and was multiplied by the area of each planning unit (**Figure 4.1**). We adopted this approach as using VMS data alone would have favoured the selection of larger planning units with lower levels of fishing effort because they tend to contain more of each conservation feature. To account for planning units with no recorded VMS data, 1 was added to the value of all planning units so that there was a cost for selecting every planning unit.

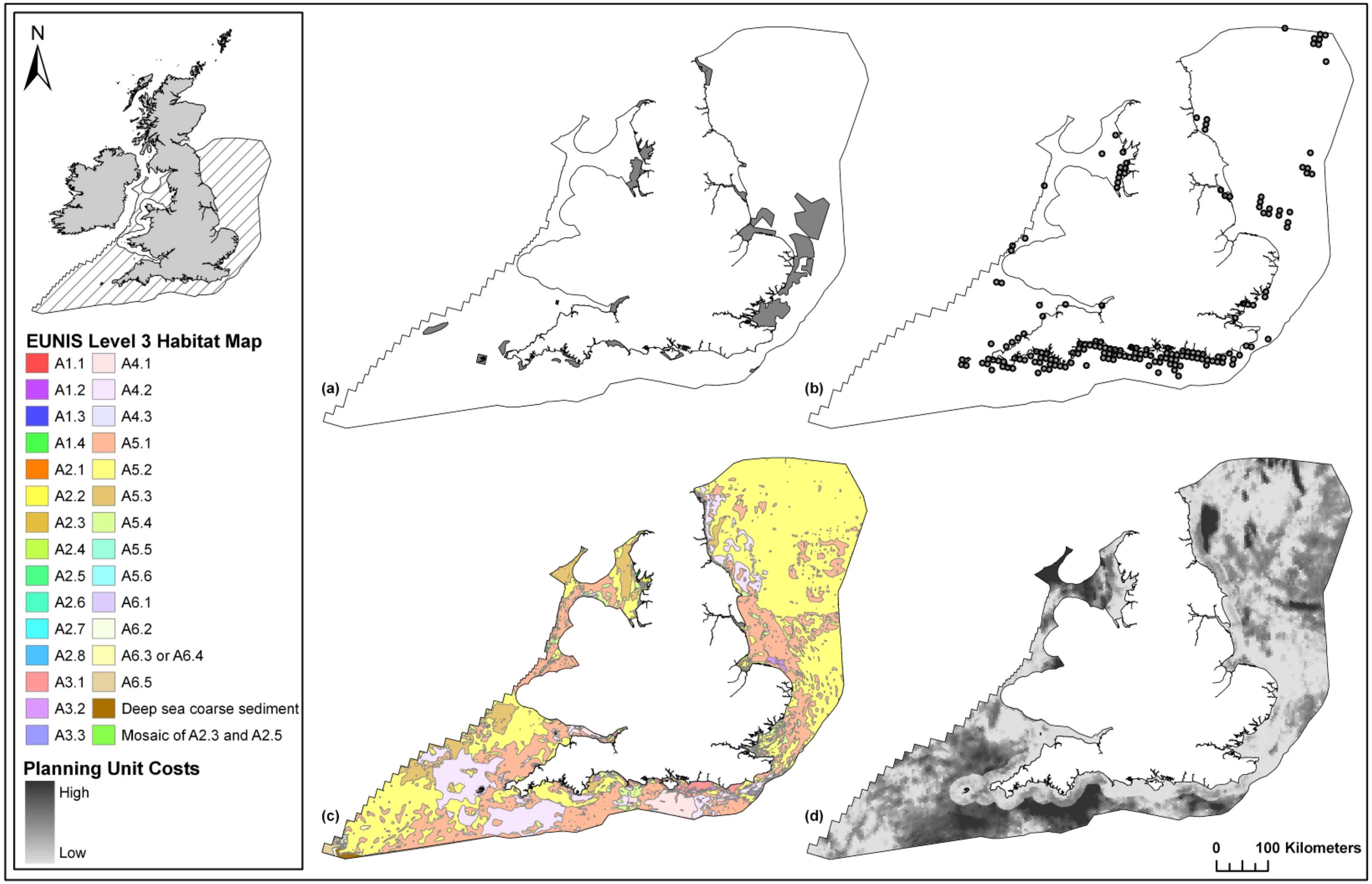

Figure 4.1 Details of the conservation planning assessment data: (a) European Marine Sites, based on designation as of July 2011; (b) species distribution records; (c) broad-scale EUNIS level 3 marine habitat map (see Table S4.1 for a description of EUNIS habitat codes); and (d) planning unit costs based on VMS data.

4.3.3 Running the conservation assessment

4.3.3.1 Using Marxan to identify portfolios of planning units

To use MinPatch to investigate the effects of setting different MPA size constraints in English waters we first ran Marxan 200 times using simulated annealing followed by iterative improvement technique, where each run consisted of two million iterations (Scenario 1; **Table 4.2**). Each run identified a portfolio of planning units that met the conservation feature targets at near-minimal cost and Marxan then identified: (i) the 'best' solution which is the cheapest of the 200 portfolios; and, (ii) a 'selection frequency' which counts the number of times each planning unit appeared in the different portfolios (Ball *et al.,* 2009). Based on a preliminary sensitivity analysis we used a boundary length modifier (BLM) value of 0.25 as this represented an acceptable trade-off between minimising the boundary of the selected areas (portfolio fragmentation) relative to the cost of the portfolio (Stewart & Possingham 2005). In addition, given that the principal aim of the MCZ project was to build on the existing network of MPAs (JNCC & Natural England 2010) we locked-in each of the EMSs so that Marxan would identify priority areas that met the current shortfall in the features targeted for protection. However, given that negotiating the boundaries of European sites are often complex (Metcalfe *et al.,* 2013b) we excluded adjacent planning units to ensure that Marxan did not identify portfolios of planning units that joined to existing protected areas.

Table 4.2 Details of marine protected area size constraint scenarios for inshore and offshore waters.

Scenario	Minimum size inshore [a] (km^2)	Added patch radius value (km)	Minimum size offshore [b] (km^2)	Added Patch Radius Value (km)	Spacing distance (km)	Buffer distance (km) [c]
1 Marxan	-	-	-	-	-	-
2 MinPatch	10	4.0	90	7.0	-	-
3 MinPatch	25	5.0	225	11.0	-	-
4 MinPatch	50	5.5	450	15.0	-	-
5 MinPatch	100	7.5	900	20.0	-	-
1a	-	-	-	-	10	5
1b	-	-	-	-	25	12.5
1c	-	-	-	-	50	25
1d	-	-	-	-	100	50
2a	10	4.0	90	7.0	10	5
2b	10	4.0	90	7.0	25	12.5

2c	10	4.0	90	7.0	50	25
2d	10	4.0	90	7.0	100	50
3a	25	5.0	225	11.0	10	5
3b	25	5.0	225	11.0	25	12.5
3c	25	5.0	225	11.0	50	25
3d	25	5.0	225	11.0	100	50
4a	50	5.5	450	15.0	10	5
4b	50	5.5	450	15.0	25	12.5
4c	50	5.5	450	15.0	50	25
4d	50	5.5	450	15.0	100	50
5a	100	7.5	900	20.0	10	5
5b	100	7.5	900	20.0	25	12.5
5c	100	7.5	900	20.0	50	25
5d	100	7.5	900	20.0	100	50

[a] Inshore waters are based on the UK territorial limit 0 - 12 nautical miles; [b] offshore waters are based on > 12 nautical miles; and [c] buffer distance = 50% of spacing distance to ensure the distance between the boundary of two MPAs does not exceed the maximum spacing distance.

4.3.3.2 Using MinPatch to apply the minimum protected area size constraint

To best represent the range of MPA size guidelines that have been developed for UK and international MPA projects (**Table 4.1**) we modified MinPatch so that we could apply different MPA size constraints in inshore and offshore waters. We then used this updated version of MinPatch to modify each of the 200 portfolios produced by Marxan (Scenario 1; **Table 4.2**) to investigate four scenarios that included different MPA size constraints for inshore and offshore waters (Scenario 2 – 5; **Table 4.2**). MinPatch then modifies each Marxan portfolio by: (i) identifying each planning unit cluster, referred to as MPAs hereafter; (ii) removing MPAs that are smaller than a user defined thresholds; (iii) adding entirely new planning unit clusters referred to as ‘patches’ that form the basis of new MPAs of the minimum size; and, (iv) converting these patches into suitable protected areas by removing any planning units (through a process named simulated whittling) that are not required to meet targets, minimise boundary costs or ensure each protected area meets the minimum size (**Figure 4.2**).

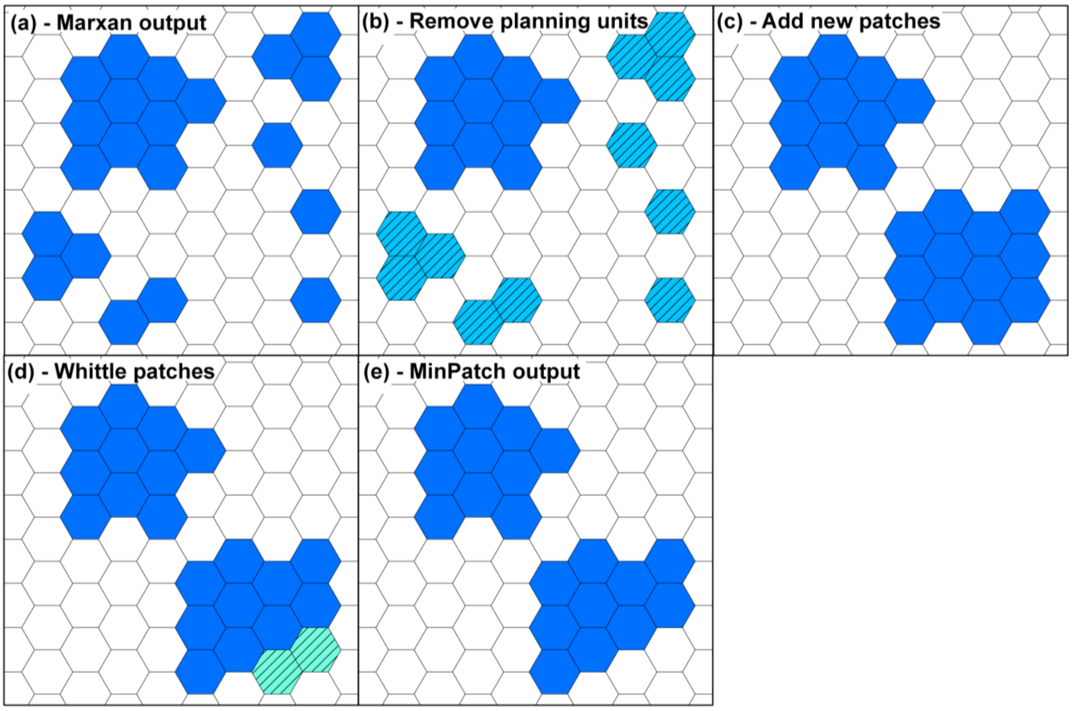

Figure 4.2 MinPatch process involves: (a) identifying each planning unit cluster from Marxan, referred to as MPAs; (b) removing MPAs that are smaller than a user defined threshold; (c) adding new planning unit clusters, referred to as patches that form basis of new MPAs of the minimum size; (d) converting these patches into suitable protected areas by removing (whittling) planning units that are not required to meet targets, minimise boundary costs or ensure each MPA meets the minimum size (e).

The user can adjust a parameter called the ‘added patch radius value’ in MinPatch to control whether the added MPAs are compact or elongated (Smith *et al.*, 2010b). Thus by specifying an added patch radius value of the threshold area plus an additional 60% (**Table 4.2**), we ensured that the initial patches did not contain many superfluous planning units and so could not be whittled down to elongated shapes. For each scenario we set the BLM value used in the simulated whittling stage to be the same as the value used in the respective Marxan analysis. To compare results from the initial Marxan analysis and the four MinPatch analyses, we used a non-parametric ANOVA, the Kruskal-Wallis test to determine whether differences existed in the 200 portfolios from the 5 analyses. More specifically we measured differences in number of patches, area of patches, total area of portfolios, boundary length, and planning unit cost among MPAs of different size. We also conducted post-hoc Mann Whitney U tests with Bonferroni correction for multiple comparisons to investigate differences between scenarios.

4.3.3.3 Measuring the connectivity of the MPA networks

To investigate how these different MPA size constraints influenced connectivity between MPAs we conducted a spatial analysis in ArcGIS (ESRI 2011). As part of this we identified a series of "connectivity networks", where each network is identified by specifying a buffer of a set size around each MPA in a portfolio. A more detailed approach would have been to develop connectivity networks based on MPAs that contain similar habitats; however, this was computationally more complex. These buffer distances thus define a patch within which species are expected to disperse from the associated MPA, so that MPAs with overlapping buffer areas are assumed to be close enough to allow dispersal between them. Thus, we defined a single connectivity network as a group of MPAs and buffer regions that form a contiguous area (**Figure 4.3**). In this research we investigated four of the maximum MPA spacing distances recommended in the literature (10 km, 25 km, 50 km, and 100 km; **Table 4.1**). We did this by using buffer distances that were half of the recommended MPA spacing distances (e.g. 5 km, 12.5 km, 25 km, and 50 km; **Table 4.2**) and so ensured that the maximum distance between MPAs belonging to the same connectivity network were within the associated spacing threshold.

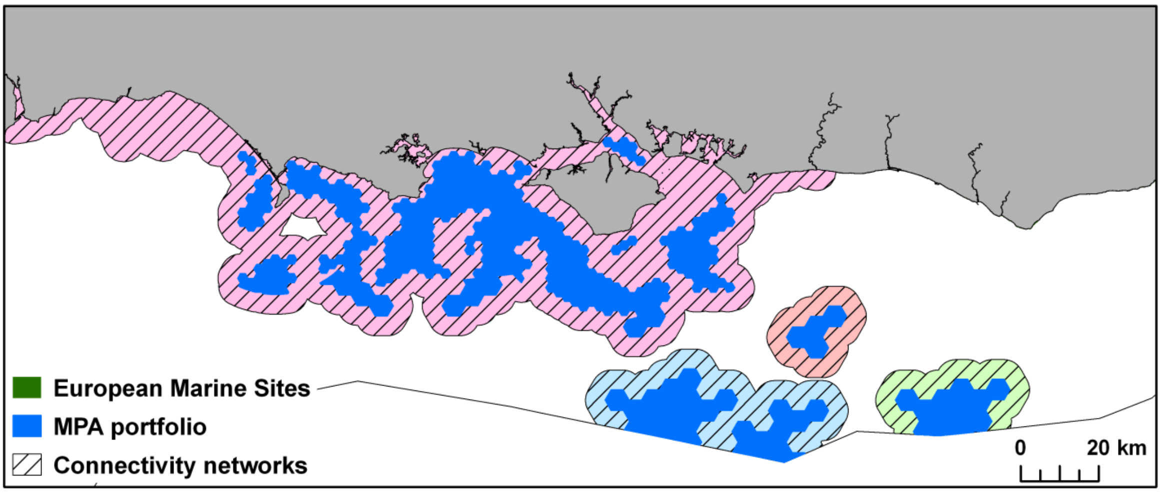

Figure 4.3 Example of four connectivity networks, based on a buffer distance of 5 km.

We carried out four spacing analyses based on the Marxan and four MinPatch analyses (i.e. 20 analyses in total). For each of the 200 portfolios produced in the Marxan and four MinPatch analyses, we used ArcGIS to: (i) create a 'buffer' to one of the four specified distance values around the existing EMSs, and MPAs identified in each portfolio; (ii) perform a 'dissolve' to remove any overlapping buffers; and, (iii) 'clip' the resulting connectivity networks to the MCZ project area; and, (iv) 'explode' to identify the number and area of each connectivity network in each portfolio. We then used the Kruskal-Wallis test to determine whether there were differences in the total number and combined area of the connectivity networks produced by the different analyses of the 200 portfolios and, conducted post-hoc Mann Whitney U tests with Bonferroni correction to investigate differences between scenarios.

In addition, to identify areas of high and low connectivity in the planning region we produced a selection frequency map based on the number of times each planning unit appeared in a connectivity network in the 200 portfolios produced for each of the 20 analyses. This involved using the Count Overlapping Polygons ArcView Extension, which produced maps showing the most important areas (Smith 2004b). To give a final overview, we also combined these selection frequency maps based on analyses with the same minimum MPA size threshold to identify the planning units that did not fall within any connectivity network, irrespective of the buffer distance.

4.4 RESULTS

4.4.1 Current levels of protection

The planning region had a total area of 243,263 km^2; of which 17,018 km^2 (7%) was contained within EMSs, and 5,460 km^2 (2%) bordered existing protected areas and were therefore excluded, leaving a total of 220,785 km^2 (91%) available for selection. The conservation assessment contained data on 76 conservation features, with targets for the different features ranging between 1 record for a pink sea fan (*Eunicella verrucosa)* to 41,608 km^2 of subtidal sand (**Table S4.1**). However, the extent to which these targets were met by the existing MPA network varied, with 33 of the conservation feature targets met or exceeded (≥ 100%), 18 under-represented (< 100%), and 25 not represented (0%) within EMSs (**Table S4.1**).

4.4.2 Conservation assessment results

4.4.2.1 Effects of using minimum MPA size constraints

The results of this study show that characteristics of MPA network portfolios differed significantly when applying different MPA size constraints; (i) number of patches (χ^2 = 959.6, d.f. = 4, p < 0.001); (ii) area of patches (χ^2 = 947.9, d.f. = 4, p < 0.001); (iii) total area of portfolios (χ^2 = 851.0, d.f. = 4, p < 0.001); (iv) boundary length (χ^2 = 919.9, d.f. = 4, p < 0.001); and (v) planning unit cost (χ^2 = 905.9, d.f. = 4, p < 0.001). Post-hoc tests confirmed that these characteristics were also significantly different between each MPA size scenario (all p < 0.05). Thus, these findings show that increasing the MPA size constraint resulted in MPA networks that are: (i) less dispersed throughout the planning region (**Figure 4.4**); (ii) comprised of a smaller number of patches that are larger in size **(Figure 4.5a**; **Figure 4.5b**); (iii) slightly larger and less fragmented, as indicated by a decrease in the boundary length of portfolios (**Figure 4.5c**; **Figure 4.5d**); and, (iv) more costly to fisheries (**Figure 4.5e)** in both inshore and offshore waters (**Figure 4.6**). For example, increasing the MPA size constraint by a factor of 10 (Scenario 5 compared to Scenario 2; **Table 4.2**) decreased the median number of patches by 21%, but increased the median area of MPA network portfolios by 0.44% and cost to fisheries by 10.15% (**Figure 4.5)**.

In addition, increasing the MPA size constraint resulted in a larger increase in the area of portfolios selected in inshore waters, in contrast to a small decrease in portfolio area selected in offshore waters (**Figure 4.6**). For example, increasing the MPA size constraint by a factor of 10 (Scenario 5 compared to Scenario 2; **Table 4.2**) increased the median area of portfolios in inshore waters by 4.5%, but decreased the median area of portfolios in offshore waters by 0.94% (**Figure 4.6**).

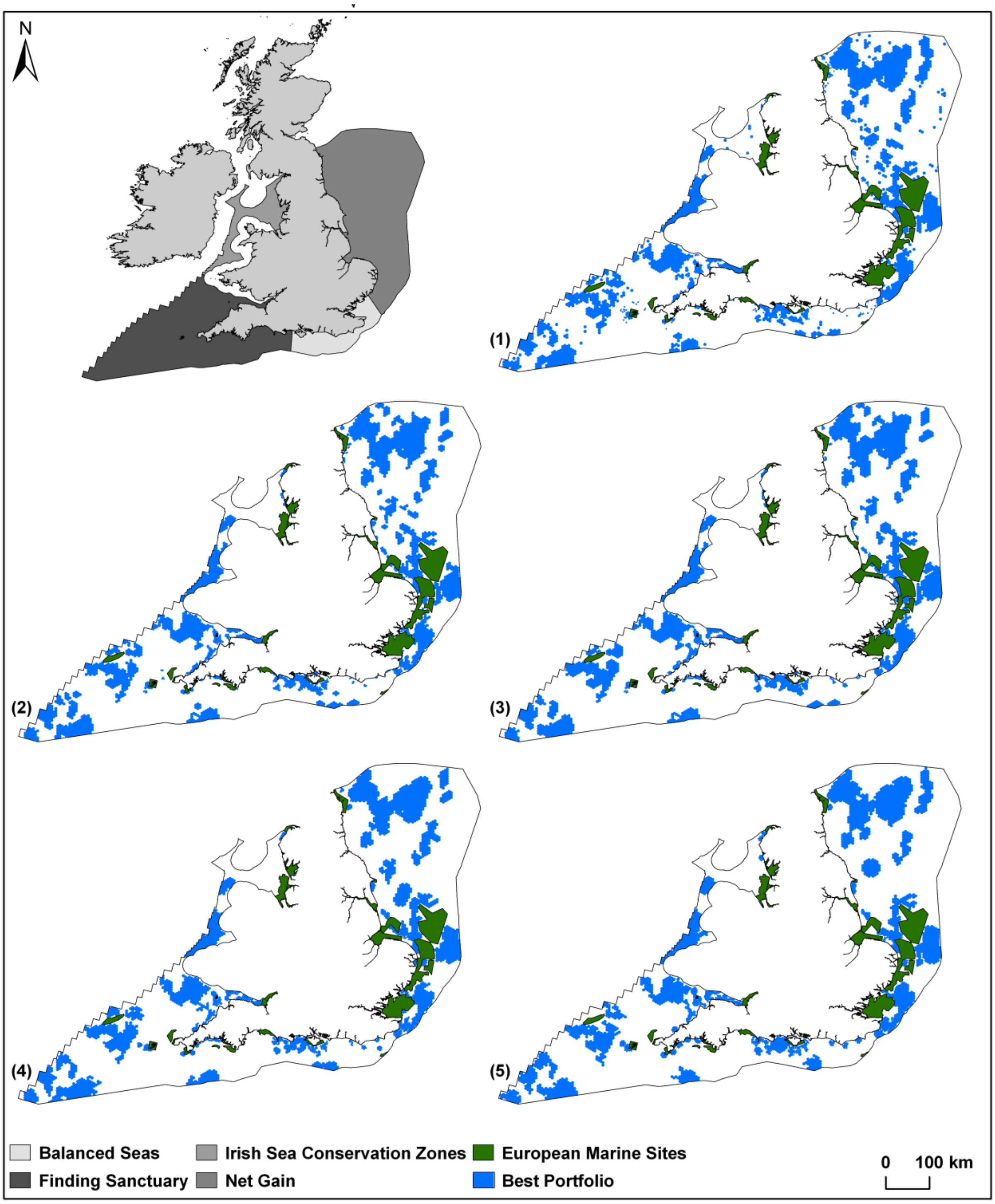

Figure 4.4 MCZ planning region showing the UK marine area for England and the location of the four regional MCZ projects and European Marine Sites, and examples of the best portfolios produced by Marxan (Scenario 1) and MinPatch (Scenario 2 – 5). See **Table 4.2** for details of each MPA size constraint scenario.

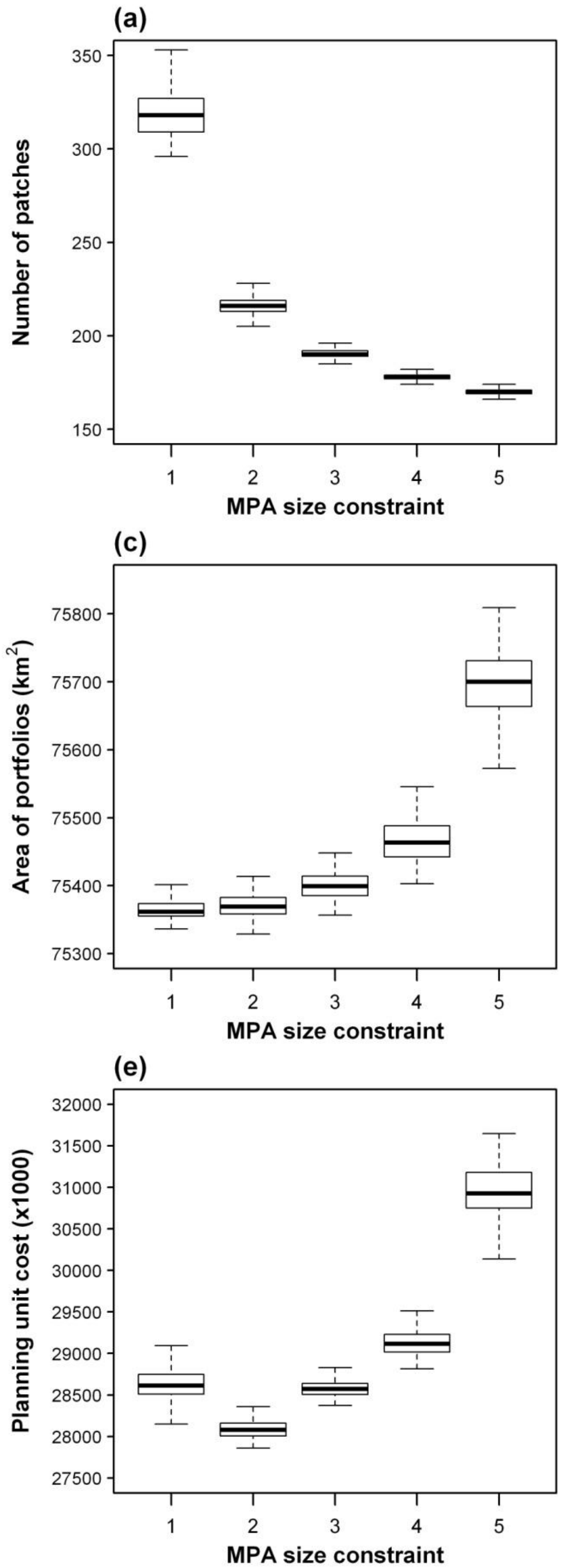

Figure 4.5 Characteristics of the 200 marine protected area network portfolios produced by Marxan (Scenario 1) and MinPatch (Scenario 2 – 5): (a) number of patches; (b) median area of patches (km^2); (c) area of portfolios (km^2); (d) boundary length (km); and, (e) planning unit cost. See **Table 4.2** for details of each MPA size constraint scenario.

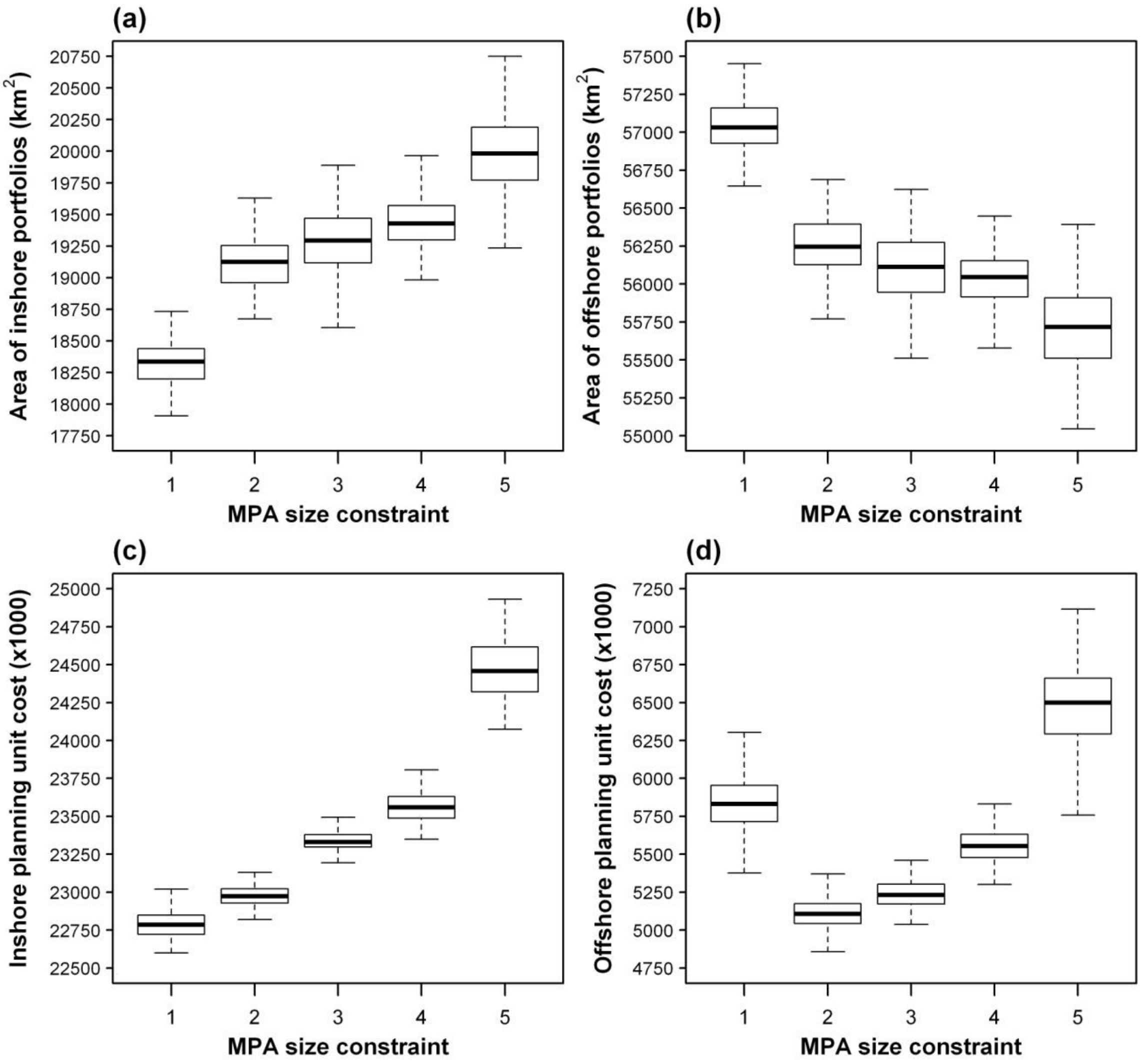

Figure 4.6 Inshore and offshore characteristics of the 200 marine protected area network portfolios produced by Marxan (Scenario 1) and MinPatch (Scenario 2 – 5): (a) area of inshore portfolios (km^2); (b) area of offshore portfolios (km^2); (c) inshore planning unit cost; and, (d) offshore planning unit cost. See **Table 4.2** for details of each MPA size constraint scenario.

4.4.2.2 Effects of using minimum MPA size constraints on portfolio connectivity

The results of this study also show that characteristics of portfolio connectivity, such as the total number and area of connectivity networks (**Figure 4.7a**; **Figure 4.7b**) differed significantly for each MPA size constraint scenario (**Table 4.3**). Post-hoc tests confirmed that these characteristics were also significantly different when connectivity networks were based on different spacing distances within each MPA size constraint scenario (all $p < 0.05$). Comparisons across scenarios of the same spacing distance (a: 10 km, b: 25 km, c: 50 km, and d: 100 km; **Figure 4.7a**; **Figure 4.7b**) revealed that the total number and area of connectivity networks also differed significantly with the minimum MPA size constraint (**Table 4.3**). However, post-hoc tests confirmed that these characteristics were not significantly different for all MPA size constraint scenarios: (i) total number of MPA networks (1b – 3b; 3c – 4c; 3c – 5c; and 4c – 5c, $p > 0.05$); and, (ii) total area of MPA networks (4d – 5d, $p > 0.05$).

Table 4.3 Results from the Kruskal-Wallis tests for the total number and area of connectivity networks in the Marxan and MinPatch portfolios for each MPA size constraint scenario, and across MPA size constraint scenarios of the same spacing distance.

Scenario	Number of MPA networks	Total area of MPA networks (km^2)
1a – 1d	$\chi^2 = 766.4$, d.f. = 3, $p < 0.001$	$\chi^2 = 749.1$, d.f. = 3, $p < 0.001$
2a – 2d	$\chi^2 = 758.6$, d.f. = 3, $p < 0.001$	$\chi^2 = 740.2$, d.f. = 3, $p < 0.001$
3a – 3d	$\chi^2 = 761.8$, d.f. = 3, $p < 0.001$	$\chi^2 = 749.1$, d.f. = 3, $p < 0.001$
4a – 4d	$\chi^2 = 764.3$, d.f. = 3, $p < 0.001$	$\chi^2 = 749.1$, d.f. = 3, $p < 0.001$
5a – 5d	$\chi^2 = 772.6$, d.f. = 3, $p < 0.001$	$\chi^2 = 749.1$, d.f. = 3, $p < 0.001$
1a – 5a	$\chi^2 = 758.9$, d.f. = 4, $p < 0.001$	$\chi^2 = 913.9$, d.f. = 4, $p < 0.001$
1b – 5b	$\chi^2 = 342.5$, d.f. = 4, $p < 0.001$	$\chi^2 = 946.5$, d.f. = 4, $p < 0.001$
1c – 5c	$\chi^2 = 711.9$, d.f. = 4, $p < 0.001$	$\chi^2 = 921.3$, d.f. = 4, $p < 0.001$
1d – 5d	$\chi^2 = 243.8$, d.f. = 4, $p < 0.001$	$\chi^2 = 652.4$, d.f. = 4, $p < 0.001$

Nonetheless, these findings show that increasing MPA size constraints generally resulted in reduced connectivity in the planning region for all spacing distances that were analysed, because the total number and area of connectivity networks decreased (**Figure 4.7a**; **Figure 4.7b**) as MPAs become larger and less dispersed (**Figure 4.4**). However, these results indicate that MPA size had a larger effect when using 10 km spacing compared to 100 km spacing. For example, increasing the MPA size constraint by a factor of 10 (Scenario 5 compared to Scenario 2; **Table 4.2**) decreased the median number and area of connectivity networks based on 10km spacing by 12.8% and 5% respectively, compared to 0% and 0.54% for networks based on 100 km spacing.

In addition, increasing minimum MPA size constraints led to an increase in the number of planning units falling outside a connectivity network (**Figure 4.8**). The selection frequency maps based on analyses with the same minimum MPA size threshold showed that these unselected planning units tended to be found on the edge of the planning region, as well as the central North Sea, the Western English Channel and the Dover Straits (**Figure 4.9**).

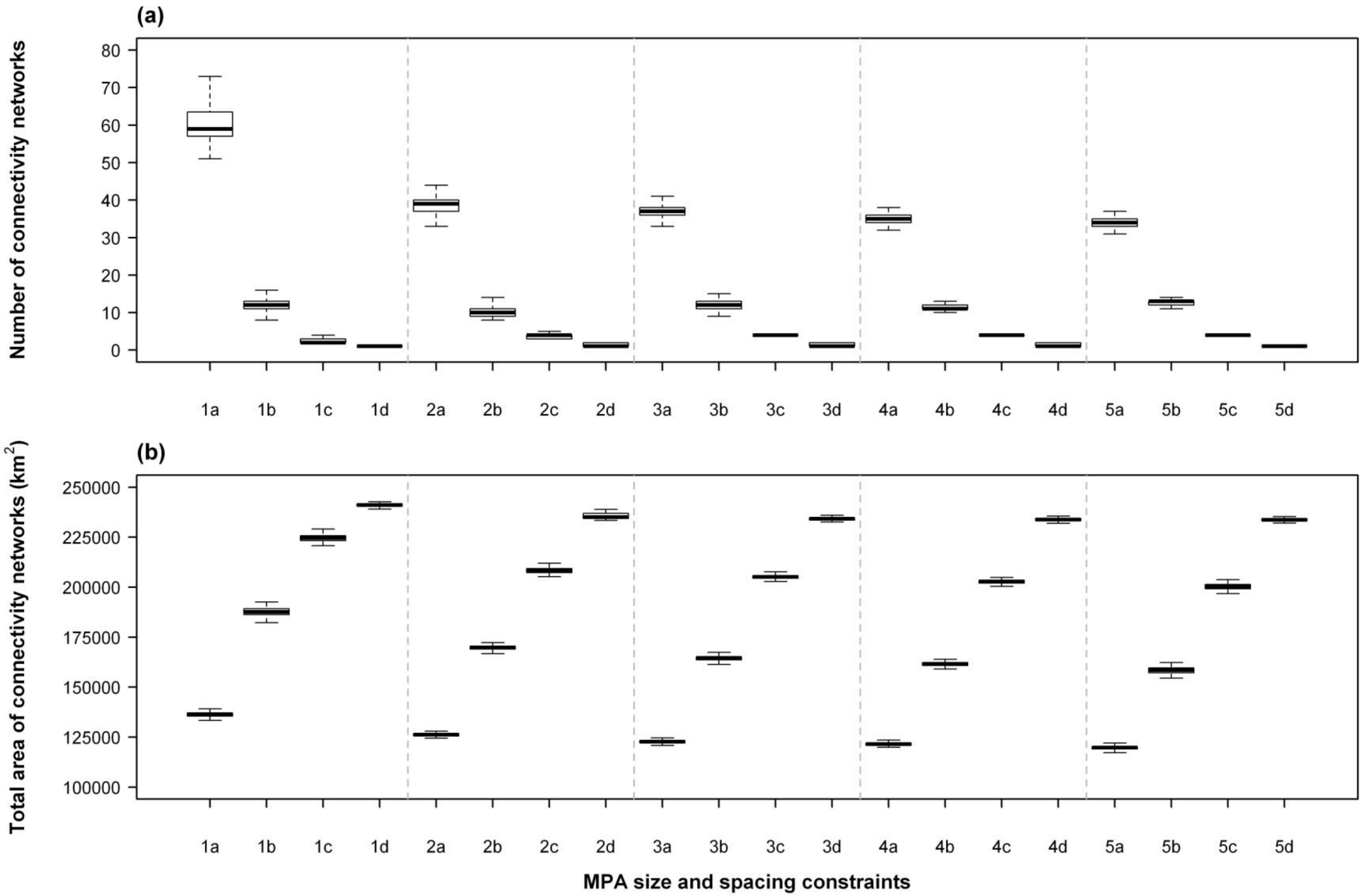

Figure 4.7 Characteristics of the 200 marine protected area network portfolios produced by Marxan (Scenario 1) and MinPatch (Scenario 2 – 5) for a range of spacing distances: (a) number of connectivity networks; and, (b) total area of connectivity networks (km^2). See **Table 4.2** for details of each MPA size constraint scenario.

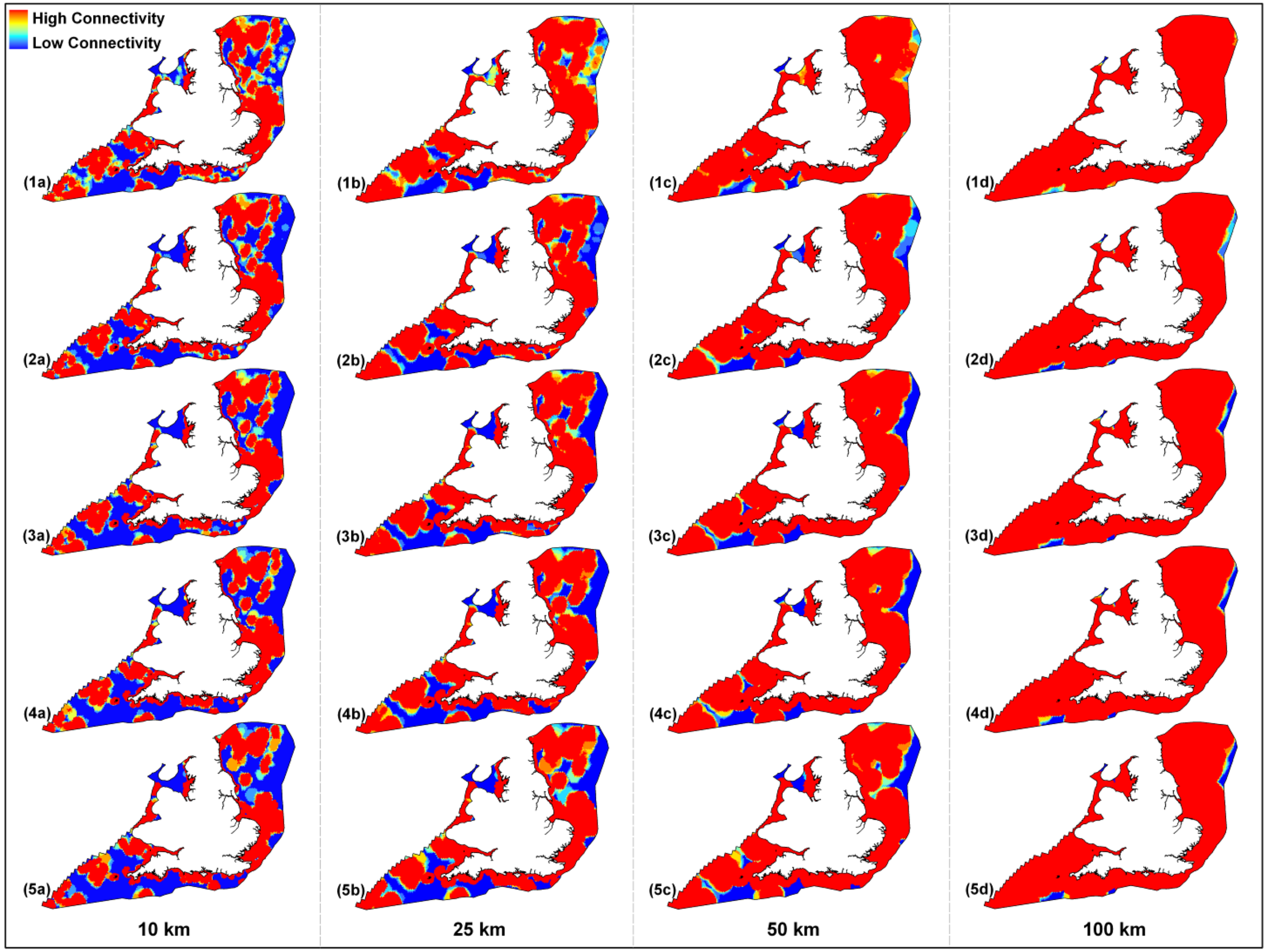

Figure 4.8 Selection frequency based on the number of times each planning unit appeared in a connectivity network based on Marxan (Scenario 1) and MinPatch (Scenario 2 – 5) outputs for: (a) 10 km; (b) 25 km; (c) 50 km; and, (d) 100 km spacing distance. See **Table 4.2** for details of each MPA size constraint scenario.

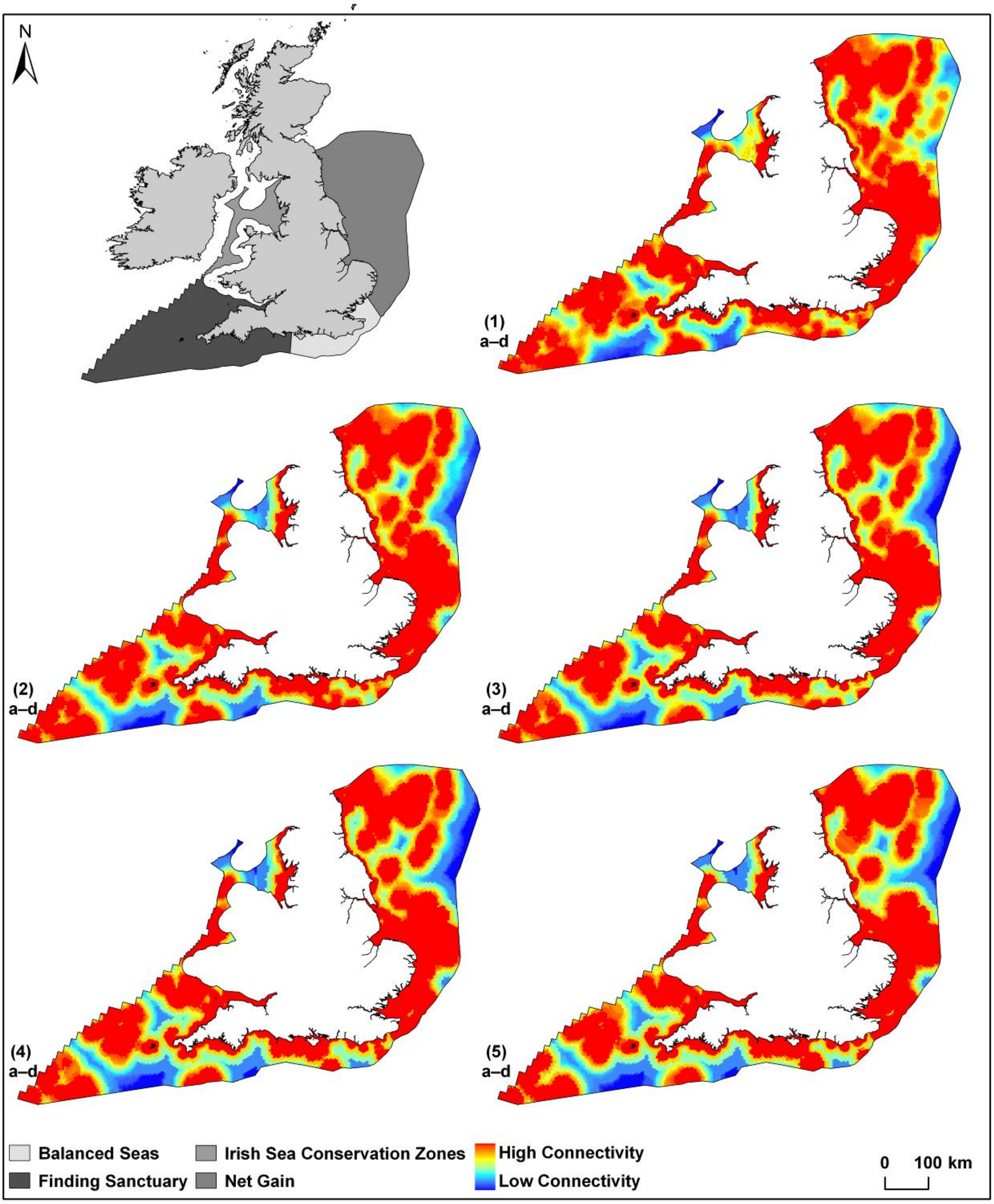

Figure 4.9 Summed selection frequency based on the number of times each planning unit appeared in a connectivity network for Marxan (Scenario 1) and MinPatch (Scenario 2 – 5) outputs for all spacing distances (10 – 100 km). See **Table 4.2** for details of each MPA size constraint scenario.

4.5 DISCUSSION

A fundamental question in marine conservation concerns the minimum size and spacing of MPAs needed to ensure the representation and persistence of the features targeted for protection (Botsford *et al.*, 2003; Moffitt *et al.*, 2010). Consequently, this is seen as a priority research area with potential to inform global marine conservation planning efforts (Gaines *et al.*, 2010a), especially as the current global extent, distribution, sizing and spacing of MPAs is considered vastly inadequate (Spalding *et al.*, 2008; Wood *et al.*, 2008). However, up until recently, no tools existed that allowed us to incorporate MPA size constraints into the MPA design process or within a systematic conservation planning framework. This paper therefore represents the first study to investigate the potential trade-offs of setting different MPA size constraints when identifying priority areas for conservation. In this section we first discuss how MPA size constraints influence the overall spatial characteristics and connectivity of MPA networks, and its impact on stakeholders. We then discuss the broader implications of our findings for MPA network design.

4.5.1 Marine protected area size

The results of this study show that including MPA size constraints had little impact on the overall location of priority areas in English waters when compared to the original Marxan MPA network portfolios (**Figure 4.4**). However, increasing the minimum MPA size constraint did have a significant impact on the spatial characteristics and cost of MPA network portfolios. For example, increasing MPA size constraints resulted in portfolios that were slightly larger and more costly to stakeholders, as increasing the size of MPAs led to an increase in the selection of planning units that were more important for fisheries in both inshore and offshore waters (**Figure 4.5**; **Figure 4.6**). This is because MPAs identified by Marxan were generally located in low cost areas, and so MinPatch tended to remove planning units that were more fragmented and either (i) create new MPAs in more expensive regions, or; (ii) add planning units at the more expensive edges of the large patches (**Figure 4.1**; **Figure 4.4**). Therefore, any planning units in these locations that were rarely selected by Marxan would end up with a much higher selection frequency in MinPatch

(**Figure 4.10**). This led to MPA network portfolios that were comprised of fewer and larger MPAs, which were increasingly located in inshore waters (**Figure 4.6**).

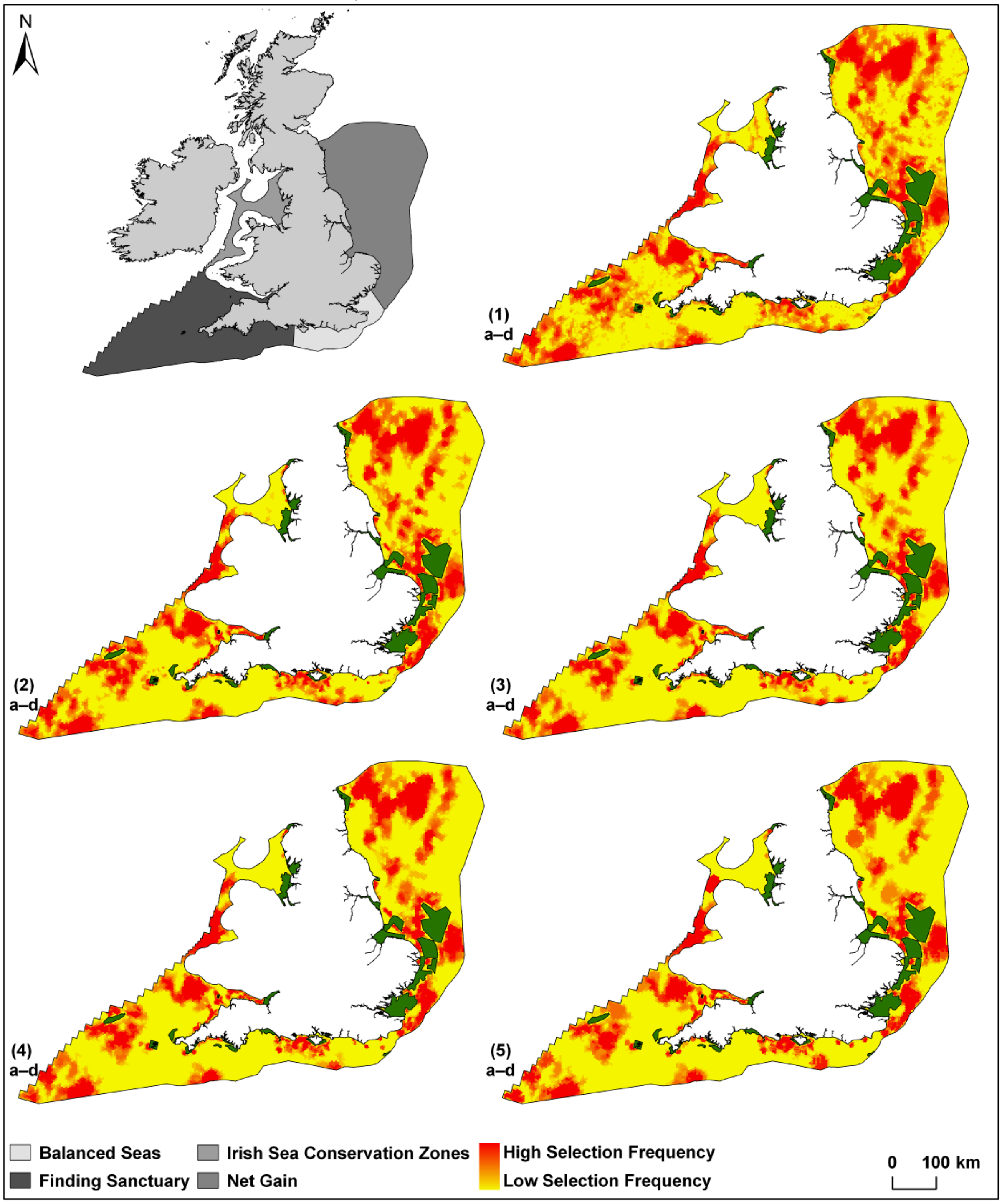

Figure 4.10 Planning unit selection frequency, which is based on the number of times each planning unit appeared in the 200 MPA network portfolios produced by Marxan (Scenario 1) and MinPatch (Scenario 2 – 5). See **Table 4.2** for details of each MPA size constraint scenario.

However, this increase in portfolio area in inshore waters could be attributed to the fact that planning unit costs were based on VMS records. This is because VMS data are based on vessels that are > 15 m in length and so under-estimates opportunity costs in inshore waters, where small-scale fisheries predominate (Lee *et al.,* 2010).

In addition, increasing MPA size constraints reduced connectivity between neighbouring MPAs (**Figure 4.8**). This is important because research has shown that 81% of species sampled in UK waters typically move less than 10 km after reaching maturity, whereas 11% move between 10 and 100 km as adults (Roberts *et al.,* 2010). Thus, our results show that increasing MPA size constraints produces networks that would limit dispersal between MPAs for many species, and especially for species that disperse 10 km or less, because increasing MPA size tends to reduce the number of MPAs in the network and so increase the distance between them (**Figure 4.9**).

4.5.2 Marine protected area network design

In the context of MPA network design, this study suggests that MPA networks comprised of fewer and larger MPAs will have a bigger impact on stakeholders in terms of opportunity costs. However, research has shown that smaller MPAs are more expensive to establish, reflecting economies of scale (McCrea-Strub *et al.,* 2011), and have higher annual running and management costs per unit area (Balmford *et al.,* 2004; Ban *et al.,* 2011). Therefore, given that MPAs are often seen as an investment of public resources, and their effectiveness depends on user compliance (Sumaila *et al.,* 2000; Sanchirico *et al.,* 2002), MPA networks containing fewer and larger MPAs may present a more financially viable management option. This could be particularly important in Europe where MPA size is likely to have an important bearing on enforcement costs as Member States share access to the same resources (EC 2008a). Consequently, reducing the number of MPAs in a network may help ensure adequate compliance from stakeholders operating in other Member States marine jurisdictions. In addition, MPA networks comprised of fewer and larger MPAs would reduce the length of MPA edge, potentially reducing the ecological impacts associated with fishing the line (Vandeperre *et al.,* 2011), where there is often a concentration of fishing effort around MPA boundaries (Murawski *et al.,* 2005; Kellner *et al.,* 2007; Stelzenmueller *et al.,* 2008).

However, if increasing the minimum size of MPAs in a network has several management benefits it is important to understand how this would influence connectivity for species. Our results suggest that increasing MPA size would reduce connectivity for species with small dispersal distances (**Figure 4.8**) because it reduces the number of MPAs in a network and so increases the distance between neighbouring MPAs (**Figure 4.4**; **Figure 4.5**). This implies that MPA networks with no or low size constraints could be more effective at ensuring the connectivity between MPAs for species with a range of dispersal distances (**Figure 4.9**), such as those that inhabit UK waters (see JNCC & Natural England 2010; Roberts *et al.,* 2010). However, research indicates that small MPAs may not be able to support self-sustaining populations that are large enough to persist, especially for mobile species, which may move beyond MPA boundaries (Botsford *et al.,* 2001; Roberts *et al.,* 2003a). In contrast, large MPAs are more likely to support viable populations that are replenished from local reproduction within their boundaries, providing they are of sufficient size (Roberts *et al.,* 2010). For example, studies of European MPAs have shown that larger MPAs are more effective than smaller MPAs at increasing the abundance, biomass, diversity and density of target species (Claudet *et al.,* 2008; Vandeperre *et al.,* 2011).

Given the enormous variability in movement and dispersal distances for species, the choice of MPA size represents an inherent compromise that could create winning and losing species (Gaines *et al.,* 2010b). However, the results of this study suggest that MPA networks comprised of moderately size MPAs that are distributed throughout the planning region are more likely to promote better connectivity for species with small dispersal distances (e.g. Scenario 3; **Figure 4.9**), as MPA size has very little impact on connectivity between MPAs for species with large dispersal distances (**Figure 4.7a**; **Figure 4.7b**). These findings are consistent with recommendations from similar studies which have also shown that moderately sized MPAs with moderate spacing are more likely to: (i) ensure population persistence for species with small dispersal distances inside

MPA boundaries; and, (ii) enhance connectivity between neighbouring MPAs for species with small to large dispersal distances (Gaines *et al.*, 2010b; Roberts *et al.*, 2010).

Nonetheless, the optimal size of MPAs is also likely to be dependent on how MPAs are managed and enforced (Roberts *et al.*, 2010). For example, no-take MPAs that restrict all activities will support more viable populations than partially protected areas, by creating more natural population structures (characterised by age, gender or individual size), leading to increased breeding success and recruitment to exploited areas, as larger older individuals are often more highly fecund (Jennings 2000; Birkeland & Dayton 2005). Moreover, higher reproductive outputs mean that strict no-take MPAs will potentially remain connected by exchange of offspring and larvae over larger distances, compared to MPAs that offer only partial protection (Roberts *et al.*, 2010). Consequently, a partially-protected MPA network will need to be more closely spaced and have larger MPAs compared to strict no-take MPA networks. However, given the dynamic nature of marine ecosystems, further work is required to evaluate what impact MPA size has on population persistence for a set of species under different MPA management scenarios.

4.6 CONCLUSION

MPA size and spacing guidelines are clearly a simple and useful way to begin the MPA network design process. However, the complexity of the process means that conservation planners should approach the design of MPAs as an iterative process, calibrating the optimal design by investigating the trade-offs of variations in size on the overall objectives of the MPA network. Here we demonstrate an analysis that could inform such an approach by using conservation planning software to identify the trade-offs with establishing different size MPAs in English waters. We show that increasing the minimum size of MPAs resulted in networks that were slightly larger and less fragmented, but more costly to stakeholders. We also show that increasing the size of MPAs meant that networks contained fewer MPAs that were spaced further apart, thus reducing potential connectivity for species that move or disperse short distances. Consequently, there is a clear trade-off between the impact on stakeholders and meeting ecological criteria that is mediated by the interaction of the size, number and distribution of MPAs in a network. These results therefore highlight the importance of testing the impact of applying MPA size constraints before making recommendations about their adoption in MPA network design.

4.7 ACKNOWLEDGEMENTS

We would like to acknowledge the following organisations and sources: Cefas for VMS data on the spatial distribution of fishing effort in UK waters, the UK Hydrographic Office for UK territorial sea limits and boundary data, and JNCC and Natural England for providing European seabed habitat data and protected area data. We are also grateful to Sue Wells of Balanced Seas and Natural England for support regarding interpretation of aspects of the Marine Conservation Zone Ecological Network Guidance.

CHAPTER 5

EXPLORING THE POTENTIAL ROLE OF SPATIAL MARINE ZONING FOR CONSERVATION AND SUSTAINABLE FISHERIES USING AN ECOSYSTEM MODEL

Kristian Metcalfe[a], Sandrine Vaz[b], George Engelhard[c], Maria-Ching Villaneuva[b], Robert J. Smith[a], and Steve Mackinson[c]

[a] Durrell Institute of Conservation and Ecology (DICE), University of Kent, Canterbury, Kent, CT2 7NR, UK

[b] Institut Français de Recherche pour l'exploitation de la Mer (IFREMER), Laboratoire Ressources Halieutiques, 150 quai Gambetta, BP699, 62321 Boulogne-sur-Mer, France

[c] The Centre for Environment, Fisheries and Aquaculture Science (CEFAS), Pakefield Road, Lowestoft, NR33 0HT, UK

5.1 ABSTRACT

Marine ecosystems are under increasing pressure from a diverse range of threats. Many national governments have responded to these threats by developing marine protected area (MPA) networks. Whilst there is increasing recognition that marine spatial planning (MSP) provides an important framework to deliver ecosystem-based management of marine resources, very little research has addressed how MPA networks should be managed. The majority of research has focused on developing no-take MPA networks, whereas the potential for broader classes of MPA management have received far less attention. Therefore, to investigate the potential trade-offs associated with adopting different spatially explicit MPA management strategies, we used Marxan and Ecopath with Ecosim software packages to determine: (i) if strict no-take MPA networks justify the cost of their implementation; or, (ii) whether MPA networks comprised of multiple zones with different management restrictions could achieve similar results. We show that broader classes of spatial management based on zoning fleet access and gear restrictions can also have conservation and fisheries benefits, which is important considering this approach is less politically contentious than strict no-take MPA networks. For example, after 50 years a 100% limited-take MPA network increased exploited ecosystem biomass by 3.41% and fisheries catches by 2.07%, relative to a system with no MPAs. However, we also show that if MPA networks are to ensure the sustainable use of fisheries they should be comprised of at least 60% no-take zones, and that a 100% no-take MPA network would increase ecosystem biomass by 14.01% and fisheries catches by 39.5%. We also show that exploited catches recover 6 times as quickly in 100% no-take MPA networks compared to 100% limited-take MPA networks. Finally, we demonstrate that these tools provide a useful policy screening method that can help evaluate the impacts of proposed MPA networks and management strategies prior to implementation.

Keywords: *Ecopath, Ecospace, English Channel, Europe, Marine protected areas, Marxan, Marxan with Zones, Systematic conservation planning*

5.2 INTRODUCTION

In the last decade there has been a rapid increase in our understanding of human impacts on coastal and marine ecosystems (Lubchenco *et al.*, 2003), resulting in international agreement on the need for increased protection (Wood 2011). In response, many national Governments are establishing marine protected area (MPA) networks, partly based on the increasing recognition that marine spatial planning (MSP) provides an important framework to improve decision making and deliver an ecosystem based management (EBM) of marine resources (Gaines *et al.*, 2010b; Halpern *et al.*, 2010). The overarching goal of MSP and EBM is to ensure that management decisions do not adversely affect marine ecosystem functioning and productivity (Rosenberg & McLeod 2005). In particular, there is growing consensus that no-take MPAs, which effectively restrict all activities, and partially protected areas, which restrict certain activities, can provide benefits for biodiversity and fisheries (e.g. Roberts *et al.,* 2001; Halpern & Warner 2002; Gell & Roberts 2003; Lester & Halpern 2008).

Nonetheless, MPA networks remain controversial as spatial management tools, particularly as fishermen often oppose restrictions on where and how they fish (Smith *et al.,* 2010a; Abbott & Haynie 2012; Rassweiler *et al.,* 2012). In this context, it is widely recognised that conservation planners need to explicitly include human dimensions, and account for opportunity costs and other social and socio-economic impacts (Arkema *et al.,* 2006; Ban & Klein 2009). This has led to the widespread adoption of systematic conservation planning (Kareiva & Marvier 2012), a target driven approach that aims to achieve the representation and long-term persistence of biodiversity, whilst minimising impacts on different stakeholders (Margules & Pressey 2000). Applications of this approach to real-world conservation planning are often supported by spatial prioritisation software tools (e.g. C-Plan, Marxan, Marxan with Zones, Zonation), which have been developed to help identify priority areas, and design MPA networks with different types of management and protection zones (Moilanen *et al.,* 2009).

Whilst these tools increasingly consider socio-economic and biodiversity factors (Klein *et al.,* 2010) they only address short term goals; that is to identify priority areas that minimise 'current' impacts on stakeholders. Thus, the major limitation with existing spatial prioritisation approaches is that they do not allow conservation practitioners and policy makers to consider the 'long-term' impacts associated with different forms of MPA management prior to implementation. In addition, even though most marine conservation plans involve some form of zoning (e.g. Klein *et al.,* 2010; Malcolm *et al.,* 2012; Grantham *et al.,* 2013) it is often unclear what impact such MPA networks will have on marine ecosystem functioning, or whether they are likely to be successful fisheries management tools (Vandeperre *et al.,* 2011). For example, closures and/or gear restrictions may induce shifts in the distribution of fishing effort and the targeting behaviour of fisheries with potential cascading effects (Abbott & Haynie 2012; Rassweiler *et al.*, 2012). Moreover, we still lack experience with MPAs implemented at large spatial scales, particularly in temperate waters (Roberts *et al.,* 2005), and scientific evidence used to support their designation are often based on studies that focus on: (i) habitat specific species associated with coral and temperate rocky reefs, which are relatively small in scale and easier to protect; and, (ii) less mobile or sedentary species that are more likely to benefit from the exclusion of human activities (e.g. Murawski *et al.,* 2000; Claudet *et al.,* 2006; McCook *et al.,* 2010).

Thus, there are still some doubts as to whether MPAs will benefit many temperate and commercially important species that are widespread across a variety of habitats, exhibit entirely different life history characteristics, or are highly mobile and move considerable distances each year (Hilborn *et al.,* 2004; Kaiser 2005). Consequently, there is increasing demand for information that could provide policy makers with a baseline from which to evaluate the potential impact of establishing different types of MPAs, and their effects on fisheries and ecosystem functioning (Agardy *et al.,* 2011; Claudet *et al.,* 2011; Vandeperre *et al.*, 2011). More importantly, stakeholders want to know how rapidly changes will occur after protection, even if natural variability can be difficult to predict (Babcock *et al.,* 2010).

Fortunately, over the last decade considerable effort has been directed at the development of multispecies assessment models, the most widely used of which is Ecopath with Ecosim (Pauly *et al.*, 2000; Christensen & Walters 2004a). These models have been used to analyse a variety of complex issues including: (i) how fisheries and environmental changes have affected marine ecosystems; (ii) the impact of establishing MPAs; and, (iii) fisheries management strategies (e.g. Mackinson & Daskalov 2007; Araujo *et al.*, 2008; Lozano-Montes *et al.*, 2012). However, no such studies have explored how MPAs developed to meet conservation goals could be managed in the long-term to support sustainable fisheries. Here we address this issue by using spatial prioritisation software and an ecosystem model to explore the impacts of different types of MPA network management goals on biodiversity, ecosystem health and fisheries. This involved using Marxan and Marxan with zones to identify MPA networks with different proportions of “no-take” and “limited-take” zones, and using Ecospace to predict how these different scenarios would affect ecosystem biomass, fisheries catches and trophic level of landings.

5.3 METHODS

5.3.1 Study area and policy context

This study was carried out in the English Channel, a shallow epicontinental sea located in the temperate North East Atlantic (Delavenne *et al.*, 2012). The English Channel encompasses a wide range of ecological conditions and is comprised of the deeper western channel (> 50 m) and shallower eastern channel (< 50 m), which can be regarded as two distinct ecosystems due to their markedly different oceanographic and ecological characteristics (Vaz *et al.*, 2007). This study focused on the eastern English Channel (EEC), a bio-geographical transition zone between the warm Atlantic oceanic system and the boreal North Sea (**Figure 5.1**). The EEC is a key area for tourism, shipping, energy production, and aggregate extraction (Metcalfe *et al.*, 2013a), and also supports one of the richest commercial fisheries in Europe in terms of both the abundance and diversity of species (Carpentier *et al.*, 2009). In addition, the EEC contains a number of important nursery, spawning areas and migratory routes that are linked to specific environmental characteristics (Martin *et al.*, 2009).

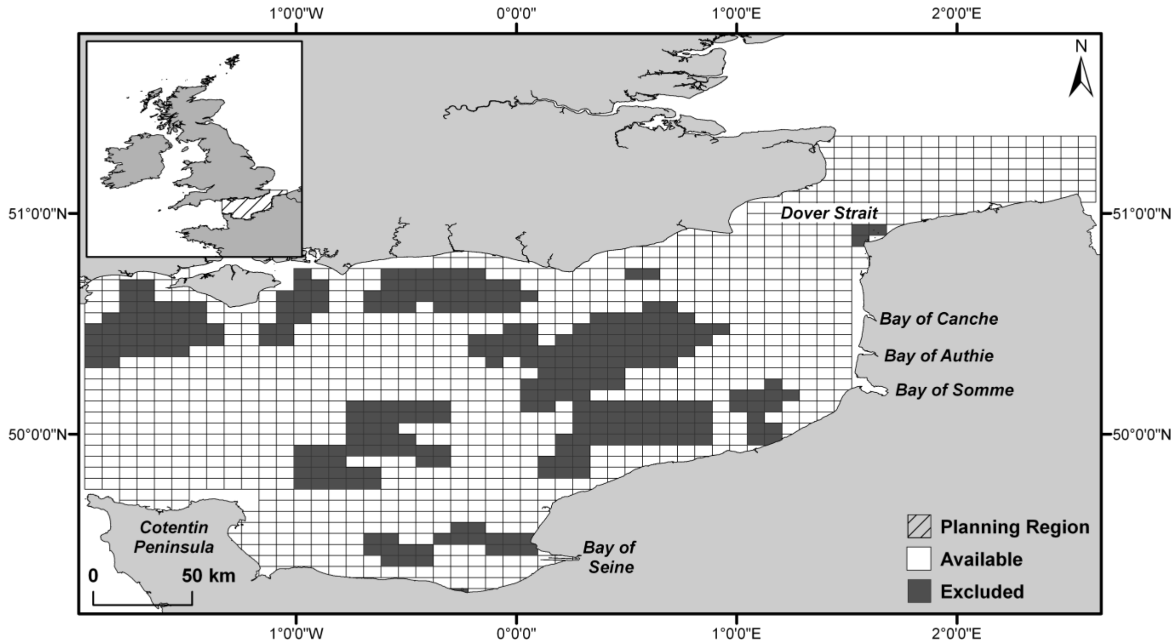

Figure 5.1 Eastern English Channel planning region, showing the location of the available, and excluded planning units that contained areas allocated to existing and proposed marine aggregate dredging and offshore wind farms.

Consequently, the biodiversity of the EEC is considered both important and threatened, and is therefore subject to several MPA designation projects that are being implemented as part of Member State commitments to a number of international and regional obligations (Metcalfe *et al.*, 2013b). For example, France is currently developing a MPA network in the Three Estuaries Region (Bay of Somme, Authie and Canche; **Figure 5.1**) and the EEC is also the focus of the Balanced Seas project which is one of four regional Marine Conservation Zone (MCZ) projects in the UK that has recommended MPAs for inshore and offshore waters in south-east England (Metcalfe *et al.*, 2013a). Moreover, both the UK and French Governments are committed to designating a proportion of these MPA networks as no-take, with the remaining sites being managed for multiple uses (e.g. Jones 2012). However, no-take MPAs are unpopular with members of the fishing sector (Jones 2009). Therefore, given that MPAs are being implemented in this region this analysis aims to inform future management decisions as there is currently a lack of data to guide policy makers about: (i) how much of these networks should be designated as no-take; and (ii) the potential role of alternative forms of management such as spatial marine zoning, particularly in temperate waters with large commercial fisheries (Rees *et al.*, 2013).

5.3.2 Designing marine protected area networks

In this study we adopted a systematic conservation planning approach to MPA network design because it provides a transparent and scientifically defensible platform for exploring different management strategies (Margules & Pressey 2000). This involved: (i) compiling biodiversity data on important species and habitat types, known collectively as conservation features; (ii) setting representation targets for how much of each feature should be protected; (iii) dividing the study area (hereafter referred to as the planning region) up into a number of planning units; (iv) calculating the amount of each conservation feature found in each planning unit; (v) assigning a cost value to each planning unit, which can be a measure of any aspect of the planning unit, such as its area, the risk of being affected by anthropogenic impacts, financial value, or opportunity costs; and (vi) running computer software to identify portfolios of planning units that are required to meet representation targets, reduce fragmentation levels and minimise planning unit costs.

Marxan and Marxan with Zones software use a simulated annealing approach to identify near-optimal portfolios of planning units that, when combined, minimise total costs (Ball *et al.*, 2009; Watts *et al.*, 2009). Both software packages use the same system for measuring costs which are calculated as the summed planning unit costs, a boundary cost that reflects the boundary length of the portfolio edge and costs for not meeting the conservation targets. In our analyses all portfolios m*et al*l the targets so the cost represents the sum of the planning unit and boundary costs.

5.3.2.1 Conservation feature data

To represent broad-scale patterns of biodiversity we used a marine habitat map that was modelled using physical and environmental data (Coggan & Diesing 2011). This map identified seventeen habitats (see **Table S5.1** in Supporting Information) and is based on the European Nature

Information System (EUNIS) level 3 and 4 habitat classification hierarchy developed by the European Environment Agency (EEA 2006). The EUNIS level 3 habitats are broken down into three habitat types and coded as follows: infralittoral rock (A3.x); circalittoral rock (A4.x); and sublittoral coarse sediment (A5.x) which was divided into its finer-scale EUNIS level 4 habitats (A5.xx). In addition, we also compiled fine-scale data on the distribution of 7 priority habitats and 34 species (comprised of 2 algae, 8 invertebrates, 14 seabirds and 10 fish) that are listed by OSPAR (OSPAR 2008c) the EU Birds and Habitats Directives (EC 1979; EC 1992), and/or in national legislation and governmental initiatives (**Table S5.1**). For eight of the fish species we used distributions modelled using regression quantiles which use abundance data to identify potential habitat areas where environmental conditions are suitable (Vaz *et al.*, 2008; Carpentier *et al.*, 2009).

5.3.2.2 Conservation feature targets

Given each countries obligation to fulfilling commitments as a signatory to the CBD and OSPAR (Metcalfe *et al.*, 2013b), and to ensure balanced representation in the planning region, we set targets for each feature in both UK and French waters (**Table S5.1)**. Targets for the broad-scale EUNIS level 3 habitats were based on the species-area relationship approach and designed to reflect the minimum proportion of habitat area required to represent 80% of species known to occur in each habitat type (JNCC & Natural England 2010). However, as no targets have been developed for EUNIS level 4 habitat types we based them on the minimum proportion of habitat area required to represent 80% of species for their parent EUNIS level 3 habitat types (**Table S5.1)**.

Setting targets for species was more problematic because the distribution maps for many of these species are affected by sampling bias. Thus, we decided to convert the point record data into presence maps by including only one record per planning unit, and set targets so that there should be a minimum of three replicates spread throughout the planning region (JNCC & Natural England 2010), with the exception of those features with < 3 records whose targets were based on the total number of records (**Table S5.1)**. To represent habitat areas that are potentially suitable for each fish species we used the ‘Zonal Statistics’ function in ArcGIS to calculate the mean habitat suitability value for each planning unit (ESRI 2008). However, as no representation targets have been developed for these species, we set a target of 10% of the total sum of the habitat suitability values across all planning units for each species. This value was determined from several studies in the scientific literature that have applied targets to features based on recommendations in policy (e.g. Klein *et al.,* 2008a; Ban *et al.,* 2009; Delavenne *et al.,* 2012).

5.3.2.3 Defining the planning region and planning units

We divided the planning region (**Figure 5.1**) up into a number of 31.4 km^2 planning units (n = 1,180) based on a system developed for an existing Ecospace model of the EEC (Daskalov *et al.*, 2011) using the ET GeoWizards Extension in ArcGIS (Tchoukanski 2012). To minimise conflict with other marine resource users we excluded planning units that contained existing and proposed marine aggregate dredging areas and offshore wind farms (n = 281; **Figure 5.1**), and then calculated the amount of each conservation feature in each planning unit using the Conservation Land-Use Zoning (CLUZ) ArcView Extension (Smith 2004a).

5.3.2.4 Developing planning unit cost data

Given that minimising socio-economic impacts is a core objective in MPA design (Ban & Klein 2009) we collected data on the spatial distribution of fishing effort for eight fisheries fleets classified according to different gear types: beam trawl; demersal otter trawl; dredges; pelagic trawl; hooks and lines; nets; seines; and traps and pots. These data were derived from an analysis of vessel monitoring system (VMS) data for vessels > 15 m in length recorded in the EEC in 2007 – 2008, and included the level of fishing effort reported as the time spent fishing (in hours) per unit area (0.05^o square degrees) for each fleet. To reflect the relative value of areas for each fleet we used the ‘Zonal Statistics’ function in ArcGIS to calculate the cost for each planning unit based on the mean number of hours fished (ESRI 2008). Data on smaller inshore vessels were not included in our analysis because high quality spatially explicit data for vessels < 15 m are unavailable for UK and French waters.

5.3.2.5 Running the conservation assessment

Marxan and Marxan with Zones software packages have the same functionality but Marxan can only include or exclude a planning unit from a proposed MPA network and so implicitly assumes two zones; reserved or not reserved. In contrast, to reflect the range of management options being considered as part of a conservation plan, Marxan with Zones is able to set targets for different management zones and specify how costs vary for a particular planning unit depending on the zone to which it is assigned (Ball *et al.,* 2009; Watts *et al.,* 2009).

However, given that both Marxan and Marxan with Zones can identify several optimal solutions that similarly meet targets (Ball *et al.,* 2009), and that we wanted to model the impact of different MPA management strategies, it was important to ensure that the different MPA network designs were comparable. Moreover, it has also been recognised that the 'best' solution identified by these software packages, which is the portfolio of planning units that meets the targets at the lowest cost may not identify priority areas that are suitable for all stakeholders. Therefore, to: (i) ensure that conservation planning outputs remain relevant within a real world context (Ferrier & Wintle 2009); and, (ii) illustrate to stakeholders that different MPA networks can be similarly effective in meeting conservation goals, we first used Marxan to identify five no-take MPA network portfolios that were spatially different. We then used Marxan with Zones to identify the most efficient way in which these five portfolios could be allocated to two different management zones based on meeting targets and minimising costs. We defined the two zones as follows: (i) 'no-take' designed to exclude all fisheries; and (ii) 'limited-take' designed to exclude fleets using active gears (beam trawl, demersal otter and dredges), because part of the reason for stock declines is that these active gears damage, degrade and destroy essential habitats (Roberts *et al.,* 2005). We set the cost of establishing no-take MPAs as being the combined cost of all eight fisheries, whereas the cost for limited-take MPAs was based on fishing effort costs for the three excluded fleets.

This approach involved running Marxan 500 times using simulated annealing followed by iterative improvement to identify portfolios of planning units that met the targets whilst minimising planning unit and boundary costs, where each run consisted of two million iterations. We used a boundary length modifier (BLM) value of 0.3 as this represented an acceptable trade-off between minimising portfolio fragmentation and cost (Stewart & Possingham 2005), and identified the five no-take MPA portfolios using an approach developed by Linke *et al.*, (2011). This involved identifying five “clusters” of solutions, based on solution similarity, and selecting the portfolio with the lowest cost found in each cluster by performing a multivariate analysis in R (R Development Core Team 2008).

We then used Marxan with Zones to investigate six scenarios for each of the five portfolios by increasingly allocating 20% (up to 100%) of the targeted amount of each conservation feature to the limited-take zone for each portfolio, with the remaining proportion targeted for protection allocated to the no-take zone. However, we did not specify zone targets for fourteen features which had a restricted distribution in the Marxan portfolios (**Table S5.1**), and allowed Marxan with Zones to allocate them to either the no-take or limited-take zone according to the efficient allocation of resources by the optimising algorithm. We ran Marxan with Zones 500 times using simulated annealing followed by iterative improvement technique, where each run consisted of two million iterations, and used the ‘planning unit lock’ and ‘planning unit zone’ feature to ensure that only planning units selected in each Marxan portfolio would be allocated to the two different zones (Watts *et al.*, 2008a). To control for the level of fragmentation within each portfolio we used a BLM value of 1 and a zone boundary cost that ensured moderate spatial clumping between zones (Watts *et al.*, 2008b). In addition, Marxan with Zones allows the user to control for the different management objectives associated with each zone and how they contribute to meeting the overall target for each feature (Watts *et al.*, 2008b). Therefore, we specified the contribution rate of the features allocated to the no-take and limited-take zone to be the same, so that they would contribute equally towards meeting targets.

5.3.3 Modelling the impact of marine protected area networks

To investigate the impacts of different proportions of no-take and limited-take MPAs in the EEC we used Ecopath with Ecosim version 5.1 (Christensen *et al.,* 2005), which is a suite of modelling tools that can be used to analyse how changes might affect the structure and functioning of an aquatic ecosystem (Pauly *et al.,* 2000). This software is comprised of three main components: (i) Ecopath; (ii) Ecosim; and (iii) Ecospace (Pauly *et al.,* 2000; Christensen & Walters 2004a), with critical analyses of these tools detailed in Aydin (2004), Plaganyi and Butterworth (2004) and Fulton and Smith (2004). Ecopath represents a static mass-balanced snapshot of a trophic network describing the average flows of mass and energy between species during a specified period of time. Ecosim represents a time-dynamic version of Ecopath that can be used to simulate the ecosystem effects of fishing mortality changes and environmental forcing over time. However, we used an existing Ecospace model (Daskalov *et al.,* 2011); a policy evaluation tool based on spatially explicit simulation of ecosystem dynamics which can be used to investigate the impact of establishing MPAs (Pelletier & Mahevas 2005). The spatial model developed in Ecospace was based on an existing Ecopath mass-balance model for parameterisation (Villanueva *et al.,* 2009), and Ecosim model that was calibrated by comparing model outputs to time series and spatial data from scientific surveys and fish stock assessments (Daskalov *et al.,* 2011).

5.3.3.1 Ecopath model

The Ecopath model was constructed for the period 1995 – 1996, and contained data on 51 functional groups (Villanueva *et al.,* 2009). Each group is represented by a particular species or group of species with similar characteristics (e.g. demography, dietary preference, spatial distribution, age and size) and included two marine mammal, one seabird, twenty-nine fish, fifteen invertebrate, and two primary producer groups (**Table S5.2**). The functional groups for large pelagic fish were split into juveniles and adults according to Walters *et al.* (1997) and two non-living groups were included to represent discard and detritus (Villanueva *et al.,* 2009). Mass-balance was achieved by adjusting biomass, diet matrix and consumption and mortality parameters (Villanueva *et al.,* 2009).

In addition, this model included data on eight fisheries fleets (classified according to different gear types: beam trawl; demersal otter trawl; dredges; pelagic trawl; hooks and lines; nets; traps and pots; and other), which were defined by their annual extraction of biomass from the functional groups (Villanueva *et al.*, 2009). The Ecopath model also included data on the value and total catch of each functional group and the relative profitability of each fleet (**Table S5.2**; **Table S5.3**). However, to represent current fishing behaviour, and ensure that the Ecospace model reflected fishing effort data used in Marxan we added an additional fleet (seine) and updated the landings data to reflect the 2007 – 2008 distribution of catches among the fleets. These data were obtained from the International Council for the Exploration of the Sea (ICES) landing statistics for ICES area VIId, and were based on the average landings for each functional group.

5.3.3.2 Ecospace model

Ecospace simulations are structured around a cell-based format to describe the two dimensional spatial distribution of biomass and fishing effort over time (Walters *et al.*, 1999; Pauly *et al.*, 2000). Movement between adjacent cells and the distribution of biomass is driven by foraging, avoidance of predation, and intrinsic dispersal rates linked to specific habitat preferences (Walters *et al.*, 1999). Habitats and functional groups were therefore assigned to each cell according to Vaz *et al.* (2007) who differentiated between four habitats and their associated biotic communities (**Table S5.4**). Dispersal rates for each functional group (**Table S5.5**) were determined from an existing North Sea Ecospace model which were based on published movement rates, and adjusted during model calibration (Mackinson & Daskalov 2007; Daskalov *et al.*, 2011). Feeding and predation risk parameters, which determine the relative dispersal, vulnerability and feeding rates in habitats that functional groups are not assigned, were set to their default values as with the existing model (**Table S5.5**). In addition, recognising that fishing effort dynamics are important to consider when evaluating MPAs (Stelzenmueller *et al.*, 2008), we specified sailing costs for each fleet based on the location of ports, where sailing costs are calculated for each cell as relative distances to these ports (Daskalov *et al.*, 2011). We then defined the distribution of the nine fisheries by assigning which habitats a fleet may operate in (**Table S5.6**) and used the gravity model to predict the spatial distribution of fishing effort (Caddy 1975), where the proportion of total effort allocated to each cell is assumed proportional to the relative profitability of fishing in that cell (Christensen *et al.,* 2005).

To simulate the potential responses to the different forms of management we overlaid each Marxan portfolio onto the basemap and assigned which fleet can operate in each MPA according to their gear restrictions (**Table S5.6**). We then ran Ecospace over a 50 year period (at 0.083 time intervals which are equal to monthly time steps), as this allowed sufficient time for longer lived species to show any signs of recovery or decline. To examine how each management scenario affected ecosystem functioning and fisheries, we compared the overall and exploited density of ecosystem biomass and catches to baseline simulations of the EEC with no MPAs. We also examined changes in the density of overall and exploited ecosystem biomass inside and outside of MPAs. The exploited part of the system was based on more threatened higher trophic level (TL) and commercially valuable functional groups, with a TL $\geq$ 2.35 (n = 29; **Table S5.2**). This was because 43% of the ecosystem biomass occurs below this TL due to the high abundance of benthic invertebrates and molluscs in the EEC (**Table S5.2)**.

5.3.4 Management indicators

To influence management effectively, scientists need to provide policy makers with indicators that summarise a variety of complex processes in a single number (Pauly & Watson 2005; Nicholson *et al.,* 2012). Therefore, to evaluate the potential impact of each MPA management scenario we used the 'marine trophic index' (MTI), an indicator endorsed by the Convention on Biological Diversity (CBD 2004a). The MTI is a measure of the mean TL of fisheries catches (landings and discards), and is often used to assess whether a fisheries is being managed sustainably (Pauly & Watson 2005). Sustainability implies some notion of stability, thus, if the MTI decreases then it could indicate that the fisheries is not being managed sustainably, as the TL of the species it exploits keeps getting lower (Pauly *et al.,* 1998; Pauly & Watson 2005). The mean TL for fisheries catches was calculated for each year *y* as follows (**Equation 5.1**):

Equation 5.1 $$TL_y = \frac{\sum_i (TL_i * Y_{iy})}{\sum_i Y_{iy}}$$

Where TL_i is the trophic level for functional group i as estimated by Ecopath, and Y_{iy} is the catch of the functional group i in year y as provided by Ecospace (**Table S5.2**). However, based on recommendations that this should be calculated to emphasise changes in more threatened higher trophic level (TL) and commercially valuable functional groups (Christensen & Walters 2004b; Pauly & Watson 2005; Araujo *et al.*, 2008), we also calculated the MTI based on the exploited part of the system (functional groups with a TL $\geq$ 2.35).

5.4 RESULTS

5.4.1 Marine protected area network design

The planning region had a total area of 37,052 km^2; of which 28,229 km^2 was available for selection, and 8,823 km^2 was excluded from the Marxan analyses (**Figure 5.1**). The spatial distribution of the different conservation features, patterns of fishing effort and the presence of wind farms and aggregate dredging in the EEC meant that the five no-take MPA portfolios identified by Marxan shared similar spatial patterns and characteristics. Consequently, most of the planning units selected to meet targets were located in: (i) the Dover Strait; (ii) Three Estuaries region; (iii) Bay of Seine; and (iv) north of the Cotentin Peninsula in the west of the EEC (**Figure 5.1**; **Figure 5.2**).

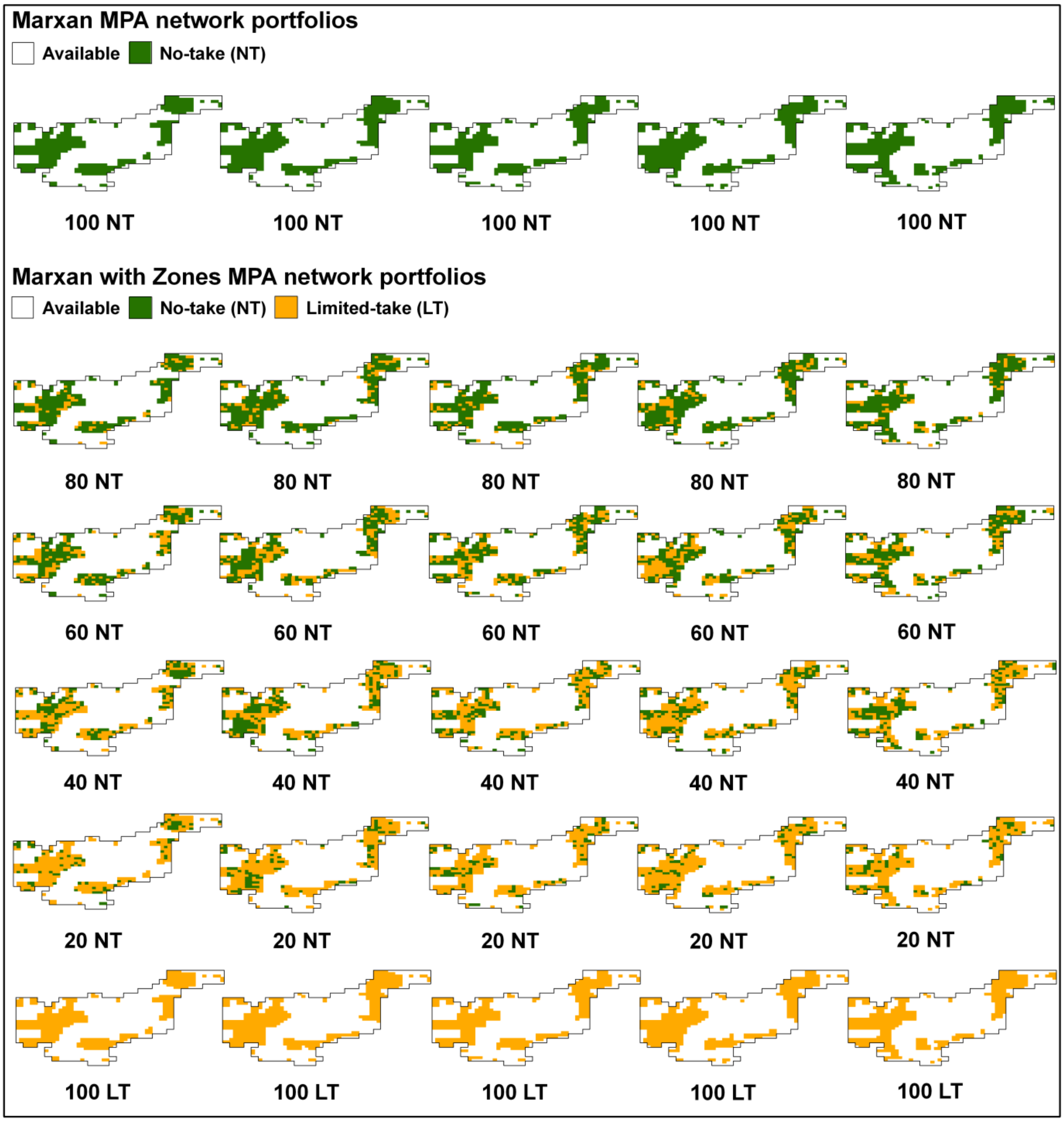

Figure 5.2 Spatial configuration of the five no-take (NT) MPA network portfolios identified by the Marxan spatial analysis (100 NT), and the five MPA network portfolios developed for each of the Marxan solutions in Marxan with Zones (80 NT, 60 NT, 40 NT, 20 NT, 100 LT) where an increasing proportion (20%) of the targeted amount of each conservation feature is allocated to the limited-take (LT) zone.

The mean area of the MPA networks allocated to the no-take zone in Marxan was 12,460 km^2, in contrast the mean area of the MPA networks allocated to the limited-take zone ranged between 2,606 km^2 and 12,460 km^2 when 20% to 100% of the features targeted for protection were allocated to the limited-take zone (**Figure 5.3a**). In addition, the mean cost of the MPA networks which is the sum of the combined planning unit costs and the boundary costs, decreased from 634,532 for a 100% no-take MPA network that excluded all fisheries to 529,049 for a 100% limited-take network that excluded active gears (**Figure 5.3b**).

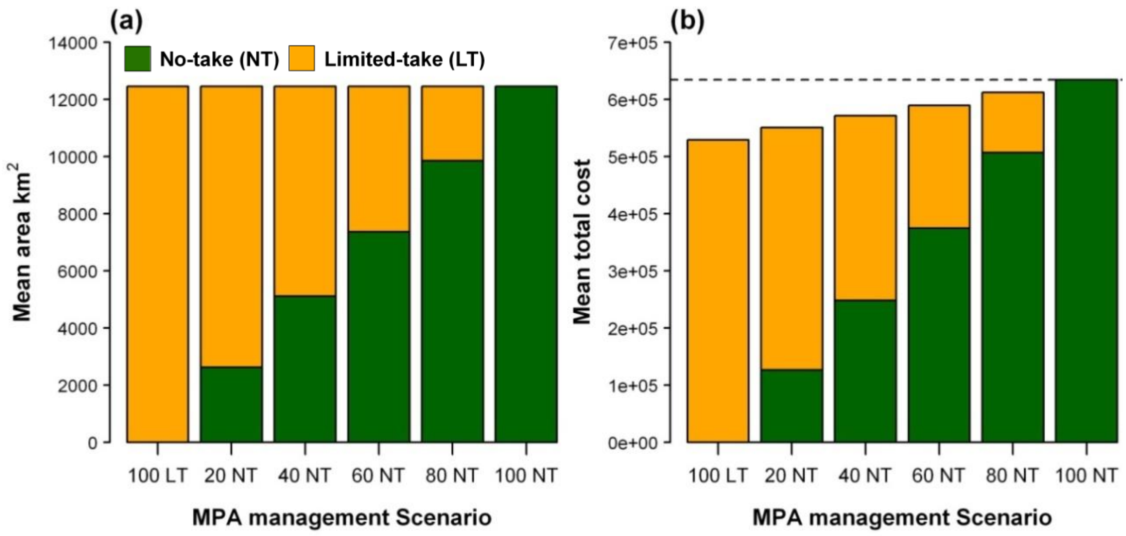

Figure 5.3 Details of mean: (a) area (km^2); and (b) cost for the MPA network portfolios identified by Marxan and Marxan with Zones.

5.4.2 Marine protected area network management

Each of the proposed MPA management scenarios had a positive impact on the overall and exploited density of ecosystem biomass compared to the baseline scenario with no MPAs (**Figure 5.4a**; **Figure 5.4b**). However, MPA networks that were comprised of a larger proportion of no-take zones had greater increases in ecosystem biomass. Nevertheless, the change in overall ecosystem biomass was smaller (**Figure 5.4a**) than the change in exploited ecosystem biomass across each MPA management scenario (**Figure 5.4b**). For example, after 50 years a 100% limited-take MPA network increased the mean density of overall and exploited ecosystem biomass by 0.19% and 3.41% respectively, relative to the baseline scenario. In contrast, a 100% no-take MPA network increased the mean density of overall and exploited ecosystem biomass by 0.85% and 14.01%. Nonetheless, the impact on each functional group varied according to the MPA management scenario (**Figure 5.5**). For example, functional groups with smaller spatial scales of dispersal (< 300 km/year) generally had greater increases in biomass when MPA networks were comprised of a larger proportion of no-take zones (**Figure 5.6**).

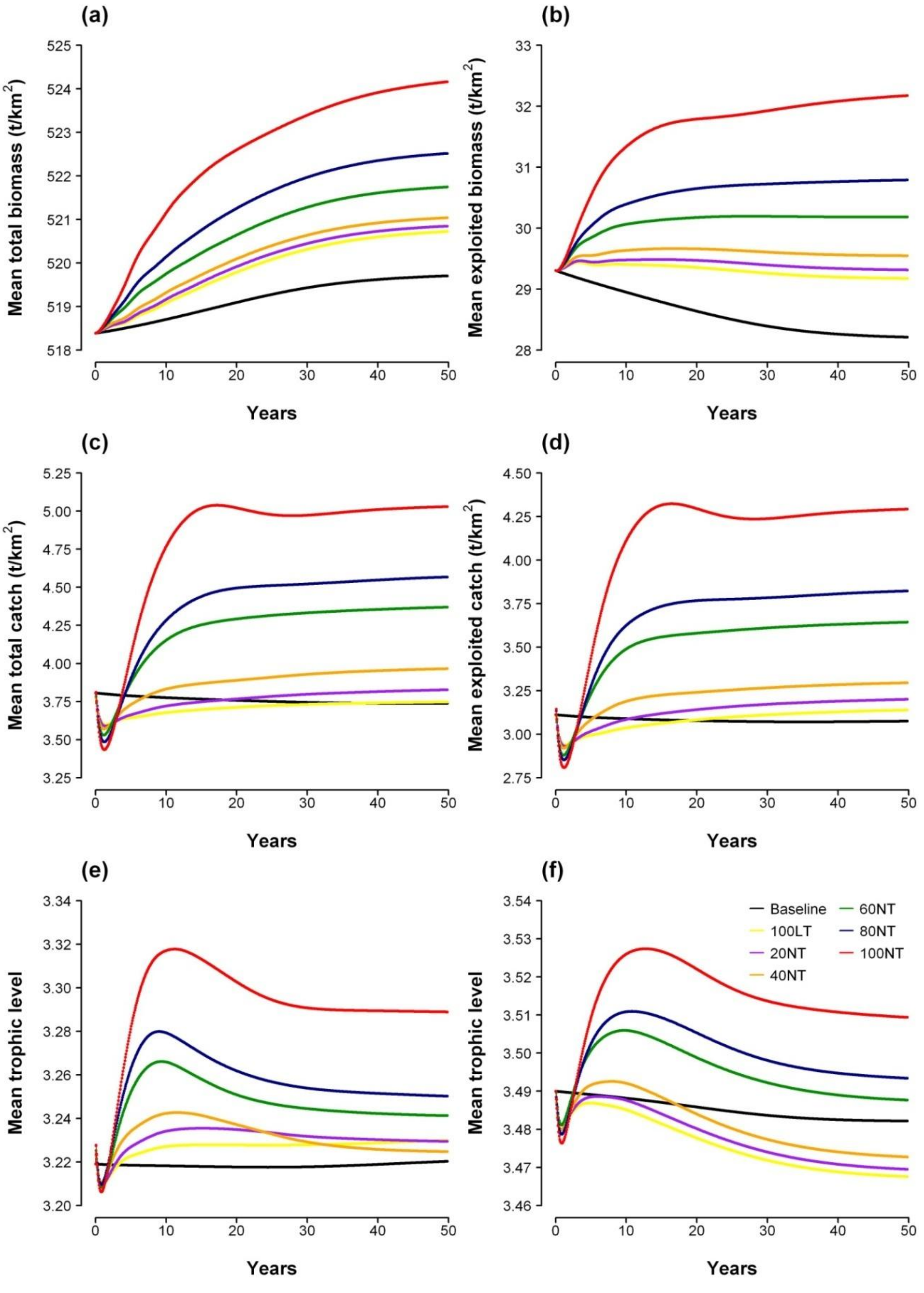

Figure 5.4 Changes over time (0 – 50 years) in: (a) mean overall ecosystem biomass (t/km^2); (b) mean exploited ecosystem biomass (t/km^2); (c) mean total catch (t/km^2); (d) mean exploited catch (t/km^2); (e) mean trophic level of overall fisheries catch; and, (f) mean trophic level of exploited fisheries catch; for each MPA management scenario relative to baseline scenario with no MPAs.

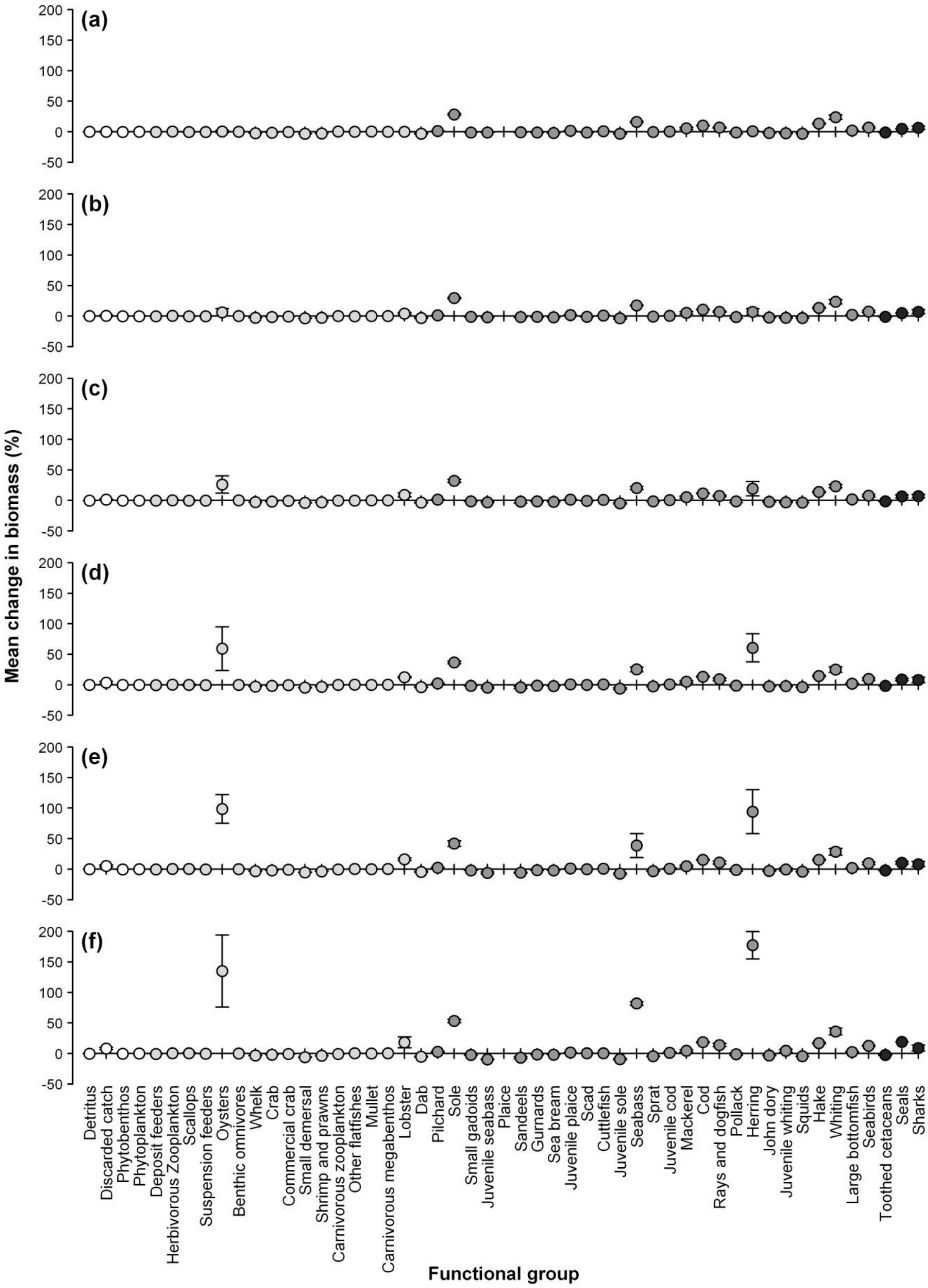

Figure 5.5 Mean percentage change (± 95% confidence intervals) in the density of biomass (t/km^2) per functional group for each MPA management scenario: (a) 100% limited-take; (b) 20% no-take; (c) 40% no-take; (d) 60% no-take; (e) 80% no-take; and (f) 100% no-take, relative to baseline scenario with no MPAs.

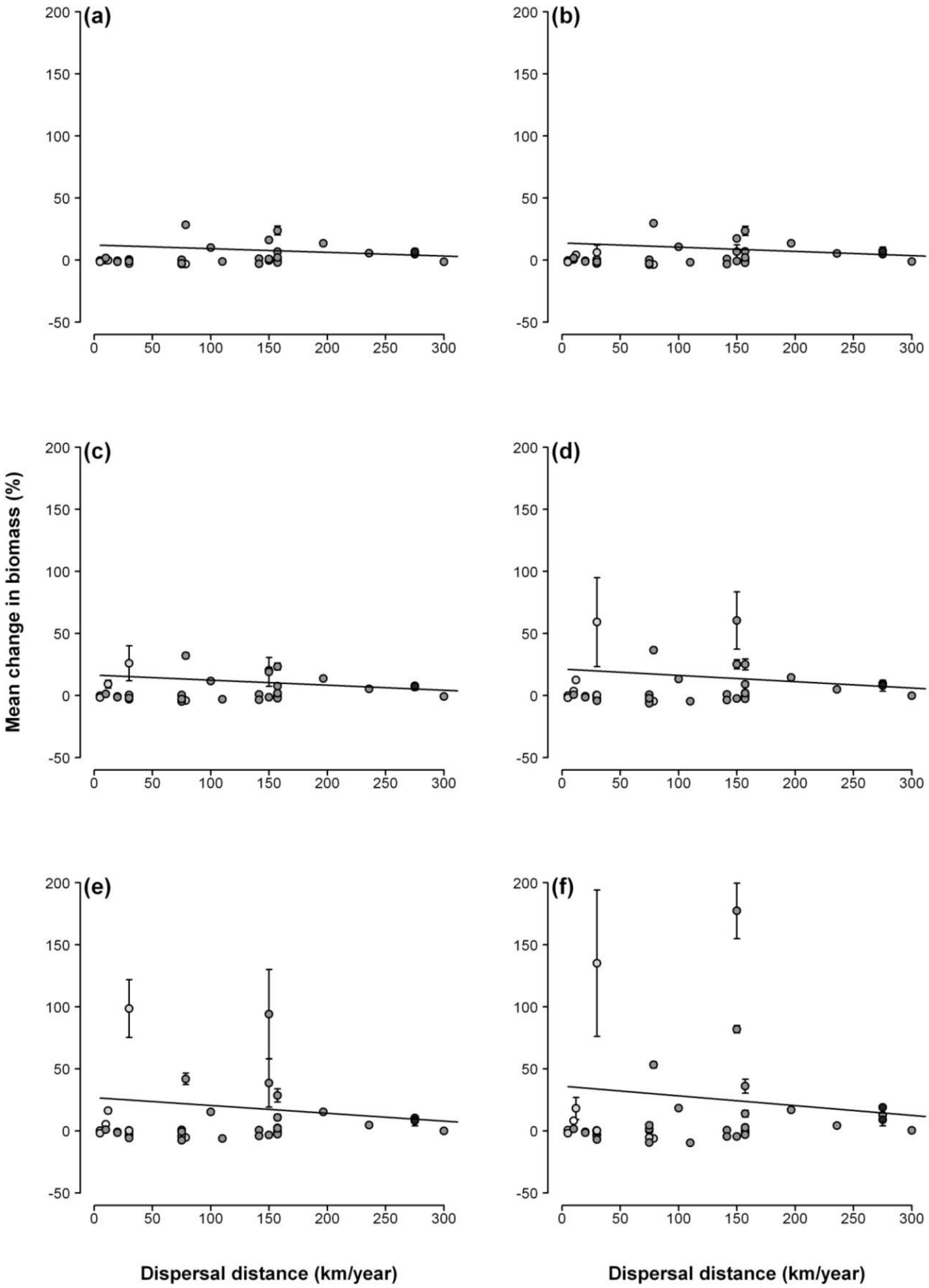

Figure 5.6 Influence of dispersal rates (km/year) on mean percentage change (± 95% confidence intervals) in the density of biomass (t/km2) of functional groups for each MPA management scenario: (a) 100% limited-take; (b) 20% no-take; (c) 40% no-take; (d) 60% no-take; (e) 80% no-take; and (f) 100% no-take, relative to baseline scenario with no MPAs.

In addition, each of the proposed MPA management scenarios had a positive impact on the density of overall and exploited ecosystem biomass inside no-take and limited-take zones (**Figure 5.7a**; **Figure 5.7b**). However, the change in ecosystem biomass was greater for MPA networks that comprised of a larger proportion no-take zones (**Figure 5.7a**; **Figure 5.7b**). For example, after 50 years a 100% limited-take MPA network increased the mean density of overall and exploited ecosystem biomass inside of limited-take zones by 0.56% and 8.60%. In contrast, the implementation of a 100% no-take MPA network increased the mean density of overall and exploited ecosystem biomass inside of no-take zones by 2.27% and 39.30%. However, even though the change in density of overall ecosystem biomass remained relatively constant outside of MPAs, increasing by approximately 0.5% across all MPA management scenarios (**Figure 5.7a**), the density of exploited ecosystem biomass outside of MPAs decreased by 5% (**Figure 5.7b**).

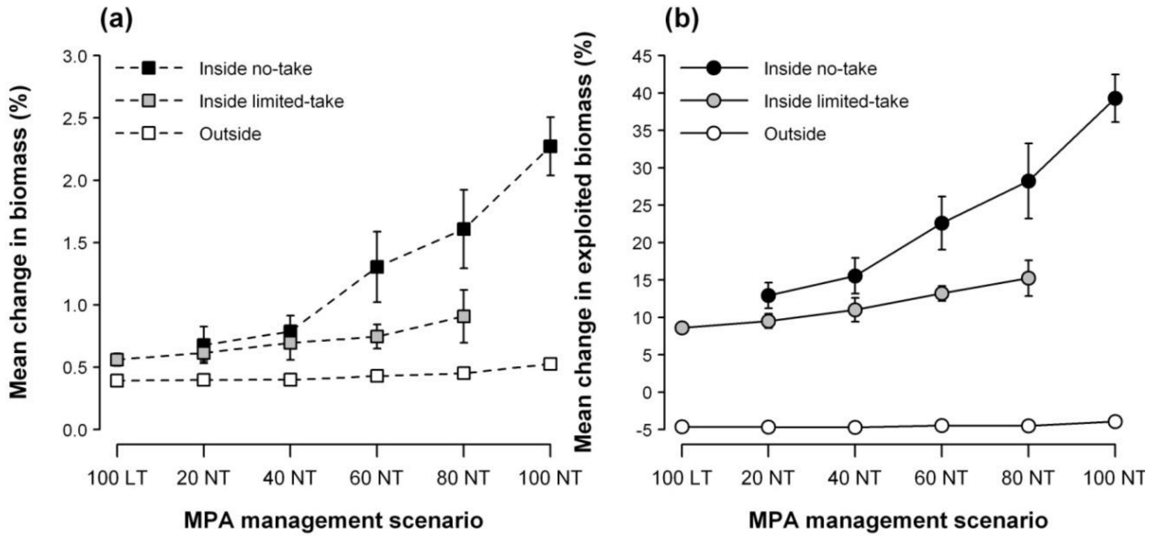

Figure 5.7 Mean percentage change (± 95% confidence intervals) in the density of: (a) overall and (b) exploited ecosystem biomass (t/km^2) inside and outside of MPAs for each management scenario.

Each of the proposed MPA management scenarios also had a positive impact on fisheries in the EEC, resulting in an increase in overall and exploited catches; this was particularly evident for MPA networks that were comprised of a larger proportion of no-take zones (**Figure 5.4c**; **Figure 5.4d**). For example, after 50 years a 100% limited-take MPA network increased the mean overall and exploited catch by 0.27% and 2.07% relative to the baseline scenario with no MPAs. In contrast, a 100% no-take MPA network increased the mean overall and exploited catch by 34.5% and 39.5%. However, there was an initial decrease in catches following MPA establishment (**Figure 5.4c**; **Figure 5.4d**). Whilst this decrease was greater for MPA networks that had a larger proportion of no-take zones, catches took longer to return to baseline levels for MPA networks that were comprised of a large proportion of limited-take zones (**Figure 5.4c**; **Figure 5.4d**). For example, overall and exploited catches took 37.4 and 18.9 years respectively to return to baseline levels for a 100% limited-take MPA network. In contrast, overall and exploited catches took 3.7 and 3.4 years to return to baseline levels for a 100% no-take MPA network. In addition, not all fisheries fleets benefitted from MPA establishment (**Figure 5.8**), this was particularly evident for fleets using active gears (beam trawl, dredges and demersal otter) which generally had a small decrease in catches across all MPA management scenarios.

Finally, the MTI indicated that each of the proposed MPA management scenarios had an influence on the mean TL of catches in the EEC (**Figure 5.4e**; **Figure 5.4f**; **Figure 5.9**). Under baseline simulations with no MPAs the mean TL of catches after 50 years was 3.22 (**Figure 5.4e**), though this increased to 3.48 when calculated to focus on the exploited part of the system (**Figure 5.4f**). However, whilst the results indicate that the mean TL of catches increased following a period of initial decline (**Figure 5.4e**), the mean TL of exploited catches decreased below baseline levels for MPA networks that were comprised of a larger proportion of limited-take zones (**Figure 5.4f**). For example, after 50 years the mean TL of exploited catches for a 100% limited-take MPA network was 3.47, in contrast, the mean TL of exploited catches for a 100% no-take MPA network was 3.51.

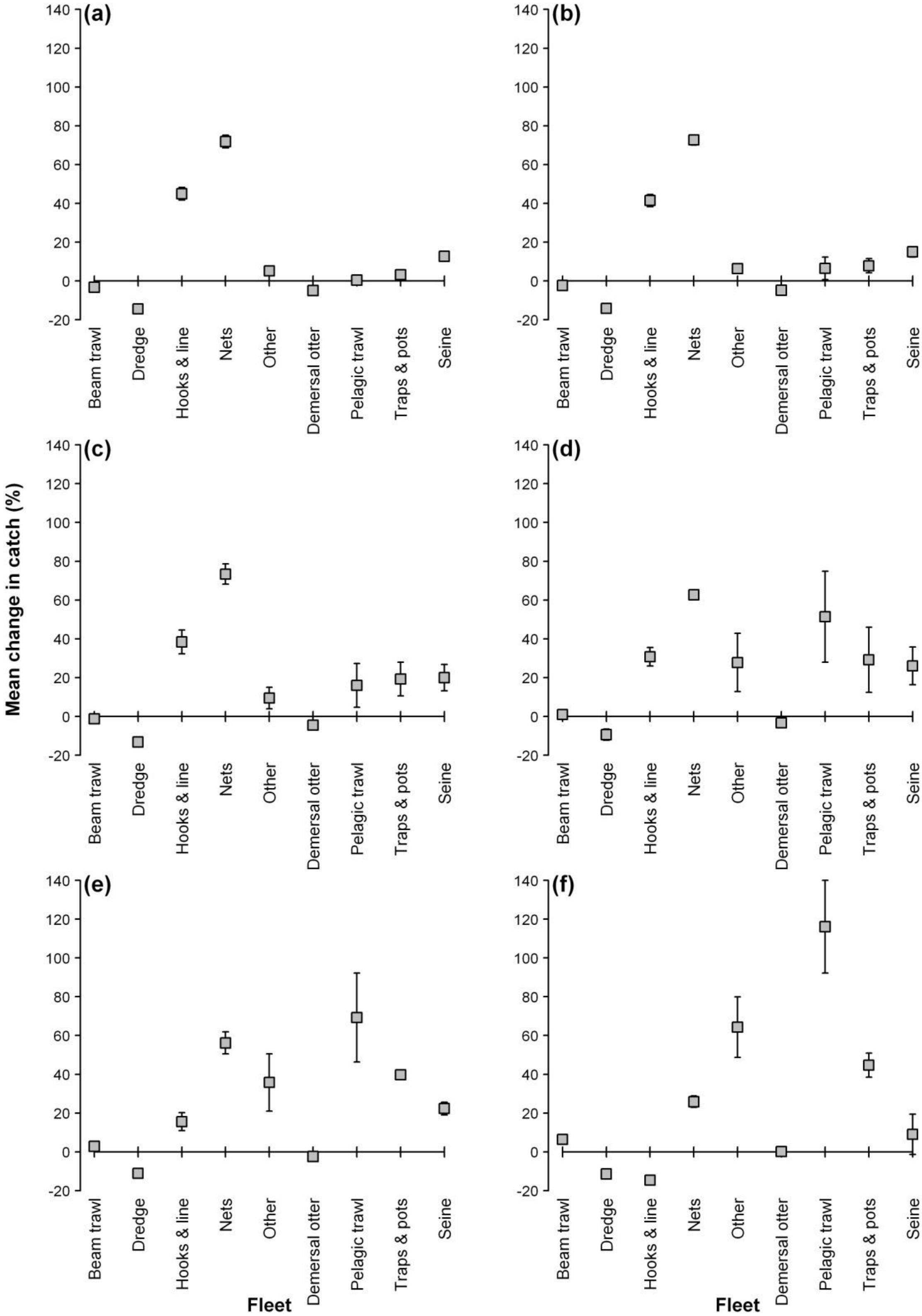

Figure 5.8 Mean percentage change (± 95% confidence intervals) in catch (t/km^2) per fisheries fleet for each MPA management scenario: (a) 100% limited-take; (b) 20% no-take; (c) 40% no-take; (d) 60% no-take; (e) 80% no-take; and (f) 100% no-take, relative to baseline scenario with no MPAs.

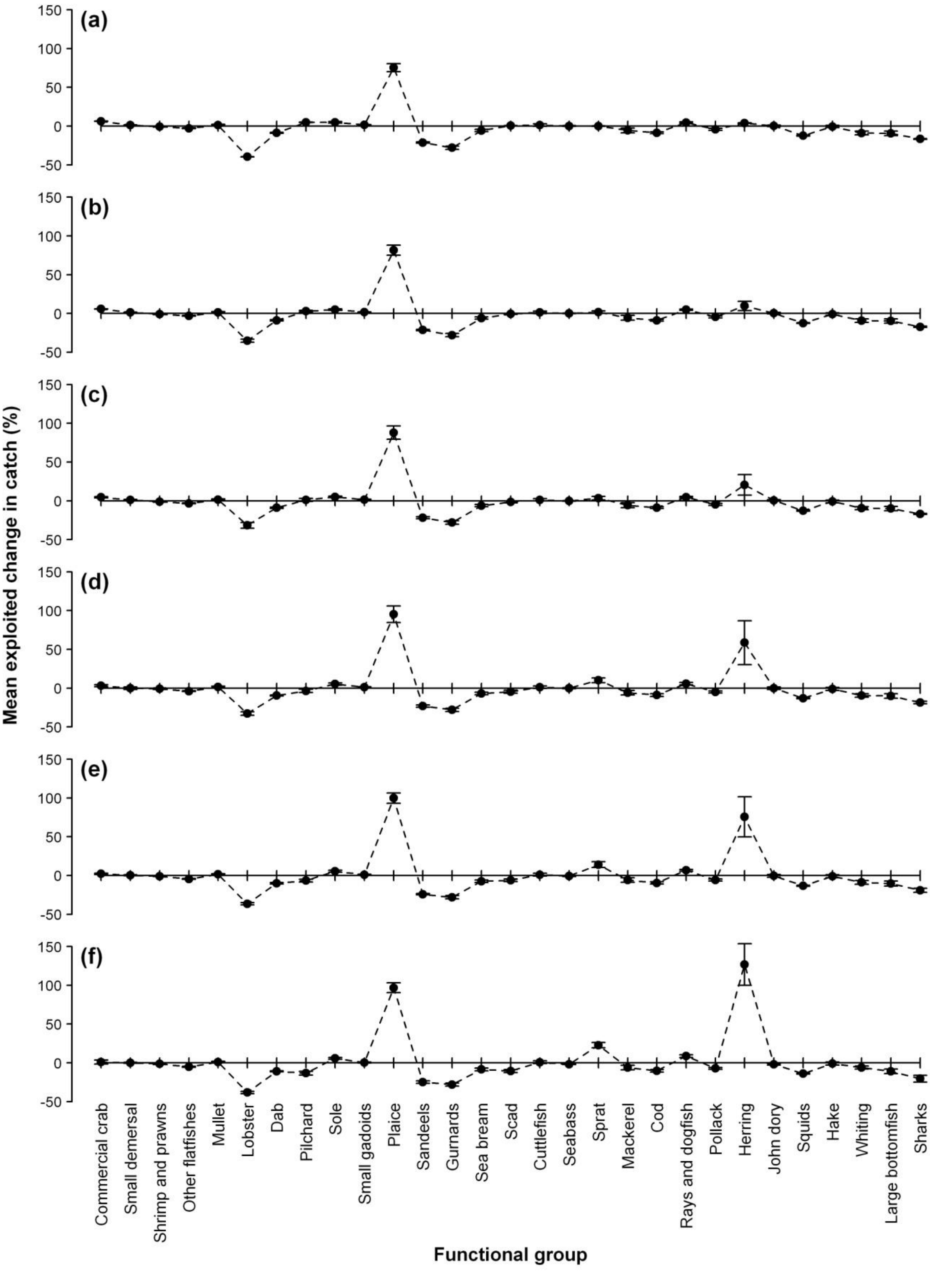

Figure 5.9 Mean percentage change (± 95% confidence intervals) in exploited catch (t/km^2) per functional group (TL ≥ 2.35) for each MPA management scenario over time (0 – 50 years): (a) 100% limited-take; (b) 20% no-take; (c) 40% no-take; (d) 60% no-take; (e) 80% no-take; and (f) 100% no-take.

5.5 DISCUSSION

MPAs developed to achieve conservation objectives are often seen as controversial, particularly from a fisheries perspective (Abbott & Haynie 2012). This is largely due to the fact that closures, especially no-take MPAs and/or gear restrictions involved in MPA management are clearly a tangible threat to the size of fishing areas (Rassweiler *et al.,* 2012). Consequently, there is increasing demand from stakeholders and policy makers for information that could be used to evaluate the potential long-term impact of different MPA management strategies (Agardy *et al.,* 2011; Claudet *et al.,* 2011; Vandeperre *et al.,* 2011). This is likely to be seen as increasingly important given that many countries have initiated ambitious plans to establish new or expand existing MPA networks by 2020, in response to global marine protection targets (Wood 2011).

The majority of these countries, however, are unlikely to be able to implement long-term research programs to determine how these networks could be managed prior to implementation (Claudet *et al.,* 2011). This paper therefore represents the first such attempt to show how two of the most widely used software tools in marine conservation planning ‘Marxan’ and ‘Ecopath with Ecosim’ can be combined to help evaluate different forms of spatial management, and thus inform stakeholders and policy makers of the potential impacts and trade-offs of different strategies. Therefore, in this section we discuss: (i) the impact of spatial marine zoning for conservation and sustainable fisheries management relative to no-take MPA networks; (ii) the broader implications of these results for global marine conservation planning efforts; and (iii) the potential limitations with the model based on current empirical evidence.

5.5.1 Spatial marine zoning versus no-take MPA networks

In the last decade considerable effort has been focused on establishing no-take MPA networks as research has indicated they produce substantial benefits in terms of increases in ecosystem biomass, and the abundance and density of both target and non-target species inside and outside of MPAs (Roberts *et al.,* 2001; Halpern & Warner 2002; Gell & Roberts 2003; McCook *et al.,* 2010). In contrast, the potential for broader classes of spatial management that include spatial zoning of fleet access and gear restrictions have received far less attention in MPA design (Rassweiler *et al.*, 2012), with the exception of the Great Barrier Reef Marine Park (McCook *et al.*, 2010). Moreover, very little research exists on the potential benefits of partially protected areas relative to no-take MPAs, particularly in temperate waters (see Lester & Halpern 2008; McCook *et al.*, 2010).

Therefore, whilst this study indicated that strict no-take MPA networks produced greater increases in overall and exploited ecosystem biomass, and yielded higher densities of catches; it also demonstrated that MPA networks comprised of partially protected areas have the potential to confer some benefits relative to a system with no MPAs. In particular, MPA networks with limited-take zones that excluded active gears (e.g. bottom trawls and dredges) produced an increase in overall and exploited ecosystem biomass, and catches of target species. However, increases in catches were smaller and often took longer to return to baseline levels for MPA networks that were comprised of a larger proportion of limited-take zones. For example, we show that exploited catches recover 6 times as quickly in 100% no-take MPA networks when compared to 100% limited-take MPA networks (**Figure 5.4d**).

In addition, these results imply that MPA networks comprised of partially protected areas also have the same potential as no-take MPA networks to produce increases in ecosystem biomass both inside and outside of MPAs. This suggests that MPAs that include limited-take zones are also able to enhance surrounding areas through processes such as spill-over and export (Gell & Roberts 2003). However, in the context of multispecies environments there are winners and losers associated with the different management strategies, both in terms of the abundance and biomass of target and non-target species, and for the different fisheries fleets (**Figure 5.5**; **Figure 5.8**). For example, changes in biomass for functional groups were largely the result of differences in dispersal distances and the level of protection offered by the different MPA management strategies. Consequently, species with lower levels of dispersal generally benefitted more from MPA networks that were comprised of a larger proportion of no-take zones, which is consistent with several studies (McCook *et al.*, 2010; Varkey *et al.*, 2012). However, the impacts on different fisheries fleets were largely limited to those using active gears (beam trawl, dredges and demersal otter), which experienced a decline in catches as a result of being restricted from operating in both no-take and limited-take zones across all MPA management scenarios.

Despite the socio-economic impacts associated with excluding all fishing activities and the limited evidence that suggests MPA networks comprised of partially protected areas do produce some benefits (e.g. Murawski *et al.,* 2000; Blyth-Skyrme *et al.,* 2006; Floeter *et al.,* 2006), no-take MPA networks are frequently established for political reasons, such as less complicated regulations and easier enforcement (Bohnsack *et al.,* 2004; Lester & Halpern 2008). However, recent research has indicated that spatial marine zoning could result in a reduced short-term socio-economic impact on fisheries, a more equitable impact on different fishing sectors, and a considerable increase in profits (Klein *et al.,* 2010; Rassweiler *et al.,* 2012), and our study showed similar trends. For example, setting higher targets for the proportion of the MPA network that was allocated to the limited-take zone reduced the impact of the proposed network on fisheries. This suggests that MPA networks that include spatial zoning of fleet access and gear restrictions could therefore represent a more politically feasible and less contentious management strategy, when compared to strict no-take MPA networks that effectively exclude all access. For example, in cases where zoning could result in long-term increases in fisheries yields whilst contributing towards conservation goals (as demonstrated herein), this information could help reduce social resistance to new regulations and increase stakeholder support (Rassweiler *et al.,* 2012) .

Nevertheless, this study indicated that the mean TL of catches was not quite as resistant to changes in management. For example, when calculating the MTI to focus on the exploited part of the system, greater zoning of fleet access and gear restrictions led to a reduction in the mean TL of catches relative to a system with no MPAs. Whilst this was not the case for all management scenarios, it does imply that MPA networks comprised of a larger proportion of limited-take zones (> 60% of proposed MPA network) could potentially increase fishing pressure on species with a higher TL, thus leading to a decline in their relative abundance and a shift in the targeting behaviour of fisheries to more abundant lower TL species (**Figure 5.9**). However, no specific MTI targets exist to help determine the most appropriate type of management intervention (Pauly & Watson 2005). Nonetheless, these results suggest that: (i) MPA networks comprised of a greater proportion of zones that have less restrictive regulations may not represent a preferable management strategy; and, (ii) MPA networks in the EEC should be comprised of at least 60% no-take zones to ensure that fisheries catches remain sustainable. This finding is particularly important from a European perspective as Member States are required by 2020 to: (i) establish networks of effectively managed MPAs that reduce direct pressures on marine biodiversity and contribute to the sustainable use of marine resources (CBD 2010b); and (ii) take necessary measures to achieve 'good environmental status' of European seas, with a particular emphasis on sustainable use (EC 2008b; Fenberg *et al.,* 2012).

5.5.2 Marine conservation planning and spatial marine zoning

Designing MPA networks that consider both socio-economic and biodiversity factors has moved to the forefront of conservation planning (Stewart *et al.*, 2003; Ban & Klein 2009; Ban *et al.,* 2011). A key aspect of this involves working with the relevant stakeholders and implementing agencies to develop a better understanding of the opportunities and constraints associated with each type of conservation intervention (Knight *et al.,* 2006b). Spatial marine zoning in particular has been shown to provide a means to spatially separate incompatible human activities and reduce conflict among user groups (Crowder *et al.,* 2006). However, in any planning process there are still likely to be trade-offs between achieving conservation objectives and minimising socio-economic impacts on stakeholders (Klein *et al.,* 2008b). Thus, if these trade-offs are not transparent in the planning process then we may not adequately conserve marine biodiversity, or foster stakeholder support (Klein *et al.,* 2010). Faced with these issues it is important that we assess the value of different MPA management strategies, especially if we want to understand whether the benefits of no-take MPAs justify the cost of their implementation, or whether there are others forms of spatial management that could achieve comparable results (Rassweiler *et al.,* 2012). Furthermore, a major objective of MPA network design is ensuring that they contribute towards sustainable fisheries, however representing this within a systematic conservation planning framework is often seen as difficult (Grantham *et al.*, 2013).

Here we demonstrate a policy screening method that can provide an insight into the potential trade-offs associated with adopting different spatially explicit management strategies. We show that broader classes of MPA management that include spatial zoning of fleet access and gear restrictions can also provide simultaneous conservation and fisheries benefits, which is important given that this approach is likely to be less politically contentious than strict no-take MPA networks. However, we also show that no-take zones must form an integral part of proposed MPA networks, as they are necessary to ensure the long-term protection of some conservation features and support sustainable fisheries. Therefore, given that the success of MPAs in meeting conservation objectives depends on user compliance (Sumaila *et al.,* 2000), this approach could be used in conjunction with existing spatial prioritisation approaches to help minimise conflict and reduce social resistance. This would involve using model outputs to inform stakeholders of the potential long-term ecological and fisheries responses to different types of management regulations. However, all conservation plans require further stakeholder consultation to establish: (i) the different types of protected area zones and management restrictions that fit within existing policy frameworks; and, (ii) how much these different types of zones would cost to enforce, as recent research has shown that management of no-take MPAs is more cost efficient than multiple use zones (Ban *et al.,* 2011). Moreover, increases in ecosystem biomass, and higher densities of catches for fisheries can only be achieved if the fishery is well understood and the regulations are strategically designed to foster stakeholder support and ensure compliance (McCook *et al.*, 2010; Rassweiler *et al.*, 2012).

5.5.3 Limitations and caveats

Determining the functional roles of species and modelling their interactions is particularly relevant in conservation planning, especially to understand the consequences of different management strategies on ecosystem functioning. However, an ecosystem model will only be as good as the data that are used to create it (Stanford & Pitcher 2004). In particular, there is often high a level of uncertainty surrounding the baseline model inputs that are used in Ecopath with Ecosim to model future simulations (Araujo *et al.,* 2008), especially when accounting for the complexity of the dynamics of, and interactions between, the various components of an ecosystem (Plaganyi & Butterworth 2004). Therefore, it is worth acknowledging that due to the complexity of ecosystem processes, and our limited understanding of these in the EEC (Villanueva *et al.,* 2009; Daskalov *et al.,* 2011), the results herein should be considered as conservative estimates of the changes that are likely to occur under the proposed MPA management scenarios. This is because empirical evidence suggests both total and exploited ecosystem biomass are likely to increase much more substantially within no-take and limited-take MPAs than the changes predicted by the Ecospace model (e.g. Murawski *et al.,* 2000; Halpern 2003; Halpern & Lester 2008). For example, in a study of 89 MPAs Halpern (2003) showed that on average no-take MPAs led to a 3 fold increase in biomass inside of MPA boundaries relative to unprotected areas, which is ~2 orders of magnitude greater than the findings of this study.

The results of this study are however, consistent with empirical studies that have shown that some species are likely to benefit more from protection than others. For example, areas closed to dredges in Georges Bank in Southern New England, totalling 17,000 km^2 led to a 9 fold increase in scallop biomass inside MPA boundaries (Murawski *et al.,* 2000). Here we show that for MPA networks comprised of 100% no-take MPAs (totalling on average 12,460 km^2) there was on average a 3, 3, 6 and 26 fold increase in biomass inside MPA boundaries for commercially valuable, and target species such as seabass (*Dicentrarchus labrax*), sole (*Solea solea*), oysters (*Ostrea edulis*) and plaice (*Pleuronectes platessa)* respectively. In addition, we show that for MPA networks comprised of 100% limited-take MPAs there was also on average a 1.5, 2 and 11 fold increase in biomass inside MPA boundaries for seabass, sole, and plaice respectively. This highlights that the exclusion of all gears is likely to result in substantially greater increases in biomass of target species compared to just excluding active gears from MPAs.

Nonetheless, given the dynamic nature of marine ecosystems, the results presented herein should be considered in qualitative rather than quantitative terms as an aid to discussions on determining the appropriate type of management (Araujo *et al.*, 2008). Moreover, where there is uncertainty this approach should be used to investigate particular management scenarios in more detail, accounting for complexities in trophic dynamics, and variability in fishing effort to ensure that predictions are based on a range of likely scenarios (Plaganyi & Butterworth 2004). However, it is worth acknowledging that ecosystem and multispecies models are able to predict or at least provide warnings against otherwise unknown, undesirable or counterintuitive responses to changes in management that are often proposed as part of conservation planning assessments (Araujo *et al.*, 2008); and so have the potential to make an important contribution to conservation and fisheries management.

5.6 ACKNOWLEDGEMENTS

We would like to acknowledge the following organisations and sources for providing access to a number of datasets used in this study: Ifremer, Cefas, JNCC, the European Register of Marine Species and the Crown Estate. We are also grateful to Bruno Nhancale for support with Marxan with Zones and Zonae Cogito.

CHAPTER 6

GENERAL DISCUSSION

6.1 BACKGROUND

Marine ecosystems are under increasing pressure from a diverse range of threats. Many national governments have responded to these threats by establishing marine protected area (MPA) networks (Lubchenco *et al.,* 2003; Wood *et al.,* 2008). One such approach for designing MPA networks is systematic conservation planning (Margules & Pressey 2000), which is now considered the most effective approach for identifying priority areas (Kareiva & Marvier 2012). However, the main exception to this trend is Europe, where the designation of MPAs is still largely based on expert opinion. Therefore, this thesis aims to show how systematic conservation planning can be used to support the identification of MPAs that fit within existing European marine policy frameworks, and so contributes to best practice in a number of areas.

6.2 CONTRIBUTIONS TO THE LITERATURE

6.2.1 Marine conservation policy in Europe

Marine conservation policy in Europe clearly has widespread political support as demonstrated by the rapid expansion of protected areas across Member States marine jurisdictions. However, research has shown that existing approaches used to guide the selection of these sites are not the most effective (Jackson *et al.,* 2004), and so there have been increasing calls to identify factors influencing the success of MPAs in Europe (Fenberg *et al.,* 2012). In order to identify these factors there needs to be a greater understanding of the legislation used to guide MPA design in Europe. In **Chapter 2** I reviewed how existing approaches used to guide selection of MPAs compared to best practice developed in systematic conservation planning, as this approach is widely considered an important framework to identify priority areas for conservation. This is the first study to show that marine conservation policy in Europe is failing to benefit from lessons learnt in systematic conservation and integrate key findings from scientific research into policy. It also highlights that existing legislation lacks key components of best practice which are designed to efficiently identify priority areas that minimise impacts on stakeholders, and increase the likelihood of implementation. In particular, existing legislation fails to: (i) translate broad policy goals into explicit quantitative targets of how much of each feature should be conserved; (ii) incorporate socio-economic data into the planning process; and (iii) include a social assessment to help translate priority areas and goals into conservation action. Thus, these findings highlight that there is a real need to adapt existing legislation to allow a more flexible approach to MPA design that accounts for local conditions and supports implementation.

6.2.2 Marine protected area network design

MPA network design and conservation planning has evolved significantly over the last two decades (Margules & Pressey 2000; Margules & Sarkar 2007; Gaines *et al.*, 2010a), and this has resulted in several ecological principles that should underpin the identification of priority areas, including: representation, replication, adequacy, viability, connectivity, and protection (e.g. Airame *et al.*, 2003; Roberts *et al.*, 2003a; Roberts *et al.*, 2003b; IUCN-WCPA 2008). In order to meet these principles most MPA projects have developed guidelines using best available evidence (e.g. Day & Roff 2000; CDFG 2008; JNCC & Natural England 2010). However, whilst these documents provide an important foundation for the development of MPA networks, conservation practitioners often apply these guidelines without considering how they could influence MPA design. Therefore, in **Chapter 3** and **Chapter 4** I focused on two approaches that are increasingly being advocated in MPA network design.

The first study (**Chapter 3**) focused on habitat targets, which are increasingly used in systematic conservation planning to determine the minimum amount of each habitat type that should be represented within a protected area network (Carwardine *et al.*, 2009). This is based on the assumption that by conserving a proportion of each habitat that most elements of biodiversity such as species, communities and physical characteristics will be adequately represented. However, up until recently targets for habitat types were frequently based on expert opinion or policy goals, which have been widely criticised for being ecologically irrelevant (Svancara *et al.*, 2005; Tear *et al.*, 2005). In response to these criticisms, researchers developed an approach based on the species-area relationship (SAR) to estimate the proportion of habitat area required to represent a given percentage of species (Desmet & Cowling 2004; Rouget *et al.*, 2004). Consequently, this approach is increasingly being used to set targets for habitat types when designing protected area networks, because it can: (i) provide a transparent an objective method for converting judgements of minimum species representation into quantitative targets; and, (ii) distinguish between different habitat types and tailors targets to account for differences in species richness and turnover. However, the SAR-based approach is used without question, despite concerns about its appropriateness for use in conservation planning (Smith 2010).

This is the first study to look at this approach in detail and address concerns raised in the literature. In particular, I show that SAR-based targets are sensitive to changes in: (i) the number of samples used to generate estimates of species richness for each habitat type; (ii) the type of estimator used to calculate richness estimates; and, (iii) the level of habitat classification employed to develop targets. However, whilst each of these tested factors had an influence on targets, this study showed that the number of samples had the greatest impact on SAR-based targets, with specific habitat targets increasing by up to 45% when sample size increased from 50 to 300. In addition, this research highlighted that targets were higher when using broader habitat classification levels or a larger study region. Thus, it is possible that national-level habitat targets could over-represent the proportion of habitat area required to represent a given percentage of species when applied at smaller regional scales.

The second study (**Chapter 4)** focused on MPA size guidelines, which are frequently based on species movement and dispersal characteristics to help determine the minimum size of MPAs in a network that will support viable populations (Moffitt *et al.,* 2010). However, whilst these guidelines are increasingly used to inform MPA network design no studies have considered how we could incorporate MPA size constraints into a systematic conservation planning framework. Moreover, given that MPA size guidelines can often range in scale from 10s to 100s of km^2 there has been no research into the potential trade-offs associated with establishing MPA networks comprised of different size MPAs. This study is therefore the first to demonstrate how we can incorporate MPA size constraints into a systematic conservation planning framework to provide a transparent platform to investigate the impact of MPA size. In particular, I show that MPA size has a significant impact on: (i) the spatial characteristics of MPA networks; (ii) stakeholders, in terms of opportunity costs; and, (iii) connectivity for species. For example, increasing MPA size by a factor of 10 in inshore and offshore waters decreased the median number of MPAs in a network by 21%, but increased the median area of MPA network portfolios by 0.44% and cost by 10.15%. Moreover, I show that increasing the minimum size of MPAs reduces connectivity for species with smaller dispersal distances because it reduces the number of MPAs in a network and so increases the distance to neighbouring MPAs.

6.2.3 Spatial prioritisation and decision support tools

In Europe most priority areas are identified on a site by site basis, based on the properties of individual sites (Ioja *et al.*, 2010), such as the presence of species and habitats of conservation concern (e.g. Stroud *et al.*, 2001; McLeod *et al.*, 2005). In contrast, systematic conservation planning involves identifying cost efficient sets of areas that meet explicit quantitative targets, and uses complementarity based methods for selecting sites, so protected areas are selected based on how much they would add to the existing network, rather than how much of each feature they contain (Margules & Pressey 2000). To help achieve these objectives systematic conservation planning is supported by a range of spatial prioritisation software (e.g. Marxan, C-Plan, Zonation; Moilanen *et al.*, 2009), which are increasingly being used to support protected area network design in Australia, North America and South Africa (e.g. Cowling *et al.*, 2003a; Pressey *et al.*, 2003; Fernandes *et al.*, 2005; Rouget *et al.*, 2006; Fernandes *et al.*, 2009). However, these tools are not widely used to support or inform MPA network design in Europe. The few studies that have used these tools have: (i) compared how different software (e.g. Marxan and Zonation) influence the location of priority areas (Delavenne *et al.*, 2012); and, (ii) been based on identifying priority areas that meet arbitrary targets (e.g. Maiorano *et al.*, 2009, Delavenne *et al.*, 2012, Giakoumi *et al.*, 2012), and so are of limited use to policy makers because they do not represent national priorities and regional goals. Therefore, this work is the first to show how spatial prioritisation software (i.e. Marxan, Marxan with Zones and Ecopath with Ecosim) can be used to support the identification of priority areas that fit within existing European policy frameworks. Moreover, I also demonstrate how these tools provide a transparent approach for investigating the potential trade-offs associated with: (i) identifying MPA networks that meet a range of design criteria (**Chapter 4** and **Chapter 5**); and (ii) determining the most appropriate type of management strategy for MPA networks, prior to implementation (**Chapter 5**).

6.2.4 Marine protected area management

MPA networks are increasingly being established to meet a broad array of conservation and resource management objectives, and can include zones with varying levels of protection; ranging from no-take areas that restrict all extractive activities to partially protected areas that allow selective extraction of resources (Jentoft *et al.,* 2011; Wood 2011). Given how complex it is to decide on the location of these zones, spatial prioritisation software is often used to help identify different zoning configurations that minimise impacts on stakeholders (Watts *et al.,* 2009). However, whilst these tools are designed to reflect the range of MPA management options being considered as part of a conservation plan (e.g. Klein *et al.,* 2010; Malcolm *et al.,* 2012; Grantham *et al.,* 2013), they fail to address the potential long-term impacts associated with establishing MPA networks comprised of different zones. In addition, there is often very little data to inform future management decisions about whether strict no-take MPA networks justify the cost of their implementation, or whether MPA networks comprised of multiple zones with different management restrictions could achieve similar results. Thus, there is often a great deal of uncertainty about how to determine the most appropriate type of management for MPA networks, and how this might impact biodiversity, environmental processes and stakeholders.

In **Chapter 5** I provide an approach that can be used to help explore the potential long-term impacts associated with adopting different spatially explicit MPA management strategies, using the eastern English Channel as a case study to illustrate. This is the first such study to combine outputs from spatial prioritisation software with an ecosystem model to evaluate how different MPA management strategies could influence marine ecosystem functioning and fisheries. In addition, it is also the first study to investigate how different MPA designs can contribute to sustainable fisheries within a systematic conservation planning framework, something that was previously considered difficult (Grantham *et al.*, 2013).

In particular, I show that broader classes of spatial management based on zoning fleet access and gear restrictions can have conservation and fishery benefits. For example, after 50 years a 100% limited-take MPA network increased exploited ecosystem biomass by 3.41% and fisheries catches by 2.07%, relative to a system with no MPAs. This is an important finding as there is often very little support for MPAs (Smith *et al.*, 2010a; Abbott & Haynie 2012; Rassweiler *et al.*, 2012), and so this type of approach could minimise conflict with stakeholders, as it is less politically contentious than strict no-take MPA networks. However, I also show that if MPA networks are to ensure the sustainable use of fisheries they should be comprised of at least 40% no-take zones and that a 100% no-take MPA network would increase exploited ecosystem biomass by 14.01% and fisheries catches by 39.5%, relative to a system with no MPAs. Moreover, I also show that exploited catches recovered 6 times as quickly in 100% no-take MPA networks when compared to 100% limited-take MPA networks.

Given that the success of MPAs, in meeting conservation objectives depends on ensuring stakeholder support and user compliance (Sumaila *et al.*, 2000), I believe this type of approach could become increasingly valuable as a policy screening method to help tailor management plans to addresses both conservation and management objectives, and the needs of stakeholders. This is because Ecopath with Ecosim provides an opportunity to forecast the potential impacts associated with future management interventions, and understand their consequences relative to the long-term objectives of the MPA network. More importantly, this type of approach can also be used to help determine when and how rapidly changes occur so that plans can be revised if they fail to achieve their long-term goals, something that is currently lacking with existing approaches. However, it is worth noting that ecosystem models require a substantial amount of time to develop and test, but this should not deter practitioners. For example, several models exist for UK and European waters (e.g. Araujo *et al.,* 2006; Lees & Mackinson 2007; Mackinson & Daskalov 2007; Araujo *et al.,* 2008; Villanueva *et al.,* 2009; Daskalov *et al.,* 2011) that could be used to help determine the most appropriate type of MPA management strategy in different regions where human pressures and resource management objectives may differ.

6.3 LIMITATIONS AND AREAS OF FURTHER RESEARCH

6.3.1 Conservation assessment data

Systematic conservation planning assessments are restricted to the best available data to identify priority areas for conservation action (Margules & Pressey 2000; Margules *et al.,* 2002; Sarkar *et al.,* 2006; Margules & Sarkar 2007). Consequently, this thesis highlighted a number of data gaps that currently exist for UK and European waters that could improve future spatial prioritisation analyses. For example, at present more fine-scale data exists for marine habitats, which is largely the result of several national and European habitat mapping projects (e.g. Coltman *et al.,* 2008; MESH 2008; Coggan & Diesing 2011; McBreen *et al.,* 2011). However, species data are often restricted to point distribution records and habitat suitability maps for species of conservation concern and commercial value respectively. Moreover, these species distribution records are often based on data collected prior to 1980, and therefore are unlikely to reflect their current distribution, which could lead to misdirected site selection.

Thus, if Member States are committed to establishing an ecologically coherent network of MPAs further research an emphasis should be placed on developing more accurate models that: (i) represent the current distribution of species of conservation concern, particularly those listed by OSPAR; and, (ii) identify areas that support key ecological processes that are difficult to define spatially, such as migratory routes, spawning and nursery areas, which were excluded from **Chapter 4** and **Chapter 5** due to their coarse resolution. This is important as a robust evidence base is critical to support and inform future decisions (Rees *et al.,* 2013), especially as many countries have adopted measures to support renewable energy sources and are therefore increasing the proportion of their offshore waters earmarked for development (Inger *et al.,* 2009).

In addition, the vessel monitoring system (VMS) data used to develop planning unit costs for the conservation assessments undertaken in **Chapter 4** and **Chapter 5** were designed to help identify areas that minimise impacts on fisheries (Lee *et al.,* 2010). Thus, planning units with high levels of fishing effort would be more costly for Marxan to select relative to planning units with lower levels of fishing effort, and so are less likely to be selected unless they are required to meet targets. However, VMS data does not provide accurate information on the spatial distribution of fishing effort and activity for vessels that are < 15 m in length (Lee *et al.,* 2010). Therefore, caution is needed when interpreting these results as they might identify priority areas that are more likely to be in areas fished by these smaller vessels, particularly in UK waters where a large proportion of inshore fisheries is by vessels < 15 m (Lee *et al.,* 2010). However, VMS data still provides a reliable and transparent method for identifying areas of high fishing effort and activity in the absence of better alternatives (Witt & Godley 2007). Therefore, further research is required to look at how fine-scale data for smaller inshore fisheries could be incorporated into future Marxan analyses, or how planning unit costs could be manipulated to increase costs in inshore waters that are important for vessels < 15 m in length.

6.3.2 Ecosystem models and marine protected area network design

Ecosystem models such as Ecopath with Ecosim (Pauly *et al.,* 2000) are clearly a powerful tool to help quantify the impacts of different types of MPAs that fit within existing policy frameworks. However, the study presented in **Chapter 5** is based on the assumption that MPAs would be designated at the same time, and therefore fails to consider that each country may schedule the implementation of priority areas in a network according to their own national priorities. For example, the UK government is proposing that the 127 recommended MCZs are to be established in a number of phases, with the first tranche of thirty-one sites due to be designated in 2013 (DEFRA 2012). However no decisions have been made on when or how many of the remaining sites will be established in the subsequent phases.

In addition, there has been no research to establish what impact this type of implementation strategy could have on marine ecosystem functioning and productivity, and whether it could still achieve similar results to designating all priority areas at the same time. Therefore, ecosystem models such as those presented in **Chapter 5** could be used to support future policy decisions on: (i) the number of MPAs that should be established at each stage; (ii) the maximum amount of time required between establishing different types of MPAs; and (ii) whether larger MPAs should be designated in the first stages. These types of questions are likely to become increasingly important as many national governments are establishing MPA networks to meet global marine protection targets by 2020, and so are likely to schedule the designation of priority areas to minimise conflict with stakeholders.

In addition, whilst **Chapter 4** provides a general insight into the potential impact of MPA size constraints, this research does not consider that the optimal size of MPA is also likely to be dependent on the level of protection afforded to MPAs. Therefore, ecosystem models such as those presented in **Chapter 5** could be used to investigate: (i) the long-term persistence of features targeted for protection in MPAs of varying size; (ii) what impact MPA size has on the persistence of features targeted for protection under future MPA management scenarios; and, (iii) the smallest size different types of MPAs (e.g.no-take and partially protected areas) have to be to ensure long-term population persistence.

6.3.3 Conservation planning and management outside of MPA boundaries

In addition, research has shown that even the most well designed and managed MPAs cannot protect habitats, species and ecosystems from the activities outside of their boundaries (Agardy *et al.*, 2011). Therefore, whilst the establishment of MPAs will remain an important part of conservation, planning beyond the boundaries of MPAs is likely to become a greater focus of marine conservation planning efforts, as many scientists argue that MPAs should be established in conjunction with other management tools (Agardy 1994; Allison *et al.*, 1998). However, what and how MPAs are managed beyond their boundaries is rarely acknowledged or even considered in

MPA network design, something that is also not explicitly considered in **Chapter 5**. Therefore, future planning exercises could benefit from using ecosystem models to investigate the impacts of combining different types of MPA management strategies with interventions outside protected area boundaries, such as a reduction in effort, gear restrictions, discard bans, and quotas.

6.4 CONCLUSIONS

This thesis highlights that if European approaches to marine conservation planning are to fulfil their original objectives of halting the loss of marine biodiversity, and meet new goals for the sustainable use and protection of the marine environment, existing legislation needs to be adapted to allow Member States to adopt more flexible and transparent approaches. In particular, there needs to be a greater focus on: (i) incorporating key components of best practice from systematic conservation planning that are designed to efficiently identify priority areas that minimise impacts on stakeholders, and increase the likelihood of implementation; and, (ii) combining existing prioritisation approaches with decision support tools, which can provide a transparent and scientifically defensible platform to explore different MPA design and management strategies. However, this can only be achieved with stronger collaboration between scientists, stakeholders and policy makers. Nonetheless, this thesis provides an important first step towards demonstrating the value of adopting a systematic conservation planning approach to MPA network design in Europe. Moreover, it shows this type of approach can play an important role as a policy screening method to ensure that that we move away from MPA network design that is dictated by policy, to science based MPA design that accounts for local conditions and supports implementation.

CHAPTER 7

REFERENCES

– A –

Abbott, J.K. & Haynie, A.C. (2012) What are we protecting? Fisher behaviour and the unintended consequences of spatial closures as a fishery management tool. *Ecological Applications,* 23(3), 762-777.

Agardy, M.T. (1993) Accommodating ecotourism in mutliple-use planning of coastal and marine protected areas. *Ocean & Coastal Management,* 20(3), 219-239.

Agardy, M.T. (1994) Advances in marine conservation - the role of marine protected areas. *Trends in Ecology & Evolution,* 9(7), 267-270.

Agardy, T., Bridgewater, P., Crosby, M.P., Day, J.C., Dayton, P.K., Kenchington, R., Laffoley, D.d.A., McConney, P., Murray, P.A., Parks, J.E. & Peau, L. (2003) Dangerous targets? Unresolved issues and ideological clashes around marine protected areas. *Aquatic Conservation-Marine and Freshwater Ecosystems,* 13(4), 353-367.

Agardy, T., di Sciara, G.N. & Christie, P. (2011) Mind the gap: Addressing the shortcomings of marine protected areas through large scale marine spatial planning. *Marine Policy,* 35(2), 226-232.

Airame, S., Dugan, J.E., Lafferty, K.D., Leslie, H., McArdle, D.A. & Warner, R.R. (2003) Applying ecological criteria to marine reserve design: A case study from the California Channel Islands. *Ecological Applications,* 13(1), S170-S184.

Allison, G.W., Lubchenco, J. & Carr, M.H. (1998) Marine reserves are necessary but not sufficient for marine conservation. *Ecological Applications,* 8(1), S79-S92.

Almany, G.R., Connolly, S.R., Heath, D.D., Hogan, J.D., Jones, G.P., McCook, L.J., Mills, M., Pressey, R.L. & Williamson, D.H. (2009) Connectivity, biodiversity conservation and the design of marine reserve networks for coral reefs. *Coral Reefs,* 28(2), 339-351.

Apostolopoulou, E. & Pantis, J.D. (2009) Conceptual gaps in the national strategy for the implementation of the European Natura 2000 conservation policy in Greece. *Biological Conservation,* 142(1), 221-237.

Araujo, J.N., Mackinson, S., Stanford, R.J. & Hart, P.J.B. (2008) Exploring fisheries strategies for the western English Channel using an ecosystem model. *Ecological Modelling,* 210(4), 465-477.

Araujo, J.N., Mackinson, S., Stanford, R.J., Sims, D.W., Southward, A.J., Hawkins, S.J., Ellis, J.R. & Hart, P.J.B. (2006) Modelling food web interactions, variation in plankton production, and fisheries in the western English Channel ecosystem. *Marine Ecology-Progress Series,* 309175-187.

Ardron, J.A. (2008) The challenge of assessing whether the OSPAR network of marine protected areas is ecologically coherent. *Hydrobiologia,* 606(1), 45-53.

Arkema, K.K., Abramson, S.C. & Dewsbury, B.M. (2006) Marine ecosystem-based management: from characterization to implementation. *Frontiers in Ecology and the Environment,* 4(10), 525-532.

Aydin, K.Y. (2004) Age structure or functional response? Reconciling the energetics of surplus production between single-species models and ECOSIM. African Journal of Marine Science, 26(1), 289-301.

Azovsky, A.I. (2011) Species-area and species-sampling effort relationships: disentangling the effects. *Ecography,* 34(1), 18-30.

– B –

Babcock, R.C., Kelly, S., Shears, N.T., Walker, J.W. & Willis, T.J. (1999) Changes in community structure in temperate marine reserves. *Marine Ecology-Progress Series,* 189125-134.

Babcock, R.C., Shears, N.T., Alcala, A.C., Barrett, N.S., Edgar, G.J., Lafferty, K.D., McClanahan, T.R. & Russ, G.R. (2010) Decadal trends in marine reserves reveal differential rates of change in direct and indirect effects. *Proceedings of the National Academy of Sciences of the United States of America,* 107(43), 18256-18261.

Badalamenti, F., Ramos, A.A., Voultsiadou, E., Lizaso, L.J.S., D'Anna, G., Pipitone, C., Mas, J., Fernandez, J.A.R., Whitmarsh, D. & Riggio, S. (2000) Cultural and socio-economic impacts of Mediterranean marine protected areas. *Environmental Conservation,* 27(2), 110-125.

Ball, I.R., Possingham, H.P. & Watts, M.E. (2009). Marxan and relatives: Software for spatial conservation prioritisation. *Spatial conservation prioritisation: Quantitative methods and computational tools*. (ed. by A. Moilanen, K.A. Wilson &H.P. Possingham). Oxford University Press, Oxford, UK.

Balmford, A. (2003) Conservation planning in the real world: South Africa shows the way. *Trends in Ecology & Evolution,* 18(435-438).

Balmford, A., Gravestock, P., Hockley, N., McClean, C.J. & Roberts, C.M. (2004) The worldwide costs of marine protected areas. *Proceedings of the National Academy of Sciences of the United States of America,* 101(26), 9694-9697.

Ban, N.C., Adams, V., Pressey, R.L. & Hicks, J. (2011) Promise and problems for estimating management costs of marine protected areas. *Conservation Letters,* 4(3), 241-252.

Ban, N.C. & Klein, C.J. (2009) Spatial socioeconomic data as a cost in systematic marine conservation planning. *Conservation Letters,* 2(5), 206-215.

Ban, N.C., Picard, C.R. & Vincent, A.C.J. (2009) Comparing and Integrating Community-Based and Science-Based Approaches to Prioritizing Marine Areas for Protection. *Conservation Biology,* 23(4), 899-910.

Beddington, J.R., Agnew, D.J. & Clark, C.W. (2007) Current problems in the management of marine fisheries. *Science,* 316(5832), 1713-1716.

Behrens, M.D. & Lafferty, K.D. (2004) Effects of marine reserves and urchin disease on southern Californian rocky reef communities. *Marine Ecology-Progress Series,* 279129-139.

Bhat, M.G. (2003) Application of non-market valuation to the Florida Keys marine reserve management. *Journal of Environmental Management,* 67(4), 315-325.

Birkeland, C. & Dayton, P.K. (2005) The importance in fishery management of leaving the big ones. *Trends in Ecology & Evolution,* 20(7), 356-358.

Bladt, J., Strange, N., Abildtrup, J., Svenning, J.-C. & Skov, F. (2009) Conservation efficiency of geopolitical coordination in the EU. *Journal for Nature Conservation,* 17(2), 72-86.

Blyth-Skyrme, R.E., Kaiser, M.J., Hiddink, J.G., Edwards-Jones, G. & Hart, P.J.B. (2006) Conservation benefits of temperate marine protected areas: Variation among fish species. *Conservation Biology,* 20(3), 811-820.

Bohnsack, J.A. (1998) Application of marine reserves to reef fisheries management. *Australian Journal of Ecology,* 23(3), 298-304.

Bohnsack, J.A., Ault, J.S. & Causey, B. (2004). Why have no-take marine protected areas? *Aquatic Protected Areas as Fisheries Management Tools*. (ed. by J.B. Shipley), pp. 185-193. American Fisheries Society, Bethesda.

Borges, P.A.V., Hortal, J., Gabriel, R. & Homem, N. (2009) Would species richness estimators change the observed species area relationship? *Acta Oecologica,* 35(1), 149-156.

Borrini-Feyerabend, G. (1999). Collaborative Management of Protected Areas. *Partnership for Protection: New Strategies for Planning and Management for Protected Areas*. (ed. by S. Stolton &N. Dudley). Earthscan Publications Ltd., London.

Botsford, Hastings & Gaines (2001) Dependence of sustainability on the configuration of marine reserves and larval dispersal distance. *Ecology Letters,* 4(2), 144-150.

Botsford, L.W., Castilla, J.C. & Peterson, C.H. (1997) The management of fisheries and marine ecosystems. *Science,* 277(5325), 509-515.

Botsford, L.W., Micheli, F. & Hastings, A. (2003) Principles for the design of marine reserves. *Ecological Applications,* 13(1), S25-S31.

Brierley, A.S. & Kingsford, M.J. (2009) Impacts of Climate Change on Marine Organisms and Ecosystems. *Current Biology,* 19(14), R602-R614.

Brose, U. (2002) Estimating species richness of pitfall catches by non-parametric estimators. *Pedobiologia,* 46(2), 101-107.

– C –

Caddy, J.F. (1975) Spatial model for an exploited shellfish population, and its application to Georges Bank Scallop fishery. *Journal of the Fisheries Research Board of Canada,* 32(8), 1305-1328.

Carpenter, K.E., Abrar, M., Aeby, G., Aronson, R.B., Banks, S., Bruckner, A., Chiriboga, A., Cortés, J., Delbeek, J.C., DeVantier, L., Edgar, G.J., Edwards, A.J., Fenner, D., Guzmán, H.M., Hoeksema, B.W., Hodgson, G., Johan, O., Licuanan, W.Y., Livingstone, S.R., Lovell, E.R., Moore, J.A., Obura, D.O., Ochavillo, D., Polidoro, B.A., Precht, W.F., Quibilan, M.C., Reboton, C., Richards, Z.T., Rogers, A.D., Sanciangco, J., Sheppard, A., Sheppard, C., Smith, J., Stuart, S., Turak, E., Veron, J.E.N., Wallace, C., Weil, E. & Wood, E. (2008) One-Third of Reef-Building Corals Face Elevated Extinction Risk from Climate Change and Local Impacts. *Science,* 321(5888), 560-563.

Carpentier, A., Martin, C.S. & Vaz, S. (2009) Channel habitat atlas for marine resource management, final report (CHARM phase II). Interreg 3a Programme. IFREMER, Boulogne-sur-mer, France.

Carr, M.H., Neigel, J.E., Estes, J.A., Andelman, S., Warner, R.R. & Largier, J.L. (2003) Comparing marine and terrestrial ecosystems: Implications for the design of coastal marine reserves. *Ecological Applications,* 13(1), S90-S107.

Carwardine, J., Klein, C.J., Wilson, K.A., Pressey, R.L. & Possingham, H.P. (2009) Hitting the target and missing the point: target based conservation planning in context. *Conservation Letters,* 2(1), 3-10.

Carwardine, J., Wilson, K.A., Watts, M., Etter, A., Klein, C.J. & Possingham, H.P. (2008) Avoiding Costly Conservation Mistakes: The Importance of Defining Actions and Costs in Spatial Priority Setting. *Plos One,* 3(7), e2586.

CBD (1992) Convention on Biological Diversity. United Nations Treaty Series. Available from: http://www.cbd.int/doc/legal/cbd-un-en.pdf.

CBD (2004a) COP 7 (Annex I - DecisionVII/30 Strategic Plan: future evaluation of progress) Convention on Biological Diversity. Kuala Lumpur, Malaysia. Available from: http://www.cbd.int/decision/cop/?id=7767.

CBD (2004b) COP 7 (DecisionVII/5 Marine and Coastal Biological Diversity) Convention on Biological Diversity. Kuala Lumpur, Malaysia. Available from: http://www.cbd.int/decision/cop/?id=7742.

CBD (2004c) COP 7 (DecisionVII/28 Protected Areas) Convention on Biological Diversity. Kuala Lumpur, Malaysia. Available from: http://www.cbd.int/decision/cop/?id=7765.

CBD (2004d) COP 7 (DecisionVII/30 Strategic Plan: future evaluation of progress) Convention on Biological Diversity. Kuala Lumpur, Malaysia. Available from: http://www.cbd.int/decision/cop/?id=7767.

CBD (2005) Subsidiary Body on Scientific, Technical and Technological Advice 10 - Draft Global Outcome - orientated Targets for the Programme of Work on Marine and Coastal Biological Diversity. UNEP/CBD/SBSTTA/10/8/Add.1. Bangkok, Thailand. 7-11 February 2005. United Nations Environment Programme.

CBD (2006). Decisions Adopted by the Conference of the Parties to the Convention on Biological Diversity at its Eighth Meeting (DecisionVIII/15, Annex IV). *Convention on Biological Diversity*), Curitiba, Brazil.

CBD (2009). Connecting Biodiversity and Climate Change Mitigation and Adaptation: Report of the Second Ad Hoc Technical Expert Group on Biodiversity and Climate Change. *Technical Series*), p. 126 pp, Montreal, Canada.

CBD (2010a) Convention on Biological Diversity. Global Biodiversity Outlook 3. Montréal, Canada. Available from: http://gbo3.cbd.int/.

CBD (2010b) COP 10 (Decision X/2 Strategic Plan for Biodiversity 2011-2020) Convention on Biological Diversity. Nagoya, Japan.

CBD (2011) Report of the Tenth Meeting of the Conference of the Parties to the Convention on Biological Diversity. Nagoya, Japan. Available from: http://www.cbd.int/doc/meetings/cop/cop-10/official/cop-10-27-en.pdf.

CDFG (2008) California Marine Life Protection Act Master Plan for Marine Protected Areas. California Department of Fish and Game, Sacremento, CA. http://www.dfg.ca.gov/mlpa/pdfs/revisedmp0108.pdf.

Chape, S., Blyth, S., Fish, L., Fox, P. & Spalding, M.D. (2003). United Nations List of Protected Areas. IUCN - World Conservation Union and UNEP World Conservation Monitoring Centre, Gland, Switzerland and Cambridge, UK.).

Charles, A. & Wilson, L. (2009) Human dimensions of Marine Protected Areas. *Ices Journal of Marine Science,* 66(1), 6-15.

Chiarucci, A., Enright, N.J., Perry, G.L.W., Miller, B.P. & Lamont, B.B. (2003) Performance of nonparametric species richness estimators in a high diversity plant community. *Diversity and Distributions,* 9(4), 283-295.

Chiarucci, A., Maccherini, S. & De Dominicis, V. (2001) Evaluation and monitoring of the flora in a nature reserve by estimation methods. *Biological Conservation,* 101(3), 305-314.

Chittaro, P.M., Kaplan, I.C., Keller, A. & Levin, P.S. (2010) Trade-Offs between Species Conservation and the Size of Marine Protected Areas. *Conservation Biology*, 24(1), 197-206.

Christensen, V. & Walters, C.J. (2004a) Ecopath with Ecosim: methods, capabilities and limitations. *Ecological Modelling*, 172(2-4), 109-139.

Christensen, V. & Walters, C.J. (2004b) Trade-offs in ecosystem-scale optimization of fisheries management policies. *Bulletin of Marine Science*, 74(3), 549-562.

Christensen, V., Walters, C.J. & Pauly, D. (2005) Ecopath with Ecosim: a user's guide. Fisheries Centre, University of British Columbia. November 2005 edition. Available from: http://www.ecopath.org.

Christie, P., McCay, B.J., Miller, M.L., Lowe, C., White, A.T., Stoffle, R., Fluharty, D.L., McManus, L.T., Chuenpagdee, R., Pomeroy, C., Suman, D.O., Blount, B.G., Huppert, D., Eisma, R.L.V., Oracion, E., Lowry, K. & Pollnac, R.B. (2003) Toward developing a complete understanding: A social science research agenda for marine protected areas. *Fisheries*, 28(12), 22-26.

Cinner, J.E., Marnane, M.J. & McClanahan, T.R. (2005) Conservation and community benefits from traditional coral reef management at Ahus Island, Papua New Guinea. *Conservation Biology*, 19(6), 1714-1723.

Claudet, J., Guidetti, P., Mouillot, D., Shears, N.T. & Micheli, F. (2011). Ecological effects of marine protected areas: conservation, restoration, and functioning. *Marine Protected Areas: A multidisciplinary approach*. (ed. by J.C. Claudet). Cambridge University Press, Cambridge.

Claudet, J., Osenberg, C.W., Benedetti-Cecchi, L., Domenici, P., Garcia-Charton, J.-A., Perez-Ruzafa, A., Badalamenti, F., Bayle-Sempere, J., Brito, A., Bulleri, F., Culioli, J.-M., Dimech, M., Falcon, J.M., Guala, I., Milazzo, M., Sanchez-Meca, J., Somerfield, P.J., Stobart, B., Vandeperre, F., Valle, C. & Planes, S. (2008) Marine reserves: size and age do matter. *Ecology Letters*, 11(5), 481-489.

Claudet, J., Pelletier, D., Jouvenel, J.Y., Bachet, F. & Galzin, R. (2006) Assessing the effects of marine protected area (MPA) on a reef fish assemblage in a northwestern Mediterranean marine reserve: Identifying community-based indicators. *Biological Conservation*, 130(3), 349-369.

Cogalniceanu, D. & Cogalniceanu, G.C. (2010) An enlarged European Union challenges priority settings in conservation. *Biodiversity and Conservation*, 19(5), 1471-1483.

Coggan, R.A. & Diesing, M. (2011) The seabed habitats of the central English Channel: A generation on from Holme and Cabioch, how do their interpretations match-up to modern mapping techniques? *Continental Shelf Research*, 31(2, Supplement), S132-S150.

Coleman, F.C. & Williams, S.L. (2002) Overexploiting marine ecosystem engineers: potential consequences for biodiversity. *Trends in Ecology & Evolution*, 17(1), 40-44.

Coltman, N., Golding, N. & Verling, E. (2008). Developing a broadscale predictive EUNIS habitat map for the MESH study area. Available from: http://www.searchmesh.net/pdf/MESH%20EUNIS%20model.pdf).

Colwell, R.K. (2009) EstimateS and User's Guide: Statistical estimation of species richness and shared species from samples. Version 8.2. Available from: http://viceroy.eeb.uconn.edu/estimates.

Colwell, R.K. & Coddington, J.A. (1994) Estimating terrestrial biodiversity through extrapolation. *Philosophical Transactions of the Royal Society B-Biological Sciences*, 345(1311), 101-118.

Colwell, R.K., Mao, C.X. & Chang, J. (2004) Interpolating, extrapolating, and comparing incidence-based species accumulation curves. *Ecology*, 85(10), 2717-2727.

Cowling, R.M., Knight, A.T., Faith, D.P., Ferrier, S., Lombard, A.T., Driver, A., Rouget, M., Maze, K. & Desmet, P.G. (2004) Nature conservation requires more than a passion for species. *Conservation Biology*, 18(6), 1674-1676.

Cowling, R.M., Knight, A.T., Privett, S.D.J. & Sharma, G. (2010) Invest in Opportunity, Not Inventory of Hotspots. *Conservation Biology*, 24(2), 633-635.

Cowling, R.M. & Pressey, R.L. (2003) Introduction to systematic conservation planning in the Cape Floristic Region. *Biological Conservation*, 112(1-2), 1-13.

Cowling, R.M., Pressey, R.L., Rouget, M. & Lombard, A.T. (2003a) A conservation plan for a global biodiversity hotspot - the Cape Floristic Region, South Africa. *Biological Conservation*, 112(1-2), 191-216.

Cowling, R.M., Pressey, R.L., Sims-Castley, R., le Roux, A., Baard, E., Burgers, C.J. & Palmer, G. (2003b) The expert or the algorithm? Comparison of priority conservation areas in the Cape Floristic Region identified by park managers and reserve selection software. *Biological Conservation*, 112(1-2), 147-167.

Cowling, R.M. & Wilhelm-Rechmann, A. (2007) Social assessment as a key to conservation success. *Oryx*, 41(2), 135-136.

Crowder, L.B., Osherenko, G., Young, O.R., Airame, S., Norse, E.A., Baron, N., Day, J.C., Bouvere, F., Ehler, C.N., Halpern, B.S., Langdon, S.J., McLeod, K.L., Ogden, J.C., Peach, R.E., Rosenberg, A.A. & Wilson, J.A. (2006) Resolving mismatches in US ocean governance. *Science*, 313(5787), 617-618.

– D –

Daskalov, G.M., Mackinson, S. & Mulligan, B. (2011) Modelling possible food-web effects of aggregate dredging in the eastern English Channel. Marine Environmental Protection Fund (MEPF). Ref No: MEPF 08/P37.

Dauvin, J.C., Thiebaut, E., Gesteira, J.L.G., Ghertsosa, K., Gentil, F., Ropert, M. & Sylvand, B. (2004) Spatial structure of a subtidal macrobenthic community in the Bay of Veys (western Bay of Seine, English Channel). *Journal of Experimental Marine Biology and Ecology*, 307(2), 217-235.

Davis, D. & Tisdell, C. (1996) Economic management of recreational scuba diving and the environment. *Journal of Environmental Management*, 48(3), 229-248.

Day, J.C. & Roff, J.C. (2000). Planning for representative marine protected areas: A framework for Canada's oceans. *Report Prepared for World Wildlife Fund Canada*), p. 134 pp, Toronto.

De Santo, E.M. & Jones, P.J.S. (2007) Offshore marine conservation policies in the North East Atlantic: Emerging tensions and opportunities. *Marine Policy*, 31(3), 336-347.

DEFRA (2010) Accessing and developing the required biophysical datasets and data layers for marine protected areas network planning and wider marine spatial planning purposes. Report No 24: Task 2i. Intertidal Habitats Data Layer. Department for Environment Food and Rural Affairs. Project Code: MB0102 Marine Biodiversity R&D Programme.

DEFRA (2012) Marine Conservation Zones: Consultation on proposals for designation in 2013. Department for Environment Food & Rural Affairs. Available from: http://www.defra.gov.uk/consult/files/mcz-condoc-121213.pdf.

Delavenne, J., Metcalfe, K., Smith, R.J., Vaz, S., Martin, C.S., Dupuis, L., Coppin, F. & Carpentier, A. (2012) Systematic conservation planning in the eastern English Channel: comparing the Marxan and Zonation decision-support tools. *Ices Journal of Marine Science,* 69(1), 75-83.

Desmet, P. & Cowling, R. (2004) Using the species-area relationship to set baseline targets for conservation. *Ecology and Society,* 9(2), 11.

Desroy, N., Warembourg, C., Dewarumez, J.M. & Dauvin, J.C. (2003) Macrobenthic resources of the shallow soft-bottom sediments in the eastern English Channel and southern North Sea. *Ices Journal of Marine Science,* 60(1), 120-131.

Dimitrakopoulos, P.G., Jones, N., Iosifides, T., Florokapi, I., Lasda, O., Paliouras, F. & Evangelinos, K.I. (2010) Local attitudes on protected areas: Evidence from three Natura 2000 wetland sites in Greece. *Journal of Environmental Management,* 91(9), 1847-1854.

Dimitrakopoulos, P.G., Memtsas, D. & Troumbis, A.Y. (2004) Questioning the effectiveness of the Natura 2000 Special Areas of Conservation strategy: the case of Crete. *Global Ecology and Biogeography,* 13(3), 199-207.

Dinmore, T.A., Duplisea, D.E., Rackham, B.D., Maxwell, D.L. & Jennings, S. (2003) Impact of a large-scale area closure on patterns of fishing disturbance and the consequences for benthic communities. *Ices Journal of Marine Science,* 60(2), 371-380.

– E –

EC (1979). Council Directive 79/409/EEC on the conservation of wild birds. Official Journal L, 103:1).

EC (1992). Council Directive 92/43/EEC on the conservation of natural habitats and of wild fauna and flora. Official Journal L, 206: 0007-0050).

EC (1997) Criteria for assessing national lists of pSCI at biogeographic level. Habitats Committee.

EC (2002a). Commission Working Document on Natura 2000. European Commission, Brussels. Available from: http://ec.europa.eu/environment/nature/info/pubs/docs/nat2000/2002_faq_en.pdf).

EC (2002b) Towards a strategy to protect and conserve the marine environment. European Commission, Brussels. Available from: http://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=COM:2002:0539:FIN:EN:PDF.

EC (2007) Guidelines for the establishment of the Natura 2000 network in the marine environment. Application of the Habitats and Birds Directives. European Commission, Brussels. Available from: http://ec.europa.eu/environment/nature/natura2000/marine/docs/marine_guidelines.pdf.

EC (2008a) The Common Fisheries Policy, A User's Guide. European Commission, Brussels, Belgium. Available from: http://ec.europa.eu/fisheries/documentation/publications/pcp2008_en.pdf.

EC (2008b). Council Directive 2008/56/EC establishing a framework for community action in the field of marine environmental policy (Marine Strategy Framework Directive). Official Journal L, 164:19).

EC (2010) Council conclusions on Convention on Biological Diversity: Outcome of and follow-up to the Nagoya Conference. 3061st Environment Council meeting Brussels.

EC (2011) Our life insurance, our natural capital: an EU biodiversity strategy to 2020. European Commission, Brussels.

Edgar, G.J., Barrett, N.S. & Stuart-Smith, R.D. (2009) Exploited reefs protected from fishing transform over decades into conservation features otherwise absent from seascapes. *Ecological Applications,* 19(8), 1967-1974.

EEA (2006) EUNIS Habitat Classification, version 200610. European Environment Agency, Copenhagen. Available from: http://eunis.eea.europa.eu/habitats.jsp.

ESRI (2002) ArcView version 3.x for windows. Redlands, CA, USA.

ESRI (2008) ArcView version 9.3. Redlands, CA, USA.

ESRI (2011) ArcGIS Desktop version 10. Redlands, CA, USA.

Evans, D. (2012) Building the European Union’s Natura 2000 network. *Nature Conservation,* 1(1), 11-26.

– F –

Farrow, S. (1996) Marine protected areas: Emerging economics. *Marine Policy,* 20(6), 439-446.

Fenberg, P.B., Caselle, J.E., Claudet, J., Clemence, M., Gaines, S.D., Antonio Garcia-Charton, J., Goncalves, E.J., Grorud-Colvert, K., Guidetti, P., Jenkins, S.R., Jones, P.J.S., Lester, S.E., McAllen, R., Moland, E., Planes, S. & Sorensen, T.K. (2012) The science of European marine reserves: Status, efficacy, and future needs. *Marine Policy,* 36(5), 1012-1021.

Fernandes, L., Day, J., Kerrigan, B., Breen, D., De'ath, G., Mapstone, B., Coles, R., Done, T., Marsh, H., Poiner, I., Ward, T., Williams, D. & Kenchington, R. (2009) A process to design a network of marine no-take areas: Lessons from the Great Barrier Reef. *Ocean & Coastal Management,* 52(8), 439-447.

Fernandes, L., Day, J., Lewis, A., Slegers, S., Kerrigan, B., Breen, D., Cameron, D., Jago, B., Hall, J., Lowe, D., Innes, J., Tanzer, J., Chadwick, V., Thompson, L., Gorman, K., Simmons, M., Barnett, B., Sampson, K., De'ath, G., Mapstone, B., Marsh, H., Possingham, H., Ball, I., Ward, T., Dobbs, K., Aumend, J., Slater, D. & Stapleton, K. (2005) Establishing representative no-take areas in the Great Barrier Reef: Large-scale implementation of theory on marine protected areas. *Conservation Biology,* 19(6), 1733-1744.

Ferrier, S. & Wintle, B.A. (2009). Quantitative approaches to spatial conservation prioritization: Matching the solution to the need. *Spatial conservation prioritisation: Quantitative methods and computational tools*. (ed. by A. Moilanen, K.A. Wilson &H.P. Possingham). Oxford University Press, Oxford, UK.

Floeter, S.R., Halpern, B.S. & Ferreira, C.E.L. (2006) Effects of fishing and protection on Brazilian reef fishes. *Biological Conservation,* 128(3), 391-402.

Fox, H.E., Mascia, M.B., Basurto, X., Costa, A., Glew, L., Heinemann, D., Karrer, L.B., Lester, S.E., Lombana, A.V., Pomeroy, R.S., Recchia, C.A., Roberts, C.M., Sanchirico, J.N., Pet-Soede, L. & White, A.T. (2012a) Reexamining the science of marine protected areas: linking knowledge to action. *Conservation Letters,* 5(1), 1-10.

Fox, H.E., Soltanoff, C.S., Mascia, M.B., Haisfield, K.M., Lombana, A.V., Pyke, C.R. & Wood, L. (2012b) Explaining global patterns and trends in marine protected area (MPA) development. *Marine Policy,* 36(5), 1131-1138.

Fulton, E.A. & Smith, A.D.M. (2004) Lessons learnt from a comparison of three ecosystem models for Port Phillip Bay, Australia. African Journal of Marine Science, 26(1), 219-243.

– G –

Gaines, S.D., Lester, S.E., Grorud-Colvert, K., Costello, C. & Pollnac, R. (2010a) Evolving science of marine reserves: New developments and emerging research frontiers. *Proceedings of the National Academy of Sciences of the United States of America,* 107(43), 18251-18255.

Gaines, S.D., White, C., Carr, M.H. & Palumbi, S.R. (2010b) Designing marine reserve networks for both conservation and fisheries management. *Proceedings of the National Academy of Sciences of the United States of America,* 107(43), 18286-18293.

Gallo, J.A., Pasquini, L., Reyers, B. & Cowling, R.M. (2009) The role of private conservation areas in biodiversity representation and target achievement within the Little Karoo region, South Africa. *Biological Conservation,* 142(2), 446-454.

Garcia-Charton, J.A., Perez-Ruzafa, A., Saanchez-Jerez, P., Bayle-Sempere, J.T., Renones, O. & Moreno, D. (2004) Multi-scale spatial heterogeneity, habitat structure, and the effect of marine reserves on Western Mediterranean rocky reef fish assemblages. *Marine Biology,* 144(1), 161-182.

Gaston, K.J., Jackson, S.E., Nagy, A., Cantu-Salazar, L. & Johnson, M. (2008) Protected areas in Europe: Principle and practice. *Annals of the New York Academy of Sciences,* 1134(1), 97-119.

GBRMPA (2002) Biophysical Operating Principles as recommended by the Scientific Steering Committee for the Representative Areas Program. Technical Information Sheet #6. Great Barrier Reef Marine Park Authority, Townsville, Australia. http://www.gbrmpa.gov.au/__data/assets/pdf_file/0011/6212/tech_sheet_06.pdf.

Gell, F.R. & Roberts, C.M. (2003) Benefits beyond boundaries: the fishery effects of marine reserves. *Trends in Ecology & Evolution,* 18(9), 448-455.

Giakoumi, S., Katsanevakis, S., Vassilopoulou, V., Panayotidis, P., Kavadas, S., Issaris, Y., Kokkali, A., Frantzis, A., Panou, A. & Mavromati, G. (2012) Could European marine conservation policy benefit from systematic conservation planning? *Aquatic Conservation: Marine and Freshwater Ecosystems,* 22(6), 762-775.

Goni, R., Quetglas, A. & Renones, O. (2006) Spillover of spiny lobsters Palinurus elephas from a marine reserve to an adjoining fishery. *Marine Ecology-Progress Series,* 308207-219.

Gotelli, N.J. & Colwell, R.K. (2001) Quantifying biodiversity: procedures and pitfalls in the measurement and comparison of species richness. *Ecology Letters,* 4(4), 379-391.

Grantham, B.A., Eckert, G.L. & Shanks, A.L. (2003) Dispersal potential of marine invertebrates in diverse habitats. *Ecological Applications,* 13(1), S108-S116.

Grantham, H.S., Agostini, V.N., Wilson, J., Mangubhai, S., Hidayat, N., Muljadi, A., Muhajir, Rotinsulu, C., Mongdong, M., Beck, M.W. & Possingham, H.P. (2013) A comparison of zoning analyses to inform the planning of a marine protected area network in Raja Ampat, Indonesia. *Marine Policy,* 38(0), 184-194.

Grodzinska-Jurczak, M. & Cent, J. (2011) Expansion of Nature Conservation Areas: Problems with Natura 2000 Implementation in Poland? *Environmental Management,* 47(1), 11-27.

Groves, C.R., Jensen, D.B., Valutis, L.L., Redford, K.H., Shaffer, M.L., Scott, J.M., Baumgartner, J.V., Higgins, J.V., Beck, M.W. & Anderson, M.G. (2002) Planning for biodiversity conservation: Putting conservation science into practice. *Bioscience,* 52(6), 499-512.

Guarderas, A.P., Hacker, S.D. & Lubchenco, J. (2011) Ecological effects of marine reserves in Latin America and the Caribbean. *Marine Ecology-Progress Series,* 429219-U847.

Guilhaumon, F., Gimenez, O., Gaston, K.J. & Mouillot, D. (2008) Taxonomic and regional uncertainty in species-area relationships and the identification of richness hotspots. *Proceedings of the National Academy of Sciences of the United States of America,* 105(40), 15458-15463.

Guilhaumon, F., Mouillot, D. & Gimenez, O. (2010) mmSAR: an R-package for multimodel species-area relationship inference. *Ecography,* 33(2), 420-424.

– H –

Halpern, B.S. (2003) The impact of marine reserves: Do reserves work and does reserve size matter? *Ecological Applications,* 13(1), S117-S137.

Halpern, B.S., Lester, S.E. & McLeod, K.L. (2010) Placing marine protected areas onto the ecosystem-based management seascape. *Proceedings of the National Academy of Sciences of the United States of America,* 107(43), 18312-18317.

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.D., Steneck, R. & Watson, R. (2008) A global map of human impact on marine ecosystems. *Science,* 319(5865), 948-952.

Halpern, B.S. & Warner, R.R. (2002) Marine reserves have rapid and lasting effects. *Ecology Letters,* 5(3), 361-366.

Halpern, B.S. & Warner, R.R. (2003) Matching marine reserve design to reserve objectives. *Proceedings of the Royal Society of London Series B-Biological Sciences,* 270(1527), 1871-1878.

Hargreaves-Allen, V., Mourato, S. & Milner-Gulland, E.J. (2011) A Global Evaluation of Coral Reef Management Performance: Are MPAs Producing Conservation and Socio-Economic Improvements? *Environmental Management,* 47(4), 684-700.

Harley, C.D.G., Hughes, A.R., Hultgren, K.M., Miner, B.G., Sorte, C.J.B., Thornber, C.S., Rodriguez, L.F., Tomanek, L. & Williams, S.L. (2006) The impacts of climate change in coastal marine systems. *Ecology Letters,* 9(2), 228-241.

Harmelin-Vivien, M., Le Direach, L., Bayle-Sempere, J., Charbonnel, E., Garcia-Charton, J.A., Ody, D., Perez-Ruzafa, A., Renones, O., Sanchez-Jerez, P. & Valle, C. (2008) Gradients of abundance and biomass across reserve boundaries in six Mediterranean marine protected areas: Evidence of fish spillover? *Biological Conservation,* 141(7), 1829-1839.

Harmelin, J.G., Bachet, F. & Garcia, G. (1995) Mediterranean marine reserves: Fish indices as tests of protection efficiency. *Marine Ecology,* 16(3), 233-250.

Harrop, S.R. (1999) Conservation Regulation: A Backward Step for Biodiversity? *Biodiversity and Conservation,* 8(5), 679-707.

Harrop, S.R. (2011) 'Living In Harmony With Nature'? Outcomes of the 2010 Nagoya Conference of the Convention on Biological Diversity. *Journal of Environmental Law,* 23(1), 117-128.

Harrop, S.R. & Pritchard, D.J. (2011) A hard instrument goes soft: The implications of the Convention on Biological Diversity's current trajectory. *Global Environmental Change,* 21(2), 474-480.

Haslett, J.R., Berry, P.M., Bela, G., Jongman, R.H.G., Pataki, G., Samways, M.J. & Zobel, M. (2010) Changing conservation strategies in Europe: a framework integrating ecosystem services and dynamics. *Biodiversity and Conservation,* 19(10), 2963-2977.

Herzog, S.K., Kessler, M. & Cahill, T.M. (2002) Estimating species richness of tropical bird communities from rapid assessment data. *Auk,* 119(3), 749-769.

Higgins, R.M., Vandeperre, F., Perez-Ruzafa, A. & Santos, R.S. (2008) Priorities for fisheries in marine protected area design and management: Implications for artisanal-type fisheries as found in southern Europe. *Journal for Nature Conservation,* 16(4), 222-233.

Hilborn, R., Stokes, K., Maguire, J.-J., Smith, T., Botsford, L.W., Mangel, M., Orensanz, J., Parma, A., Rice, J., Bell, J., Cochrane, K.L., Garcia, S., Hall, S.J., Kirkwood, G.P., Sainsbury, K., Stefansson, G. & Walters, C. (2004) When can marine reserves improve fisheries management? *Ocean and Coastal Management,* 47(3-4), 197-205.

Hortal, J., Borges, P.A.V. & Gaspar, C. (2006) Evaluating the performance of species richness estimators: sensitivity to sample grain size. *Journal of Animal Ecology,* 75(1), 274-287.

Horwood, J.W., Nichols, J.H. & Milligan, S. (1998) Evaluation of closed areas for fish stock conservation. *Journal of Applied Ecology,* 35(6), 893-903.

Hoskin, M.G., Coleman, R.A., von Carlshausen, E. & Davis, C.M. (2011) Variable population responses by large decapod crustaceans to the establishment of a temperate marine no-take zone. *Canadian Journal of Fisheries and Aquatic Sciences,* 68(2), 185-200.

Hughes, T.P., Baird, A.H., Bellwood, D.R., Card, M., Connolly, S.R., Folke, C., Grosberg, R., Hoegh-Guldberg, O., Jackson, J.B.C., Kleypas, J., Lough, J.M., Marshall, P., Nyström, M., Palumbi, S.R., Pandolfi, J.M., Rosen, B. & Roughgarden, J. (2003) Climate Change, Human Impacts, and the Resilience of Coral Reefs. *Science,* 301(5635), 929-933.

Hutchings, J.A. (2000) Collapse and recovery of marine fishes. *Nature,* 406(6798), 882-885.

– I –

Inger, R., Attrill, M.J., Bearhop, S., Broderick, A.C., Grecian, W.J., Hodgson, D.J., Mills, C., Sheehan, E., Votier, S.C., Witt, M.J. & Godley, B.J. (2009) Marine renewable energy: potential benefits to biodiversity? An urgent call for research. *Journal of Applied Ecology,* 46(6), 1145-1153.

Ioja, C.I., Patroescu, M., Rozylowicz, L., Popescu, V.D., Verghelet, M., Zotta, M.I. & Felciuc, M. (2010) The efficacy of Romania's protected areas network in conserving biodiversity. *Biological Conservation,* 143(11), 2468-2476.

Islam, M.S. & Tanaka, M. (2004) Impacts of pollution on coastal and marine ecosystems including coastal and marine fisheries and approach for management: a review and synthesis. *Marine Pollution Bulletin,* 48(7-8), 624-649.

IUCN-WCPA (2008) Establishing Marine Protected Area Networks - Making It Happen. IUCN World Commission on Protected Areas, National Oceanic and Atmospheric Administration and The Nature Conservancy. 118 pp.

IUCN (1993) Parks for Life: Report of the IVth IUCN World Congress on National Parks and Protected Areas. IUCN, Gland, Switzerland.

IUCN (2000) Protected Areas, Benefits beyond Boundaries. International Union for Conservation of Nature and Natural Resources (IUCN).Gland, Switzerland.

IUCN (2003) Recommendation 5.22: building a global system of marine and coastal protected areas. Vth IUCN World Parks Congress, Durban, South Africa. IUCN, Gland, Switzerland.

– J –

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(5530), 629-638.

Jackson, S.F., Kershaw, M. & Gaston, K.J. (2004) The performance of procedures for selecting conservation areas: waterbirds in the UK. *Biological Conservation,* 118(2), 261-270.

Jameson, S.C., Tupper, M.H. & Ridley, J.M. (2002) The three screen doors: can marine "protected" areas be effective? *Marine Pollution Bulletin,* 44(11), 1177-1183.

Jenness, J. (2005) Repeating Shapes for ArcView. Jenness Enterprises. Available from: http://www.jennessent.com/arcview/repeat_shapes.htm.

Jennings, S. (2000) Patterns and prediction of population recovery in marine reserves. *Reviews in Fish Biology and Fisheries,* 10(2), 209-231.

Jentoft, S., Chuenpagdee, R. & Pascual-Fernandez, J.J. (2011) What are MPAs for: On goal formation and displacement. *Ocean & Coastal Management,* 54(1), 75-83.

JNCC & Natural England (2010) Marine Conservation Zone Project. Ecological Network Guidance. Joint Nature Conservation Committee and Natural England. Available from: http://www.jncc.gov.uk/pdf/100608_ENG_v10.pdf.

Jones, G.P., McCormick, M.I., Srinivasan, M. & Eagle, J.V. (2004) Coral decline threatens fish biodiversity in marine reserves. *Proceedings of the National Academy of Sciences of the United States of America,* 101(21), 8251-8253.

Jones, P.J.S. (1999) Marine nature reserves in Britain: past lessons, current status and future issues. *Marine Policy,* 23(4-5), 375-396.

Jones, P.J.S. (2009) Equity, justice and power issues raised by no-take marine protected area proposals. *Marine Policy,* 33(5), 759-765.

Jones, P.J.S. (2012) Marine protected areas in the UK: challenges in combining top-down and bottom-up approaches to governance. . *Environmental Conservation,* 39(3), 248-258.

Jouvenel, J.Y., Bachet, F., Harmelin, J.G. & Bellan-Santini, D. (2004) Biological monitoring of a marine reserve ('Cote bleue' marine park, Marseilles Bay, Mediterranean Sea, France). *Revue D Ecologie-La Terre Et La Vie,* 59(1-2), 243-251.

Justus, J., Fuller, T. & Sarkar, S. (2008) Influence of representation targets on the total area of conservation-area networks. *Conservation Biology,* 22(3), 673-682.

– K –

Kaiser, M.J. (2005) Are marine protected areas a red herring or fisheries panacea? *Canadian Journal of Fisheries and Aquatic Sciences,* 62(5), 1194-1199.

Kaplan, D.M., Botsford, L.W., O'Farrell, M.R., Gaines, S.D. & Jorgensen, S. (2009) Model-based assessment of persistence in proposed marine protected area designs. *Ecological Applications*, 19(2), 433-448.

Kareiva, P. & Marvier, M. (2012) What is conservation science? *Bioscience,* 62(11), 962-969.

Kark, S., Levin, N., Grantham, H.S. & Possingham, H.P. (2009) Between-country collaboration and consideration of costs increase conservation planning efficiency in the Mediterranean Basin. *Proceedings of the National Academy of Sciences of the United States of America,* 106(36), 15368-15373.

Keller, B.D., Gleason, D.F., McLeod, E., Woodley, C.M., Airame, S., Causey, B.D., Friedlander, A.M., Grober-Dunsmore, R., Johnson, J.E., Miller, S.L. & Steneck, R.S. (2009) Climate Change, Coral Reef Ecosystems, and Management Options for Marine Protected Areas. *Environmental Management*, 44(6), 1069-1088.

Kellner, J.B., Tetreault, I., Gaines, S.D. & Nisbet, R.M. (2007) Fishing the line near marine reserves in single and multispecies fisheries. *Ecological Applications,* 17(4), 1039-1054.

Klein, C.J., Chan, A., Kircher, L., Cundiff, A.J., Gardner, N., Hrovat, Y., Scholz, A., Kendall, B.E. & Airame, S. (2008a) Striking a balance between biodiversity conservation and socioeconomic viability in the design of marine protected areas. *Conservation Biology,* 22(3), 691-700.

Klein, C.J., Steinback, C., Scholz, A.J. & Possingham, H.P. (2008b) Effectiveness of marine reserve networks in representing biodiversity and minimizing impact to fishermen: a comparison of two approaches used in California. *Conservation Letters,* 1(1), 44-51.

Klein, C.J., Steinback, C., Watts, M., Scholz, A.J. & Possingham, H.P. (2010) Spatial marine zoning for fisheries and conservation. *Frontiers in Ecology and the Environment,* 8(7), 349-353.

Knight, A.T., Cowling, R.M. & Campbell, B.M. (2006a) An operational model for implementing conservation action. *Conservation Biology,* 20(2), 408-419.

Knight, A.T., Cowling, R.M., Rouget, M., Balmford, A., Lombard, A.T. & Campbell, B.M. (2008) Knowing but not doing: Selecting priority conservation areas and the research-implementation gap. *Conservation Biology,* 22(3), 610-617.

Knight, A.T., Driver, A., Cowling, R.M., Maze, K., Desmet, P.G., T Lombard, A., Rouget, M., Botha, M.A., Boshoff, A.E., Castley, G., Goodman, P.S., MacKinnon, K., Pierce, S.M., Sims-Castley, R., Stewart, W.I. & Von Hase, A. (2006b) Designing systematic conservation assessments that promote effective implementation: Best practice from South Africa. *Conservation Biology,* 20(3), 739-750.

– L –

Laurel, B.J. & Bradbury, I.R. (2006) "Big" concerns with high latitude marine protected areas (MPAs): trends in connectivity and MPA size. *Canadian Journal of Fisheries and Aquatic Sciences,* 63(12), 2603-2607.

Leathwick, J., Moilanen, A., Francis, M., Elith, J., Taylor, P., Julian, K., Hastie, T. & Duffy, D. (2008) Novel methods for the design and evaluation of marine protected areas in offshore waters. *Conservation Letters*, 1(2), 91-102.

Lee, J., South, A.B. & Jennings, S. (2010) Developing reliable, repeatable, and accessible methods to provide high-resolution estimates of fishing-effort distributions from vessel monitoring system (VMS) data. *Ices Journal of Marine Science*, 67 (6), 1260-1271.

Lees, K. & Mackinson, S. (2007) An ecopath model of the Irish Sea: ecosystem properties and sensitivity analysis. Science Series Technical Report no.138, Cefas, Lowestoft.

Leslie, H., Ruckelshaus, M., Ball, I.R., Andelman, S. & Possingham, H.P. (2003) Using siting algorithms in the design of marine reserve networks. *Ecological Applications,* 13(1), S185-S198.

Leslie, H.M. (2005) Synthesis of marine conservation planning approaches. *Conservation Biology,* 19(6), 1701-1713.

Lester, S.E. & Halpern, B.S. (2008) Biological responses in marine no-take reserves versus partially protected areas. *Marine Ecology-Progress Series,* 36749-56.

Linke, S., Watts, M., Stewart, R. & Possingham, H.P. (2011) Using multivariate analysis to deliver conservation planning products that align with practitioner needs. *Ecography,* 34(2), 203-207.

Lloret, J., Zaragoza, N., Caballero, D. & Riera, V. (2008) Biological and socio-economic implications of recreational boat fishing for the management of fishery resources in the marine reserve of Cap de Creus (NW Mediterranean). *Fisheries Research,* 91(2-3), 252-259.

Lomolino, M.V. (2000) Ecology's most general, yet protean pattern: the species-area relationship. *Journal of Biogeography,* 27(1), 17-26.

Lozano-Montes, H.M., Babcock, R. & Loneragan, N.R. (2012) Exploring the effects of spatial closures in a temperate marine ecosystem in Western Australia: A case study of the western rock lobster (Panulirus cygnus) fishery. *Ecological Modelling,* 24531-40.

Lubchenco, J., Palumbi, S.R., Gaines, S.D. & Andelman, S. (2003) Plugging a hole in the ocean: The emerging science of marine reserves. *Ecological Applications,* 13(1), S3-S7.

Lynch, T.P., Wilkinson, E., Melling, L., Hamilton, R., Macready, A. & Feary, S. (2004) Conflict and impacts of divers and anglers in a marine park. *Environmental Management,* 33(2), 196-211.

– M –

Mackinson, S. & Daskalov, G. (2007) An ecosystem model of the North Sea to support an ecosystem approach to fisheries management: description and parameterisation. Science Series Technical Report no.142, Cefas, Lowestoft.

Maiorano, L., Falcucci, A., Garton, E.O. & Boitani, L. (2007) Contribution of the natura 2000 network to biodiversity conservation in Italy. *Conservation Biology,* 21(6), 1433-1444.

Malcolm, H.A., Foulsham, E., Pressey, R.L., Jordan, A., Davies, P.L., Ingleton, T., Johnstone, N., Hessey, S. & Smith, S.D.A. (2012) Selecting zones in a marine park: Early systematic planning improves cost-efficiency; combining habitat and biotic data improves effectiveness. *Ocean & Coastal Management,* 59(1), 1-12.

Mangi, S. & Hattam, C. (2009) Examining the contribution of marine protected areas to human well-being. Final Report to NERC and DEFRA. Living With Environmental Change (LWEC) programme. Available from: http://www.nerc.ac.uk/research/programmes/lwec/documents/lwec-pr-mangi-report.pdf.

Margules, C.R. & Pressey, R.L. (2000) Systematic conservation planning. *Nature*, 405(6783), 243-253.

Margules, C.R., Pressey, R.L. & Williams, P.H. (2002) Representing biodiversity: data and procedures for identifying priority areas for conservation. *Journal of Biosciences,* 27(4), 309-326.

Margules, C.R. & Sarkar, S. (2007) *Systematic Conservation Planning*, Cambrige University Press.

Martin, C.S., Carpentier, A., Vaz, S., Coppin, F., Curet, L., Dauvin, J.C., Delavenne, J., Dewarumez, J.M., Dupuis, L., Engelhard, G., Ernande, B., Foveau, A., Garcia, C., Gardel, L., Harrop, S., Just, R., Koubbi, P., Lauria, V., Meaden, G.J., Morin, J., Ota, Y., Rostiaux, E., Smith, R.J., Spilmont, N., Verin, Y., Villanueva, C. & Warembourg, C. (2009) The Channel habitat atlas for marine resource management (CHARM): an aid for planning and decision-making in an area under strong anthropogenic pressure. *Aquatic Living Resources,* 22(4), 499-508.

Mascia, M.B. & Claus, C.A. (2009) A Property Rights Approach to Understanding Human Displacement from Protected Areas: the Case of Marine Protected Areas. *Conservation Biology,* 23(1), 16-23.

Mascia, M.B., Claus, C.A. & Naidoo, R. (2010) Impacts of Marine Protected Areas on Fishing Communities. *Conservation Biology,* 24(5), 1424-1429.

MCAA (2009). Marine and Coastal Access Act 2009 Chapter 23. The Stationery Office. London. Available from: http://www.opsi.gov.uk/acts/acts2009/ukpga_20090023_en_1).

McBreen, F., Askew, N., Cameron, A., Connor, D.W., Ellwood, H. & Carter, A. (2011) UKSeaMap 2010: Predictive mapping seabed habitats in UK waters. JNCC Report 446. Joint Nature Conservation Committee, Peterborough.

McClanahan, T.R. (1999) Is there a future for coral reef parks in poor tropical countries? *Coral Reefs,* 18(4), 321-325.

McCook, L.J., Ayling, T., Cappo, M., Choat, J.H., Evans, R.D., De Freitas, D.M., Heupel, M., Hughes, T.P., Jones, G.P., Mapstone, B., Marsh, H., Mills, M., Molloy, F.J., Pitcher, C.R., Pressey, R.L., Russ, G.R., Sutton, S., Sweatman, H., Tobin, R., Wachenfeld, D.R. & Williamson, D.H. (2010) Adaptive management of the Great Barrier Reef: A globally significant demonstration of the benefits of networks of marine reserves. *Proceedings of the National Academy of Sciences of the United States of America,* 107(43), 18278-18285.

McCrea-Strub, A., Zeller, D., Sumaila, U.R., Nelson, J., Balmford, A. & Pauly, D. (2011) Understanding the cost of establishing marine protected areas. *Marine Policy,* 35(1), 1-9.

McLeod, C.R., Yeo, M., Brown, A.E., Burn, A.J., Hopkins, J.J. & Way, S.F. (2005). The Habitats Directive: selection of Special Areas of Conservation in the UK, 2nd edition. Joint Nature Conservation Committee. Peterborough. Available from: http://www.jncc.gov.uk/SACselection).

MESH (2008). MESH. The Mesh Blue Book. A summary of achievements of the MESH project. 24 pp. Available at: http://www.searchmesh.net/PDF/Final%20MESH%20EXEC%20SUM%20v7.pdf).

Metcalfe, K., Delavenne, J., Garcia, C., Foveau, A., Dauvin, J.C., Coggan, R.A., Vaz, S., Harrop, S.R. & Smith, R.J. (2013a) Impacts of data quality on the setting of conservation planning targets using the species-area relationship. *Diversity and Distributions,* 19(1), 1-13.

Metcalfe, K., Roberts, T.M., Smith, R.J. & Harrop, S.R. (2013b) Marine conservation science and governance in North-West Europe: conservation planning and international law and policy. *Marine Policy,* 39(3), 289-295.

Moffitt, E.A., Botsford, L.W., Kaplan, D.M. & O'Farrell, M.R. (2009) Marine reserve networks for species that move within a home range. *Ecological Applications,* 19(7), 1835-1847.

Moffitt, E.A., Wilson White, J. & Botsford, L.W. (2010) The utility and limitations of size and spacing guidelines for designing marine protected area (MPA) networks. *Biological Conservation,* 144(1), 306-318.

Moilanen, A., Wilson, K.A. & Possingham, H.P. (2009). Spatial Conservation Prioritization: Quantitative Methods and Computational Tools), p. 320. Oxford University Press.

Mumby, P.J., Dahlgren, C.P., Harborne, A.R., Kappel, C.V., Micheli, F., Brumbaugh, D.R., Holmes, K.E., Mendes, J.M., Broad, K., Sanchirico, J.N., Buch, K., Box, S., Stoffle, R.W. & Gill, A.B. (2006) Fishing, trophic cascades, and the process of grazing on coral reefs. *Science,* 311(5757), 98-101.

Mumby, P.J., Harborne, A.R., Williams, J., Kappel, C.V., Brumbaugh, D.R., Micheli, F., Holmes, K.E., Dahlgren, C.P., Paris, C.B. & Blackwell, P.G. (2007) Trophic cascade facilitates coral recruitment in a marine reserve. *Proceedings of the National Academy of Sciences of the United States of America,* 104(20), 8362-8367.

Murawski, S.A., Brown, R., Lai, H.L., Rago, P.J. & Hendrickson, L. (2000) Large-scale closed areas as a fishery-management tool in temperate marine systems: The Georges Bank experience. *Bulletin of Marine Science,* 66(3), 775-798.

Murawski, S.A., Wigley, S.E., Fogarty, M.J., Rago, P.J. & Mountain, D.G. (2005) Effort distribution and catch patterns adjacent to temperate MPAs. *Ices Journal of Marine Science,* 62(6), 1150-1167.

– N –

Naidoo, R., Balmford, A., Ferraro, P.J., Polasky, S., Ricketts, T.H. & Rouget, M. (2006) Integrating economic costs into conservation planning. *Trends in Ecology & Evolution,* 21(12), 681-687.

NBN (2010) National Biodiversity Network Gateway Version 3.4.1. Available from: http://data.nbn.org.uk/

Neigel, J.E. (2003) Species-area relationships and marine conservation. *Ecological Applications,* 13(1), S138-S145.

Nel, J.L., Roux, D.J., Maree, G., Kleynhans, C.J., Moolman, J., Reyers, B., Rouget, M. & Cowling, R.M. (2007) Rivers in peril inside and outside protected areas: a systematic approach to conservation assessment of river ecosystems. *Diversity and Distributions,* 13(3), 341-352.

Nhancale, B. & Smith, R. (2011) The influence of planning unit characteristics on the efficiency and spatial pattern of systematic conservation planning assessments. *Biodiversity and Conservation,* 20(8), 1821-1835.

Nicholson, E., Collen, B., Barausse, A., Blanchard, J.L., Costelloe, B.T., Sullivan, K.M.E., Underwood, F.M., Burn, R.W., Fritz, S., Jones, J.P.G., McRae, L., Possingham, H.P. & Milner-Gulland, E.J. (2012) Making Robust Policy Decisions Using Global Biodiversity Indicators. *Plos One,* 7(7), e41128.

Noss, R.F. (1987) From plant communities to landscapes in conservation inventories - a look at The Nature Conservancy (USA). *Biological Conservation*, 41(1), 11-37.

– O –

Oldfield, T.E.E., Smith, R.J., Harrop, S.R. & Leader-Williams, N. (2004) A gap analysis of terrestrial protected areas in England and its implications for conservation policy. *Biological Conservation*, 120(3), 303-309.

OSPAR (1992) Convention for the Protection of the Marine Environment of the North-East Atlantic. OSPAR Commission.

OSPAR (2003a). Guidelines for the identification and selection of marine protected areas in the OSPAR maritime area. OSPAR Commission (Reference Number: 2003-17)).

OSPAR (2003b). Joint HELCOM/OSPAR Work Programme on Marine Protected Areas. First Joint Managerial Meeting of the Helsinki and OSPAR Commissions, Agenda Item 6, Annex 7).

OSPAR (2006) Guidance on Developing an Ecologically Coherent Network of OSPAR Marine Protected Areas. OSPAR Commission (Reference Number: 2006-3).

OSPAR (2007). 2006 Report on the status of the OSPAR network of marine protected areas. OSPAR Commission Biodiversity Series, 319.).

OSPAR (2008a) Case Reports for the OSPAR List of threatened and/or declining species and habitats. OSPAR Commission. Biodiversity Series No. 358.

OSPAR (2008b) Guidance for good practice for communicating with stakeholder on the establishment and management of marine protected areas. OSPAR Commission (Reference Number: 2008-2).

OSPAR (2008c). OSPAR List of Threatened and/or Declining Species and Habitats. OSPAR Commission. (Reference Number: 2008-6)).

OSPAR (2010a) 2009/2010 Status report on the OSPAR network of marine protected areas. OSPAR Commission. Biodiversity Series.

OSPAR (2010b) The North-East Atlantic Environment Strategy. Strategy of the OSPAR Commission for the Protection of the Marine Environment of the North-East Atlantic 2010–2020. OSPAR Commission (OSPAR Reference Number: 2010-3).

– P –

Palumbi, S.R. (2003) Population genetics, demographic connectivity, and the design of marine reserves. *Ecological Applications,* 13(1), S146-S158.

Palumbi, S.R. (2004) Marine reserves and ocean neighborhoods: The spatial scale of marine populations and their management. *Annual Review of Environment and Resources,* 2931-68.

Pauly, D., Christensen, V., Dalsgaard, J., Froese, R. & Torres, F. (1998) Fishing down marine food webs. *Science,* 279(5352), 860-863.

Pauly, D., Christensen, V. & Walters, C. (2000) Ecopath, Ecosim, and Ecospace as tools for evaluating ecosystem impact of fisheries. *Ices Journal of Marine Science,* 57(3), 697-706.

Pauly, D. & Watson, R. (2005) Background and interpretation of the 'Marine Trophic Index' as a measure of biodiversity. *Philosophical Transactions of the Royal Society B-Biological Sciences*, 360(1454), 415-423.

Pauly, D., Watson, R. & Alder, J. (2005) Global trends in world fisheries: impacts on marine ecosystems and food security. *Philosophical Transactions of the Royal Society B-Biological Sciences*, 360(1453), 5-12.

Pelletier, D. & Mahevas, S. (2005) Spatially explicit fisheries simulation models for policy evaluation. *Fish and Fisheries*, 6(4), 307-349.

Pillans, S., Pillans, R.D., Johnstone, R.W., Kraft, P.G., Haywood, M.D.E. & Possingham, H.P. (2005) Effects of marine reserve protection on the mud crab Scylla serrata in a sex-biased fishery in subtropical Australia. *Marine Ecology-Progress Series*, 295201-213.

Pinnegar, J.K. & Engelhard, G.H. (2008) The 'shifting baseline' phenomenon: a global perspective. *Reviews in Fish Biology and Fisheries*, 18(1), 1-16.

Pinnegar, J.K., Polunin, N.V.C., Francour, P., Badalamenti, F., Chemello, R., Harmelin-Vivien, M.L., Hereu, B., Milazzo, M., Zabala, M., D'Anna, G. & Pipitone, C. (2000) Trophic cascades in benthic marine ecosystems: lessons for fisheries and protected-area management. *Environmental Conservation*, 27(2), 179-200.

Plaganyi, E.E. & Butterworth, D.S. (2004) A critical look at the potential of ecopath with ECOSIM to assist in practical fisheries management. African Journal of Marine Science, 26(1), 261-287.

Pollnac, R., Christie, P., Cinner, J.E., Dalton, T., Daw, T.M., Forrester, G.E., Graham, N.A.J. & McClanahan, T.R. (2010) Marine reserves as linked social-ecological systems. *Proceedings of the National Academy of Sciences of the United States of America*, 107(43), 18262-18265.

Prendergast, J.R., Quinn, R.M. & Lawton, J.H. (1999) The gaps between theory and practice in selecting nature reserves. *Conservation Biology*, 13(3), 484-492.

Pressey, R.L. (1994) Ad hoc reservations: forward or backward steps in developing representative reserve systems. *Conservation Biology*, 8(3), 662-668.

Pressey, R.L. (1999) Systematic conservation planning for the real world. *Parks*, 91-6.

Pressey, R.L., Cabeza, M., Watts, M.E., Cowling, R.M. & Wilson, K.A. (2007) Conservation planning in a changing world. *Trends in Ecology & Evolution*, 22(11), 583-592.

Pressey, R.L., Cowling, R.M. & Rouget, M. (2003) Formulating conservation targets for biodiversity pattern and process in the Cape Floristic Region, South Africa. *Biological Conservation*, 112(1-2), 99-127.

Pressey, R.L., Humphries, C.J., Margules, C.R., Vanewright, R.I. & Williams, P.H. (1993) Beyond opportunism - key principles for systematic reserve selection. *Trends in Ecology & Evolution*, 8(4), 124-128.

– R –

R Development Core Team (2008) R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. ISBN 3-900051-07-0, URL http://www.R-project.org/.

Rabaut, M., Degraer, S., Schrijvers, J., Derous, S., Bogaert, D., Maes, F., Vincx, M. & Cliquet, A. (2009) Policy analysis of the 'MPA-process' in temperate continental shelf areas. *Aquatic Conservation-Marine and Freshwater Ecosystems,* 19(5), 596-608.

Rassweiler, A., Costello, C. & Siegel, D.A. (2012) Marine protected areas and the value of spatially optimized fishery management. *Proceedings of the National Academy of Sciences,* 109(29), 11884-11889.

Rees, S., Fletcher, S., Glegg, G., Marshall, C., Rodwell, L., Jefferson, R., Campbell, M., Langmead, O., Ashley, M., Bloomfield, H., Brutto, D., Colenutt, A., Conversi, A., Earll, B., Hattam, C., Ingram, S., McKinley, E., Mee, L., Oates, J., Peckett, F., Portus, J., Reed, M., Rogers, S., Saunders, J., Scales, K. & Wynn, R. (2013) Priority questions to shape the marine and coastal policy research agenda in the United Kingdom. *Marine Policy,* 38(0), 531-537.

Reyers, B., Rouget, M., Jonas, Z., Cowling, R.M., Driver, A., Maze, K. & Desmet, P. (2007) Developing products for conservation decision-making: lessons from a spatial biodiversity assessment for South Africa. *Diversity and Distributions,* 13(5), 608-619.

Rijnsdorp, A.D., Buys, A.M., Storbeck, F. & Visser, E.G. (1998) Micro-scale distribution of beam trawl effort in the southern North Sea between 1993 and 1996 in relation to the trawling frequency of the sea bed and the impact on benthic organisms. *Ices Journal of Marine Science,* 55(3), 403-419.

Roberts, C.M. (2003) Our shifting perspectives on the oceans. *Oryx,* 37(2), 166-177.

Roberts, C.M., Andelman, S., Branch, G., Bustamante, R.H., Castilla, J.C., Dugan, J., Halpern, B.S., Lafferty, K.D., Leslie, H., Lubchenco, J., McArdle, D., Possingham, H.P., Ruckelshaus, M. & Warner, R.R. (2003a) Ecological criteria for evaluating candidate sites for marine reserves. *Ecological Applications,* 13(1), S199-S214.

Roberts, C.M., Bohnsack, J.A., Gell, F., Hawkins, J.P. & Goodridge, R. (2001) Effects of marine reserves on adjacent fisheries. *Science,* 294(5548), 1920-1923.

Roberts, C.M., Branch, G., Bustamante, R.H., Castilla, J.C., Dugan, J., Halpern, B.S., Lafferty, K.D., Leslie, H., Lubchenco, J., McArdle, D., Ruckelshaus, M. & Warner, R.R. (2003b) Application of ecological criteria in selecting marine reserves and developing reserve networks. *Ecological Applications,* 13(1), S215-S228.

Roberts, C.M. & Hawkins, J.P. (2000). Fully-protected marine reserves: a guide. *WWF Endangered Seas Campaign, 1250 24th Street, NW, Washington, DC 20037, USA and Environment Department, University of York, York, YO10 5DD, UK*), p. 137 pp.

Roberts, C.M., Hawkins, J.P., Fletcher, J., Hands, S., Raab, K. & Ward, S. (2010). Guidance on the size and spacing of Marine Protected Areas in England. NECR037, Sheffield. Natural England. 87pp.).

Roberts, C.M., Hawkins, J.P. & Gell, F.R. (2005) The role of marine reserves in achieving sustainable fisheries. *Philosophical Transactions of the Royal Society B-Biological Sciences,* 360(1453), 123-132.

Roberts, T. & Jones, P.J.S. (2009) Shellfishing, Eider Ducks, and Nature Conservation on the Wash: Questions Raised by a Fractured Partnership. *Society & Natural Resources,* 22(6), 538-553.

Rodrigues, A.S.L. & Gaston, K.J. (2001) How large do reserve networks need to be? *Ecology Letters,* 4(6), 602-609.

Rondinini, C. (2011a) Meeting the MPA network design principles of representativity and adequacy: Developing species-area curves for habitats. JNCC Report 439. Joint Nature Conservation Committee, Peterborough.

Rondinini, C. (2011b) A review of methodologies that could be used to formulate ecologically meaningful targets for marine habitat coverage within the UK MPA network. JNCC Report No. 438. Joint Nature Conservation Committee, Peterborough.

Rondinini, C. & Chiozza, F. (2010) Quantitative methods for defining percentage area targets for habitat types in conservation planning. *Biological Conservation*, 143(7), 1646-1653.

Rondinini, C., Wilson, K.A., Boitani, L., Grantham, H. & Possingham, H.P. (2006) Tradeoffs of different types of species occurrence data for use in systematic conservation planning. *Ecology Letters*, 9(10), 1136-1145.

Rosenberg, A.A. & McLeod, K.L. (2005) Implementing ecosystem-based approaches to management for the conservation of ecosystem services. *Marine Ecology-Progress Series*, 300270-274.

Rouget, M., Cowling, R.M., Lombard, A.T., Knight, A.T. & Graham, I.H.K. (2006) Designing large-scale conservation corridors for pattern and process. *Conservation Biology*, 20(2), 549-561.

Rouget, M., Reyers, B., Jonas, Z., Desmet, P., Driver, A., Maze, K., Egoh, B., Cowling, R.M., Mucina, L. & Rutherford, M.C. (2004) South African National Spatial Biodiversity Assessment 2004: Technical Report. Volume 1: Terrestrial Component. Pretoria: South African National Biodiversity Institute.

Russ, G.R., Alcala, A.C., Maypa, A.P., Calumpong, H.P. & White, A.T. (2004) Marine reserve benefits local fisheries. *Ecological Applications*, 14(2), 597-606.

Russ, G.R., Cheal, A.J., Dolman, A.M., Emslie, M.J., Evans, R.D., Miller, I., Sweatman, H. & Williamson, D.H. (2008) Rapid increase in fish numbers follows creation of world's largest marine reserve network. *Current Biology*, 18(12), R514-R515.

– S –

Sanchirico, J.N. (2000) Marine Protected Area as Fishery Policy: A Discussion of Potential Costs and Benefits. Washington D.C., Resources for the Future. Discussion Paper, 00 - 23.

Sanchirico, J.N., Cochran, K.A. & Emerson, P.M. (2002) Marine Protected Areas: Economic and Social Implications. Washington D.C., Resources for the Future. Discussion Paper, 02 - 26.

Sanvicente-Anorve, L., Leprêtre, A. & Davoult, D. (2002) Diversity of benthic macrofauna in the eastern English Channel: comparison among and within communities. *Biodiversity and Conservation*, 11(2), 265-282.

Sarkar, S., Pressey, R.L., Faith, D.P., Margules, C.R., Fuller, T., Stoms, D.M., Moffett, A., Wilson, K.A., Williams, K.J., Williams, P.H. & Andelman, S. (2006) Biodiversity conservation planning tools: Present status and challenges for the future. *Annual Review of Environment and Resources*, 31123-159.

Scholz, A.J., Steinback, C., Kruse, S.A., Mertens, M. & Silverman, H. (2010) Incorporation of Spatial and Economic Analyses of Human-Use Data in the Design of Marine Protected Areas. *Conservation Biology*, 25(3), 485-492.

Shanks, A.L., Grantham, B.A. & Carr, M.H. (2003) Propagule dispersal distance and the size and spacing of marine reserves. *Ecological Applications*, 13(1), S159-S169.

Shears, N.T. & Babcock, R.C. (2003) Continuing trophic cascade effects after 25 years of no-take marine reserve protection. *Marine Ecology-Progress Series,* 2461-16.

Shears, N.T. & Ross, P.M. (2010) Toxic cascades: multiple anthropogenic stressors have complex and unanticipated interactive effects on temperate reefs. *Ecology Letters,* 13(9), 1149-1159.

Smith, A.B. (2010) Caution with curves: Caveats for using the species-area relationship in conservation. *Biological Conservation,* 143(3), 555-564.

Smith, M.D., Lynham, J., Sanchirico, J.N. & Wilson, J.A. (2010a) Political economy of marine reserves: Understanding the role of opportunity costs. *Proceedings of the National Academy of Sciences of the United States of America,* 107(43), 18300-18305.

Smith, R.J. (2004a). Conservation Land-Use Zoning (CLUZ) Software. http://www.mosaic-conservation.org/cluz. Durrell Institute of Conservation and Ecology, Canterbury, UK).

Smith, R.J. (2004b). Count Overlapping Polygons Software. http://www.mosaic-conservation.org/gis/overlap.html. Durrell Institute of Conservation and Ecology, Canterbury, UK).

Smith, R.J., Easton, J., Nhancale, B.A., Armstrong, A.J., Culverwell, J., Dlamini, S.D., Goodman, P.S., Loffler, L., Matthews, W.S., Monadjem, A., Mulqueeny, C.M., Ngwenya, P., Ntumi, C.P., Soto, B. & Leader-Williams, N. (2008) Designing a transfrontier conservation landscape for the Maputaland centre of endemism using biodiversity, economic and threat data. *Biological Conservation,* 141(8), 2127-2138.

Smith, R.J., Eastwood, P.D., Ota, Y. & Rogers, S.I. (2009) Developing best practice for using Marxan to locate Marine Protected Areas in European waters. *Ices Journal of Marine Science*, 66(1), 188-194.

Smith, R.J., Goodman, P.S. & Matthews, W.S. (2006) Systematic conservation planning: a review of percieved limitations and an illustration of the benefits, using a case study from Maputaland, South Africa. *Oryx,* 40(4), 400-410.

Smith, R.J., Minin, E., Linke, S., Segan, D. & Possingham, H.P. (2010b) An approach for ensuring minimum protected area size in systematic conservation planning. *Biological Conservation,* 143(11), 2525-2531.

Spalding, M.D., Fish, L. & Wood, L.J. (2008) Towards representative protection of the world's coasts and oceans – progress, gaps and opportunities. *Conservation Letters,* 1(5), 217 – 226.

Stanford, R.J. & Pitcher, T.J. (2004) Ecosystem simulations of the English Channel: climate and trade-offs. Fisheries Research Reports 12(3). The University of British Columbia, Vancouver.

Stefansson, G. & Rosenberg, A.A. (2005) Combining control measures for more effective management of fisheries under uncertainty: quotas, effort limitation and protected areas. *Philosophical Transactions of the Royal Society B-Biological Sciences,* 360(1453), 133-146.

Stelzenmueller, V., Maynou, F., Bernard, G., Cadiou, G., Camilleri, M., Crec'hriou, R., Criquet, G., Dimech, M., Esparza, O., Higgins, R., Lenfant, P. & Perez-Ruzafa, A. (2008) Spatial assessment of fishing effort around European marine reserves: Implications for successful fisheries management. *Marine Pollution Bulletin,* 56(12), 2018-2026.

Stewart, R.R., Noyce, T. & Possingham, H.P. (2003) Opportunity cost of ad hoc marine reserve design decisions: an example from South Australia. *Marine Ecology-Progress Series,* 25325-38.

Stewart, R.R. & Possingham, H.P. (2005) Efficiency, costs and trade-offs in marine reserve system design. *Environmental Modeling & Assessment*, 10(3), 203-213.

Stiles, A. & Scheiner, S.M. (2007) Evaluation of species-area functions using Sonoran Desert plant data: not all species-area curves are power functions. *Oikos*, 116(11), 1930-1940.

Stroud, D.A., Chambers, D., Cook, S., Buxton, N., Fraser, B., Clement, P., Lewis, P., McLean, I., Baker, H. & Whitehead, S. (2001). The UK SPA network: its scope and content. Volume 1: Rationale for the selection of sites. Joint Nature Conservation Committee. Peterborough.).

Sumaila, U.R., Guenette, S., Alder, J. & Chuenpagdee, R. (2000) Addressing ecosystem effects of fishing using marine protected areas. *ICES Journal of Marine Science,* 57(3), 752-760.

Sundblad, G., Bergström, U. & Sandström, A. (2011) Ecological coherence of marine protected area networks: a spatial assessment using species distribution models. *Journal of Applied Ecology,* 48(1), 112-120.

Svancara, L.K., Brannon, R., Scott, J.M., Groves, C.R., Noss, R.F. & Pressey, R.L. (2005) Policy-driven versus evidence-based conservation: A review of political targets and biological needs. *Bioscience,* 55(11), 989-995.

– T –

Tchoukanski, I. (2012) ET GeoWizards version 10.2. Available from:http://www.ian-ko.com/.

Tear, T.H., Kareiva, P., Angermeier, P.L., Comer, P., Czech, B., Kautz, R., Landon, L., Mehlman, D., Murphy, K., Ruckelshaus, M., Scott, J.M. & Wilhere, G. (2005) How much is enough? The recurrent problem of setting measurable objectives in conservation. *Bioscience,* 55(10), 835-849.

Tjorve, E. & Tjorve, K.M.C. (2008) The species-area relationship, self-similarity, and the true meaning of the z-value. *Ecology,* 89(12), 3528-3533.

Torpova, C., Meliane, I., Laffoley, D., Matthews, E. & Spalding, M.D. (2010) *Global ocean protection: present status and future possibilities*, Brest, France: Agence des aires marines protégées, Gland, Switzerland, Washington, DC and New York, USA: IUCN WCPA, Cambridge, UK : UNEP-WCMC, Arlington, USA: TNC, Tokyo, Japan: UNU, New York, USA: WCS. .

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 Management and Ecology,* 6(5), 401-420.

– U –

United Nations (2002). Plan of Implementation of the World Summit on Sustainable Development. Available from: http://www.un.org/esa/sustdev/documents/WSSD_POI_PD/English/WSSD_PlanImpl.pdf).

– V –

van Haastrecht, E.K. & Toonen, H.M. (2011) Science-Policy Interactions in MPA Site Selection in the Dutch Part of the North Sea. *Environmental Management,* 47(4), 656-670.

Vandeperre, F., Higgins, R.M., Sanchez-Meca, J., Maynou, F., Goni, R., Martin-Sosa, P., Perez-Ruzafa, A., Afonso, P., Bertocci, I., Crec'hriou, R., D'Anna, G., Dimech, M., Dorta, C., Esparza, O., Falcon, J.M., Forcada, A., Guala, I., Le Direach, L., Marcos, C., Ojeda-Martinez, C., Pipitone, C., Schembri, P.J., Stelzenmuller, V., Stobart, B. & Santos, R.S. (2011) Effects of no-take area size and age of marine protected areas on fisheries yields: a meta-analytical approach. *Fish and Fisheries*, 12(4), 412-426.

Varkey, D., Ainsworth, C.H. & Pitcher, T.J. (2012) Modelling Reef Fish Population Responses to Fisheries Restrictions in Marine Protected Areas in the Coral Triangle. *Journal of Marine Biology,* Article ID 72148318.

Vaz, S., Carpentier, A. & Coppin, F. (2007) Eastern English Channel fish assemblages: measuring the structuring effects of habitats on distinct sub-communities. *Ices Journal of Marine Science,* 64(2), 271-287.

Vaz, S., Martin, C.S., Eastwood, P.D., Ernande, B., Carpentier, A., Meaden, G.J. & Coppin, F. (2008) Modelling species distributions using regression quantiles. *Journal of Applied Ecology,* 45(1), 204-217.

Villanueva, M.C., Ernande, B. & Mackinson, S. (2009). Trophic Network. *Channel habitat atlas for marine resource management, final report (CHARM phase II). Interreg 3a Programme*. (ed. by A. Carpentier, C.S. Martin &S. Vaz). IFREMER, Boulogne-sur-mer, France.

Vimal, R., Rodrigues, A.S.L., Mathevet, R. & Thompson, J.D. (2011) The sensitivity of gap analysis to conservation targets. *Biodiversity and Conservation,* 20(3), 531-543.

Vincent, M.A., Atkins, S.M., Lumb, C.M., Golding, N., Lieberknecht, L.M. & Webster, M. (2004) Marine nature conservation and sustainable development - the Irish Sea Pilot. Joint Nature Conservation Commitee, Peterborough, UK.

– W –

Walters, C., Christensen, V. & Pauly, D. (1997) Structuring dynamic models of exploited ecosystems from trophic mass-balance assessments. *Reviews in Fish Biology and Fisheries,* 7(2), 139-172.

Walters, C., Pauly, D. & Christensen, V. (1999) Ecospace: Prediction of mesoscale spatial patterns in trophic relationships of exploited ecosystems, with emphasis on the impacts of marine protected areas. *Ecosystems,* 2(6), 539-554.

Walther, B.A. & Moore, J.L. (2005) The concepts of bias, precision and accuracy, and their use in testing the performance of species richness estimators, with a literature review of estimator performance. *Ecography,* 28(6), 815-829.

Warman, L.D., Sinclair, A.R.E., Scudder, G.G.E., Klinkenberg, B. & Pressey, R.L. (2004) Sensitivity of systematic reserve selection to decisions about scale, biological data, and targets: Case study from Southern British Columbia. *Conservation Biology,* 18(3), 655-666.

Watts, M.E., Ball, I.R., Stewart, R.S., Klein, C.J., Wilson, K., Steinback, C., Lourival, R., Kircher, L. & Possingham, H.P. (2009) Marxan with Zones: Software for optimal conservation based land- and sea-use zoning. *Environmental Modelling & Software,* 24(12), 1513-1521.

Watts, M.E., Klein, C.J., Stewart, R.R., Ball, I.R. & Possingham, H.P. (2008a) Marxan With Zones (V1.0.1): Conservation Zoning using Spatially Explicit Annealing.

Watts, M.E., Steinback, C. & Klein, C.J. (2008b) Applying Marxan with Zones to North central coast of California. User Guide.

West, P., Igoe, J. & Brockington, D. (2006) Parks and peoples: The social impact of protected areas. *Annual Review of Anthropology,* 35(1), 251-277.

Whittaker, R.J., Willis, K.J. & Field, R. (2001) Scale and species richness: towards a general, hierarchical theory of species diversity. *Journal of Biogeography,* 28(4), 453-470.

Wilson, K.A., Meijaard, E., Drummond, S., Grantham, H.S., Boitani, L., Catullo, G., Christie, L., Dennis, R., Dutton, I., Falcucci, A., Maiorano, L., Possingham, H.P., Rondinini, C., Turner, W.R., Venter, O. & Watts, M. (2010) Conserving biodiversity in production landscapes. *Ecological Applications*, 20(6), 1721-1732.

Witt, M.J. & Godley, B.J. (2007) A Step Towards Seascape Scale Conservation: Using Vessel Monitoring Systems (VMS) to Map Fishing Activity. *Plos One*, 2(10), e1111.

Wood, L. (2011) Global Marine Protection Targets: How S.M.A.R.T are They? *Environmental Management,* 47(4), 525-535.

Wood, L.J. (2007) MPA Global: A database of the world's marine protected areas. Sea Around Us Project, UNEP-WCMC & WWF. Available from: http://www.mpaglobal.org.

Wood, L.J., Fish, L., Laughren, J. & Pauly, D. (2008) Assessing progress towards global marine protection targets: shortfalls in information and action. *Oryx*, 42(3), 340-351.

Worm, B., Barbier, E.B., Beaumont, N., Duffy, J.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(5800), 787-790.

CHAPTER 8

APPENDIX

8.1 APPENDIX I - CHAPTER 3 SUPPLEMENTARY TABLES AND FIGURES

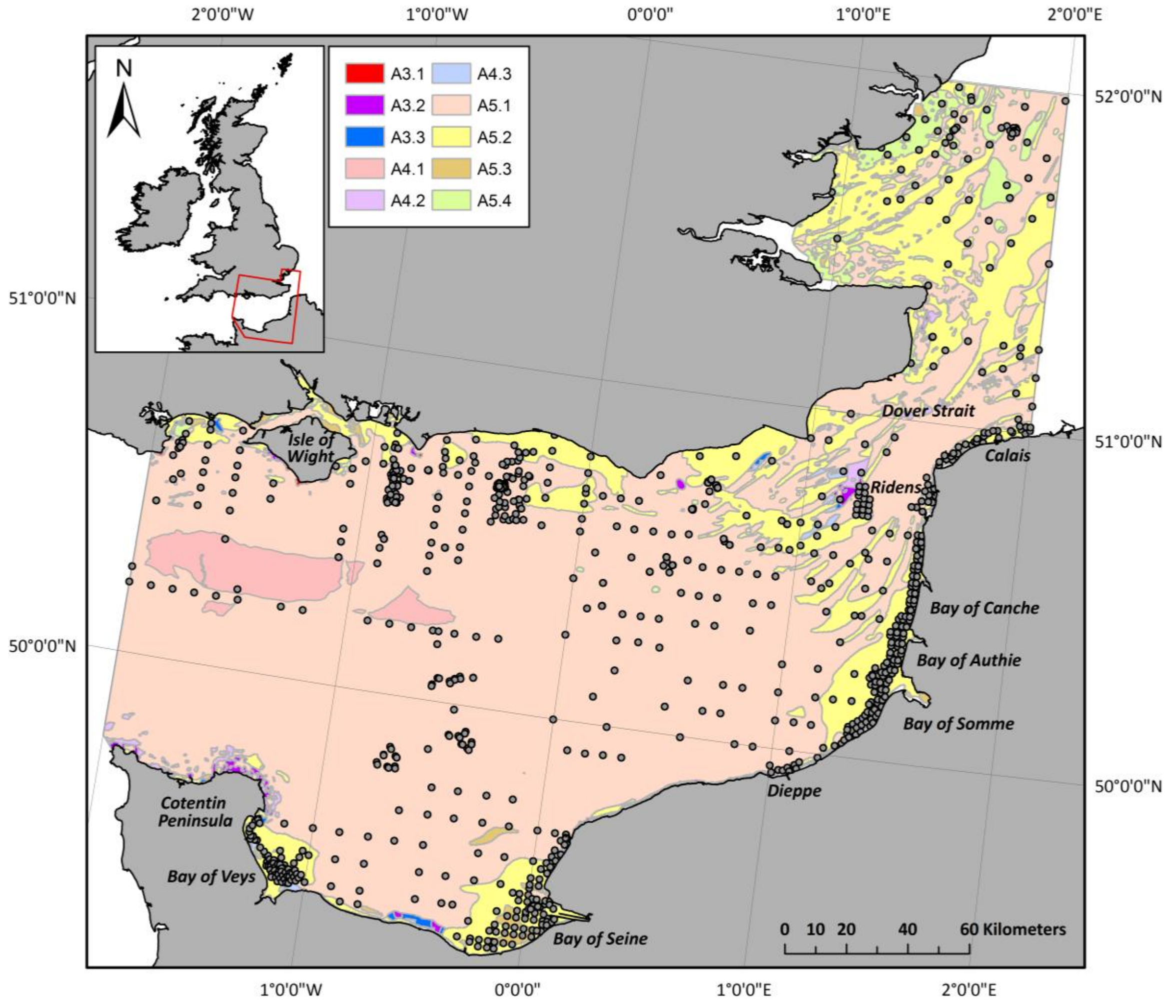

Figure S3.1 Broad-scale EUNIS level 3 habitat map for the eastern English Channel showing the location of the 1314 sampling points. Map projected in Europe Albers Equal Area Conic. See **Table S3.1** for a key to EUNIS habitat codes, levels and descriptions.

Table S3.1 Key to EUNIS codes, levels, and descriptions referred to in the text, figures and tables (EUNIS version 200611; EEA, 2006).

EUNIS Code	EUNIS Level	EUNIS Habitat / Biotope Description
A3.1	3	High-energy infralittoral rock
A3.2	3	Moderate energy infralittoral rock
A3.3	3	Low-energy infralittoral rock
A4.1	3	High-energy circalittoral rock
A4.2	3	Moderate energy circalittoral rock
A4.3	3	Low-energy circalittoral rock
A5.1	3	Sublittoral coarse sediment
A5.13	4	Infralittoral coarse sediment
A5.14	4	Circalittoral coarse sediment
A5.15	4	Deep circalittoral coarse sediment
A5.2	3	Sublittoral sand
A5.23	4	Infralittoral fine sand
A5.24	4	Infralittoral muddy sand
A5.25	4	Circalittoral fine sand
A5.26	4	Circalittoral muddy sand
A5.27	4	Deep circalittoral sand
A5.3	3	Sublittoral mud
A5.33	4	Infralittoral sandy mud
A5.34	4	Infralittoral fine mud
A5.35	4	Circalittoral sandy mud
A5.36	4	Circalittoral fine mud
A5.37	4	Deep circalittoral mud
A5.4	3	Sublittoral mixed sediments
A5.43	4	Infralittoral mixed sediments
A5.44	4	Circalittoral mixed sediments
A5.45	4	Deep mixed sediments

Table S3.2 Species richness estimates (values rounded to nearest whole number) for each of the broad-scale EUNIS level 3 and 4 habitats, calculated using the ICE, Chao2, Jackknife1, Jackknife2 and Bootstrap incidence-based estimators.

EUNIS Code	EUNIS Level	EUNIS Habitat Description	Area of habitat (km^2)	Number of samplin g points	Average area (m^2) of samples	Average number of species per sample	Total number of observed species	Non-parametric estimators [b] ICE	Chao2	Jackknife1	Jackknife2	Bootstrap	Mean Estimate	Estimate Range
A3.3	3	Low-energy infralittoral rock	116	11	0.5	10	60	118	96	91	107	74	97	44
A4.3	3	Low-energy circalittoral rock	108	5	0.5	38	142	333	288	223	268	178	258	155
A5.1 [†]	3	Subtidal coarse sediment	29889	725	0.26	53	1520	1798	1902	1846	2032	1665	1849	367
A5.13	4	Infralittoral coarse sediment	4092	263	0.2	46	971	1157	1136	1195	1267	1079	1167	116
A5.14	4	Circalittoral coarse sediment	18934	373	0.31	59	1326	1610	1654	1643	1806	1470	1637	336
A5.15	4	Deep circalittoral coarse sediment	6863	89	0.25	49	825	1099	1067	1094	1212	950	1084	262
A5.2 [a]	3	Subtidal Sand	7633	469	0.45	18	714	1001	994	958	1096	823	974	273
A5.23 or A5.24	4	Infralittoral fine sand or muddy sand	3701	288	0.45	18	590	841	798	799	903	684	805	219
A5.25 or A5.26	4	Circalittoral fine sand or muddy sand	3046	165	0.45	18	454	783	803	656	798	539	716	264
A5.27	4	Deep circalittoral sand	886	16	0.28	14	128	254	224	199	241	160	216	94
A5.3 [a]	3	Subtidal mud	335	28	0.48	21	198	376	361	296	362	240	327	136
A5.33 or A5.34	4	Infralittoral sandy mud or fine mud	196	17	0.49	18	139	261	261	212	261	170	233	91
A5.35 or A5.36	4	Circalittoral sandy mud or fine mud	134	11	0.46	26	131	230	254	195	237	158	215	96
A5.4 [a]	3	Subtidal mixed sediments	900	64	0.26	25	333	519	519	472	558	393	492	165
A5.44	4	Circalittoral mixed sediments	477	50	0.3	25	245	371	404	344	411	287	363	124
A5.45	4	Deep mixed sediments	198	14	0.11	25	164	339	302	251	307	202	280	137

[a] Species Richness estimates and corresponding *z*-values for these EUNIS level 3 habitats are obtained from their combined EUNIS level 4 habitat and survey data; A5.1 = (A5.13, A5.14, A5.15); A5.2 = (A5.23 or A5.24, A5.25 or A5.26, A5.27); A5.3 = (A5.33 or A5.34, A5.35 or A5.36); and A5.4 = (A5.44, A5.45). [b] Each estimate calculated in *EstimateS* represents the mean of 1000 estimates based on 1000 randomisations of sample accumulation order without replacement, with Chao2 computed using the classic formula (see Colwell 2009).

Table S3.3 Number of survey stations (sampling sites) required to reach stable estimates of species richness for each EUNIS habitat type based on five estimators of species richness. Habitats with an insufficient number of survey stations to reach stable estimates are denoted as follows '**-**'.

EUNIS Code	EUNIS Level	EUNIS Habitat Description	Number of sampling points	Number of sampling points required to reach a stable estimate of species richness				
				ICE	Chao2	Jackknife1	Jackknife2	Bootstrap
A3.3	3	Low-energy infralittoral rock	11	-	-	-	-	-
A4.3	3	Low-energy circalittoral rock	5	-	-	-	-	-
A5.1	3	Subtidal coarse sediment	725	93	56	39	86	65
A5.13	4	Infralittoral coarse sediment	263	86	72	50	78	67
A5.14	4	Circalittoral coarse sediment	373	71	50	33	66	53
A5.15	4	Deep circalittoral coarse sediment	89	61	50	61	58	52
A5.2	3	Subtidal Sand	469	293	409	366	291	276
A5.23 or A5.24	4	Infralittoral fine sand or muddy sand	288	211	271	-	214	208
A5.25 or A5.26	4	Circalittoral fine sand or muddy sand	165	146	-	-	143	133
A5.27	4	Deep circalittoral sand	16	-	-	-	15	15
A5.3	3	Subtidal mud	28	-	-	-	-	27
A5.33 or A5.34	4	Infralittoral sandy mud or fine mud	17	-	-	-	-	-
A5.35 or A5.36	4	Circalittoral sandy mud or fine mud	11	-	-	-	-	-
A5.4	3	Subtidal mixed sediments	64	54	-	-	51	44
A5.44	4	Circalittoral mixed sediments	50	45	-	-	42	38
A5.45	4	Deep mixed sediments	14	-	-	-	-	13

Table S3.4 Habitat specific *z*-values calculated using the ICE, Chao2, Jackknife1, Jackknife2 and Bootstrap incidence-based estimators.

EUNIS Code	EUNIS Level	EUNIS Habitat Description	Area of habitat (km^2)	Number of samplin g points	Average area (m^2) of samples	Average number of species per sample	Total number of observed species	z-values				
								ICE	Chao2	Jackknife1	Jackknife2	Bootstrap
A3.3	3	Low-energy infralittoral rock	116	11	0.5	10	60	0.128	0.117	0.115	0.123	0.104
A4.3	3	Low-energy circalittoral rock	108	5	0.5	38	142	0.113	0.106	0.092	0.102	0.080
A5.1	3	Sublittoral coarse sediment	29889	725	0.26	53	1520	0.138	0.141	0.139	0.143	0.135
A5.13	4	Infralittoral coarse sediment	4092	263	0.2	46	971	0.136	0.135	0.137	0.140	0.133
A5.14	4	Circalittoral coarse sediment	18934	373	0.31	59	1326	0.133	0.134	0.134	0.138	0.129
A5.15	4	Deep circalittoral coarse sediment	6863	89	0.25	49	825	0.129	0.128	0.129	0.133	0.123
A5.2	3	Sublittoral sand	7633	469	0.45	18	714	0.171	0.170	0.169	0.174	0.162
A5.23 or A5.24	4	Infralittoral fine sand or muddy sand	3701	288	0.45	18	590	0.168	0.166	0.166	0.171	0.159
A5.25 or A5.26	4	Circalittoral fine sand or muddy sand	3046	165	0.45	18	454	0.167	0.168	0.159	0.168	0.150
A5.27	4	Deep circalittoral sand	886	16	0.28	14	128	0.132	0.127	0.121	0.130	0.111
A5.3	3	Sublittoral mud	335	28	0.48	21	198	0.142	0.140	0.130	0.140	0.120
A5.33 or A5.34	4	Infralittoral sandy mud or fine mud	196	17	0.49	18	139	0.135	0.135	0.125	0.135	0.113
A5.35 or A5.36	4	Circalittoral sandy mud or fine mud	134	11	0.46	26	131	0.112	0.117	0.103	0.113	0.093
A5.4	3	Sublittoral mixed sediments	900	64	0.26	25	333	0.148	0.148	0.141	0.150	0.130
A5.44	4	Circalittoral mixed sediments	477	50	0.3	25	245	0.127	0.131	0.124	0.132	0.115
A5.45	4	Deep mixed sediments	198	14	0.11	25	164	0.122	0.117	0.108	0.118	0.098

8.2 APPENDIX II - CHAPTER 4 SUPPLEMENTARY TABLES AND FIGURES

Table S4.1 Details of the conservation features, targets, and the amount of each conserved in the current European Marine Sites.

Conservation features	Target (km^2 / Records)	Conserved (km^2 / Records)	Total (km^2 / Records)	Target met (%)
Habitat Conservation Feature Data				
High Energy Intertidal Rock (A1.1)	12.029	18.135	31.739	150.76
Moderate Energy Intertidal Rock (A1.2)	20.048	28.330	52.758	141.31
Low Energy Intertidal Rock (A1.3)	7.760	15.789	19.896	203.47
Features of Littoral Rock (A1.4) [a]	0.002	0.007	0.007	261.10
Intertidal Coarse Sediments (A2.1)	17.315	24.206	41.423	139.80
Intertidal Sand & Muddy Sand (A2.2)	330.234	698.418	790.032	211.49
Intertidal Mud (A2.3)	427.674	959.628	1023.143	224.38
Intertidal Mixed Sediments (A2.4)	22.235	30.650	53.409	137.29
Coastal Saltmarshes and Saline Reedbeds (A2.5) [a]	37.539	88.639	89.806	236.12
Intertidal Mud & Coastal Saltmarshes and Saline Reedbeds Mosaic (A2.3 and A2.5) [a]	18.863	45.127	45.127	239.23
Intertidal Sediments Dominated by Aquatic Angiosperms (A2.6) [a]	4.767	11.291	11.403	236.87
Intertidal Biogenic Reefs (A2.7) [a]	8.905	11.656	21.305	130.89
Features of Littoral Sediment (A2.8) [a]	0.033	0.079	0.079	239.23
High Energy Infralittoral Rock (A3.1)	657.922	506.307	2157.120	76.96
Moderate Energy Infralittoral Rock (A3.2)	405.811	200.526	1252.503	49.41
Low Energy Infralittoral Rock (A3.3)	34.432	71.016	108.961	206.25

High Energy Circalittoral Rock (A4.1)	1168.165	64.625	4599.076	5.53
Moderate Energy Circalittoral Rock (A4.2)	7502.758	982.807	26891.065	13.10
Low Energy Circalittoral Rock (A4.3)	21.405	9.920	67.953	46.35
Sublittoral Coarse Sediment (A5.1)	21106.624	4556.058	64896.986	21.67
Sublittoral Sand (A5.2)	36651.566	7290.449	122580.488	19.89
Sublittoral Mud (A5.3)	3242.835	435.687	10881.996	13.44
Sublittoral Mixed Sediments (A5.4)	1734.316	820.690	5436.728	47.32
Sublittoral Macrophyte Dominated Sediments (A5.5) [a]	2.981	9.498	9.617	318.59
Sublittoral Biogenic Reefs (A5.6) [a]	1.187	3.829	3.829	322.58
Deep-sea rock & artificial hard substrata (A6.1) [a*]	11.389	0.000	27.246	0.00
Deep-sea mixed substrata (A6.2) [a*]	149.291	0.000	357.156	0.00
Deep-sea sand or Deep-sea mud (A6.3 or A6.4) [a*]	110.331	0.000	263.951	0.00
Deep-sea mud (A6.5) [a*]	243.156	0.000	581.712	0.00
Deep Sea Coarse Sediment [a*]	193.628	0.000	463.225	0.00
Species Conservation Feature Data				
BS Ocean Quahog (*Arctica islandica*) [b †]	3	2	3	66.67
BS Lagoon Sandworm *(Armandia cirrhosa)* [b † ‡]	0	0	1	0.00
BS Defolin's Lagoon Snail *(Caecum armoricum)* [b †]	1	0	1	0.00
BS Pink Sea-Fan (*Eunicella verrucosa*) [b †]	1	0	1	0.00
BS Lagoon Sand Shrimp *(Gammarus insensibilis)* [b † ‡]	0	0	1	0.00
BS Stalked Jellyfish (*Haliclystus auricula*) [b †]	2	2	2	100.00
BS Long Snouted Seahorse *(Hippocampus guttulatus)* [b †]	3	0	5	0.00
BS Short Snouted Seahorse (*Hippocampus hippocampus*) [b †]	3	1	4	33.33

BS Stalked Jellyfish (*Lucernariopsis campanulata*) [b †]	1	0	1	0.00
BS Starlet Sea Anemone (*Nematostella vectensis*) [b †]	3	2	5	66.67
BS Native Oyster (*Ostrea edulis*) [b †]	3	11	75	366.67
BS Sea Snail (*Paludinella littorina*) [b †]	3	3	3	100.00
BS Common Maerl (*Phymatolithon calcareum*) [b †]	1	0	1	0.00
BS Lagoon sea slug *(Tenellia adspersa)* [b † ‡]	2	2	3	100.00
FS Tentacled Lagoon Worm *(Alkmaria romijni)* [b †]	3	3	4	100.00
FS Sea-Fan Anemone (*Amphianthus dohrnii*) [b †]	3	0	12	0.00
FS Ocean Quahog (*Arctica islandica*) [b †]	3	0	16	0.00
FS Lagoon Sandworm *(Armandia cirrhosa)* [b †]	1	0	1	0.00
FS Fan Mussel *(Atrina pectinata)* [b †]	3	1	4	33.33
FS Defolin's Lagoon Snail *(Caecum armoricum)* [b †]	1	0	1	0.00
FS Burgundy maerl Paint Weed *(Cruoria cruoriaeformis)* [b †]	2	1	2	50.00
FS Pink Sea-Fan (*Eunicella verrucosa*) [b †]	3	9	89	300.00
FS Lagoon Sand Shrimp *(Gammarus insensibilis)* [b †]	3	0	4	0.00
FS Giant Goby *(Gobius cobitis)* [b † ‡]	0	0	1	0.00
FS Couch's Goby (*Gobius couchi*) [b †]	3	1	4	33.33
FS Stalked Jellyfish (*Haliclystus auricula*) [b †]	3	2	10	66.67
FS Long Snouted Seahorse *(Hippocampus guttulatus)* [b † ‡]	0	0	1	0.00
FS Short Snouted Seahorse (*Hippocampus hippocampus*) [b † ‡]	1	0	2	0.00
FS Sunset Cup Coral (*Leptopsammia pruvoti*) [b †]	3	0	7	0.00
FS Coral Maerl (*Lithothamnion corallioides*) [b † ‡]	2	2	3	100.00
FS Stalked Jellyfish (*Lucernariopsis campanulata*) [b † ‡]	2	2	5	100.00
FS Stalked Jellyfish (*Lucernariopsis cruxmelitensis*) [b †]	1	0	1	0.00

FS Starlet Sea Anemone (*Nematostella vectensis*) [b † ‡]	2	1	4	50.00
FS Native Oyster (*Ostrea edulis*) [b †]	3	9	40	300.00
FS Peacock's Tail (*Padina pavonica*) [b †]	3	3	5	100.00
FS Spiny Lobster (*Palinurus elephas*) [b †]	3	4	18	133.33
FS Sea Snail (*Paludinella littorina*) [b †]	3	1	13	33.33
FS Common Maerl (*Phymatolithon calcareum*) [b †]	3	4	13	133.33
FS Lagoon sea slug *(Tenellia adspersa)* [b † ‡]	1	0	2	0.00
ISCZ Ocean Quahog (*Arctica islandica*) [b †]	3	8	15	266.67
ISCZ Native Oyster (*Ostrea edulis*) [b †]	3	10	11	333.33
NG Tentacled Lagoon Worm *(Alkmaria romijni)* [b †]	3	3	3	100.00
NG Ocean Quahog (*Arctica islandica*) [b †]	3	3	28	100.00
NG Amphipod Shrimp *(Gitanopsis bispanosa)* [b †]	1	0	1	0.00
NG Starlet Sea Anemone (*Nematostella vectensis*) [b †]	1	1	1	100.00
NG Native Oyster (*Ostrea edulis*) [b †]	2	0	2	0.00

[a] Targets for these broad-scale EUNIS habitats were set based on the mean proportion of habitat area required to represent 80% of species for their EUNIS habitat class, with the exception of deep-sea bed habitats ([a*]) which were based on the highest proportion of habitat area to represent 80% of species in the MCZ project area (41.8%). [b] Targets for species features of conservation interest (FOCI) were based on the MCZ ecological network guidance and developed to ensure that each of the MCZ project areas contained at least a minimum of three replicates of each species. However, for species which had < 3 records the target was set to equal the total number of records, with the exception of species FOCI records which were located in excluded planning units ([‡]*)*. The location of species records within each MCZ project area is indicated by the following prefix: BS – Balanced Seas; FS – Finding Sanctuary; ISCZ – Irish Sea Conservation Zones; and NG – Net Gain. [†] Refers to conservation features that are based on point location data.

8.3 APPENDIX III - CHAPTER 5 SUPPLEMENTARY TABLES AND FIGURES

Table S5.1 Details of the conservation features and targets used in the Marxan analyses.

Id	Conservation feature data	Total (km^2 / records)	Target (km^2 / records)
Habitat conservation feature data			
1	UK High energy infralittoral rock (A3.1) [a]	11.79	3.60
2	UK Moderate energy infralittoral rock (A3.2) [a]	77.15	25.00 ±
3	UK Low energy infralittoral rock (A3.3) [a]	31.61	9.99 ±
4	UK High energy circalittoral rock (A4.1) [a]	1144.54	290.71
5	UK Moderate energy circalittoral rock (A4.2) [a]	80.36	22.42
6	UK Low energy circalittoral rock (A4.3) [a]	72.06	22.70 ±
7	UK Infralittoral coarse sediment (A5.13) [a *]	1161.68	376.38
8	UK Circalittoral coarse sediment (A5.14) [a *]	7186.01	2328.27
9	UK Deep circalittoral coarse sediment (A5.15) [a *]	2623.98	850.17
10	UK Infralittoral fine sand or muddy sand (A5.23 or A5.24) [a *]	1080.58	323.09
11	UK Circalittoral fine sand or muddy sand (A5.25 or A5.26) [a *]	1066.68	318.94
12	UK Deep circalittoral sand (A5.27) [a *]	218.50	65.33 ±
13	UK Infralittoral sandy mud or fine mud (A5.33 or A5.34) [a *]	15.99	4.76
14	UK Circalittoral sandy mud or fine mud (A5.35 or A5.36) [a *]	8.89	2.65 ±
15	UK Infralittoral mixed sediments (A5.43) [a *]	41.96	13.39
16	UK Circalittoral mixed sediments (A5.44) [a *]	28.71	9.16 ±
18	FR High energy infralittoral rock (A3.1) [a]	1.80	0.55
19	FR Moderate energy infralittoral rock (A3.2) [a]	48.82	15.82
20	FR Low energy infralittoral rock (A3.3) [a]	63.29	20.00 ±
21	FR High energy circalittoral rock (A4.1) [a]	1.92	0.49
22	FR Moderate energy circalittoral rock (A4.2) [a]	59.18	16.51
23	FR Low energy circalittoral rock (A4.3) [a]	32.98	10.39 ±
24	FR Infralittoral coarse sediment (A5.13) [a *]	2315.17	750.11
25	FR Circalittoral coarse sediment (A5.14) [a *]	10862.84	3519.56
26	FR Deep circalittoral coarse sediment (A5.15) [a *]	3199.75	1036.72
27	FR Infralittoral fine sand or muddy sand (A5.23 or A5.24) [a *]	1504.00	449.70
28	FR Circalittoral fine sand or muddy sand (A5.25 or A5.26) [a *]	787.61	235.50 ±
29	FR Deep circalittoral sand (A5.27) [a *]	66.93	20.82 ±
30	FR Infralittoral sandy mud or fine mud (A5.33 or A5.34) [a *]	99.06	29.52 ±
31	FR Circalittoral sandy mud or fine mud (A5.35 or A5.36) [a *]	85.29	25.42 ±
32	FR Deep circalittoral mud (A5.37) [a *]	0.54	0.16 ±
33	FR Circalittoral mixed sediments (A5.44) [a *]	2.15	0.68 ±
34	UK Intertidal mussel beds (*Mytilus edulis*) [a, b †]	1	1
35	UK Intertidal mudflats [a, b †]	4	3
36	UK Littoral chalk communities [a, b †]	11	3
37	UK Maerl beds [a, b †]	3	3
38	UK Honeycomb worm (*Sabellaria alveolata*) reefs [a, b †]	7	3

39	UK Ross worm (*Sabellaria spinulosa*) reefs [a, b †]	54	3
40	UK Seagrass beds (*Zostera marina*) [a, b †]	3	3
41	FR Intertidal mudflats [a, b †]	21	3
42	FR Littoral chalk communities [a, b †]	17	3
43	FR Maerl beds [a, b †]	3	3
44	FR Ross worm (*Sabellaria spinulosa*) reefs [a, b †]	5	3
45	FR Seagrass beds (*Zostera marina*) [a, b †]	2	2

Species Conservation Feature Data

46	UK Common maerl (Phymatolithon calcareum) [c, d, e †]	4	3
47	UK Dog whelk (*Nucella lapillus*) [b †]	7	3
48	UK Fan mussel (*Atrina fragilis*) [e †]	1	1
49	UK Horse mussel (*Modiolus modiolus*) [a, b †]	12	3
50	UK Native/flat oyster (*Ostrea edulis*) [b, c, e †]	42	3
51	UK Ocean quahog (*Arctica islandica*) [b, c †]	6	3
52	UK Peacocks tail (*Padina pavonica*) [c, e †]	4	3
53	UK Short-snouted seahorse (*Hippocampus hippocampus*) [c, e †]	2	2
54	UK Stalked jellyfish (*Haliclystus auricula*) [c, e †]	2	1
55	UK Stalked jellyfish (*Lucernariopsis campanulata*) [c †]	1	1
56	UK Starlet sea anemone (*Nematostella vectensis*) [c, e †]	6	3
57	FR Dog whelk (*Nucella lapillus*) [b]	20	3
58	FR Horse mussel (*Modiolus modiolus*) [a, b †]	7	3
59	FR Long-snouted seahorse (*Hippocampus guttulatus*) [c, e †]	1	1
60	FR Native/flat oyster (*Ostrea edulis*) [b, c, e †]	8	3
61	FR Ocean quahog (*Arctica islandica*) [b, c †]	1	1
62	FR Short-snouted seahorse (*Hippocampus hippocampus*) [c, e †]	3	3
63	FR Stalked jellyfish (*Haliclystus auricula*) [c, e †]	1	1
64	FR Starlet sea anemone (*Nematostella vectensis*) [c, e †]	1	1
65	UK Arctic tern (*Sterna paradisaea*) [d †]	2	1
66	UK Black-headed gull (*Larus ribibundus*) [d †]	19	3
67	UK Black legged kittiwake (*Rissa tridactyla*) [b †]	115	3
68	UK Common guillemot (*Uria aalge*) [b, d †]	72	3
69	UK Common tern (*Sterna hirundo*) [d †]	4	3
70	UK European herring gull (Larus argentatus) [d †]	114	3
71	UK European shag (*Phalacrocorax aristotelis*) [d†]	1	1
72	UK Great Black-backed gull (*Larus marinus*) [d †]	130	3
73	UK Lesser black-backed gull (*Larus fuscus*) [b, d †]	126	3
74	UK Little gull (*Larus minutus*) [d †]	2	1
75	UK Mew gull (*Larus canus*) [d †]	55	3
76	UK Sandwich tern (Sterna sandvicensis) [d †]	5	3
77	FR Black-headed gull (*Larus ribibundus*) [d †]	12	3
78	FR Black legged kittiwake (*Rissa tridactyla*) [b †]	111	3
79	FR Common guillemot (*Uria aalge*) [b, d †]	55	3
80	FR Common tern (*Sterna hirundo*) [d †]	1	1
81	FR European herring gull (Larus argentatus) [d †]	161	3
82	FR European shag (*Phalacrocorax aristotelis*) [d†]	3	3
83	FR European storm petrel (*Hydrobates pelagicus*) [d †]	1	1
84	FR Great Black-backed gull (*Larus marinus*) [d †]	60	3

85	FR Lesser black-backed gull (*Larus fuscus*) [b, d †]	54	3
86	FR Little gull (*Larus minutus*) [d †]	20	3
87	FR Mediterranean gull (*Larus melanocephalus*) [d †]	6	3
88	FR Mew gull (*Larus canus*) [d †]	23	3
89	FR Sandwich tern (Sterna sandvicensis) [d †]	2	2
90	UK Atlantic cod (*Gadus morhua*) [b, e ‡]	211.41	21.14
91	UK Atlantic mackerel (*Scomber scombrus*) [e ‡]	185.63	18.56
92	UK Common sole (*Solea solea*) [e ‡]	100.42	10.04
93	UK Horse mackerel (*Trachurus trachurus*) [e ‡]	252.11	25.21
94	UK Plaice (*Pleuronectes platessa*) [e ‡]	277.99	27.80
95	UK Spotted ray (*Raja montagui*) [b ‡]	139.09	13.91
96	UK Thornback ray (*Raja clavata*) [b ‡]	315.87	31.59
97	UK Whiting (*Merlangius merlangus*) [e ‡]	148.83	14.88
98	FR Atlantic cod (*Gadus morhua*) [b, e ‡]	321.96	32.20
99	FR Atlantic mackerel (*Scomber scombrus*) [e ‡]	312.68	31.27
100	FR Common sole (*Solea solea*) [e ‡]	174.82	17.48
101	FR Horse mackerel (*Trachurus trachurus*) [e ‡]	252.11	25.21
102	FR Plaice (*Pleuronectes platessa*) [e ‡]	277.99	27.80
103	FR Spotted ray (*Raja montagui*) [b ‡]	190.56	19.06
104	FR Thornback ray (*Raja clavata*) [b ‡]	475.69	47.57
105	FR Whiting (*Merlangius merlangus*) [e ‡]	192.02	19.20

[a] Broad-scale habitats and habitat features of conservation interest (FOCI) to be protected within MPAs as identified by the MCZ ecological network guidance (JNCC & Natural England 2010), [b] Threatened and/or declining species and habitats listed by OSPAR (OSPAR 2008c), [c] Low or limited mobility species FOCI to be protected within MPAs as identified by the MCZ ecological network guidance (JNCC & Natural England 2010), [d] Species listed by EU Birds and Habitats Directives (EC 1979; EC 1992), and [e] Marine species listed by French and or UK national legislation (e.g. UK BAP). [*] Refers to habitat targets based on their parent EUNIS level 3 habitats: A5.1x = 0.324; A5.2x = 0.299; A5.3x = 0.298; A5.4x = 0.319; (JNCC & Natural England 2010). [†] Refers to conservation features that are based on point location data. [‡] Refers to conservation features that are based on species distribution model data, where the total refers to sum of the mean habitat suitability values across all planning units, and thus has no units. [±] Refers to conservation features which have a restricted distribution in the Marxan solutions and therefore did not have specified zone targets in Marxan with Zones.

Table S5.2 Ecopath input data and composition of the species included in each of the 51 functional groups. Those values estimated by Ecopath (outputs) are given in *italic*. TL: trophic level; B: biomass (t/km^2); P/B: production to biomass ratio; Q/B: consumption to biomass ratio; EE: ecotrophic efficiency; P/Q: consumption to biomass ratio; and C: catches (t/km^2).

Functional groups	**Species**	***TL***	**B (t/km^2)**	**P/B (year)**	**Q/B (year)**	***EE***	**P/Q (year)**	**C (t/km^2)**
Phytoplankton		*1.00*	20.0000	40.0000	-	*0.5150*	-	0.0000
Phytobenthos		*1.00*	64.1200	60.0000	-	*0.5439*	-	0.0000
Scallops	King scallop *(Pecten maximus), Queen scallop (Aequipecten opercularis),* Variegated scallop *(Chlamys varia)*	*2.00*	1.7270	0.9000	10.0000	*0.8014*	0.0900	0.4100
Suspension feeders	White furrow shell *(Abra alba), Mussel (Mytilus edulis),* Common cockle *(Cerastoderma edule),* Pectinids *(Chlamys varia and Aequipecten opercularis),* Banded carpet shell *(Paphia rhomboids)* and Clams *(Donax sp, Mercenaria merenaria, Ruditapes philippinarum)*	*2.00*	22.4800	3.0000	20.0000	*0.1961*	0.1500	0.3980
Deposit feeders	Worms, gastropods and small invertebrates	*2.00*	20.0000	2.5000	16.6667	*0.6625*	0.1500	0.0000
Herbivorous zooplankton	Copepods, cladocerans and tunicates	*2.00*	27.2219	35.0000	60.0000	*0.9000*	0.5833	0.0000
Oysters	European flat oyster *(Ostrea edulis)*	*2.01*	2.1000	0.9040	35.0000	*0.9945*	0.0258	1.0560
Benthic omnivores		*2.14*	134.2664	0.9000	6.0000	*0.9000*	0.1500	0.0000
Whelk	Common whelk (*Buccinum undatum*)	*2.24*	0.2200	1.4000	9.3333	*0.7934*	0.1500	0.0210
Crabs	Common shore crab (Carcinus maenas), Hermit crab (*Pagurus bernhardus*), Velvet swimming crab *(Necora puber)*	*2.34*	10.8000	1.0500	3.0670	*0.7878*	0.3424	0.0000
Commercial crabs	Edible crab (*Cancer pagurus*), Spider crab *(Maja squinado)*	*2.35*	4.2720	1.0100	7.0000	*0.9000*	0.1443	0.1040

Small demersal fish	Pogge *(Agonus cataphractus)*, Common dragonet (*Callionymus lyra*)	*2.52*	6.7045	1.3190	10.3840	*0.8000*	0.1270	0.0040
Shrimps and prawns	Brown shrimp *(Crangon crangon)*	*2.62*	12.2554	1.7000	38.4600	*0.9000*	0.0442	0.0040
Carnivorous zooplankton	Fish larvae, chaetognaths and ctenophores	*2.71*	14.7800	18.0000	23.3300	*0.9000*	0.7715	0.0000
Other flatfish		*2.84*	0.2000	1.9900	5.4640	*0.9257*	0.3642	0.0500
Mullet	Grey thick-lipped mullet (*Chelon labrosus)*	*2.85*	2.5000	0.5000	5.1670	*0.6729*	0.0968	0.0930
Carnivorous megabenthos	Starfish *(Porania (Porania) pulvillus)*, Holothurians *(Thyone fusus)*, Featherstar *(Leptometra celtica)* and Brittlestars *(Ophiothrix fragilis)*	*2.90*	120.0000	0.6000	6.9350	*0.5977*	0.0865	0.0000
Lobster	European Lobster *(Homarus gammarus)*, Spiny lobster *(Palinrus elephas)*	*2.93*	0.0130	1.0800	5.8500	*0.3633*	0.1846	0.0050
Dab	*Limanda limanda*	*2.97*	0.6000	0.4000	6.4080	*0.8637*	0.0624	0.0640
Pilchard	*Sardina pilchardus*	*3.17*	0.4420	0.9880	7.2040	*0.5039*	0.1372	0.0610
Adult sole	*Solea solea*	*3.17*	0.3370	0.6500	5.0630	*0.9253*	0.1284	0.1960
Small Gadoids	Pouting *(Trisopterus luscus)*, Poor cod *(Trisopterus minutus)*	*3.20*	3.5000	1.2430	5.1670	*0.9629*	0.2406	0.0940
Juvenile seabass	*Dicentrarchus labrax*	*3.24*	0.0320	1.2450	6.3480	*0.1167*	0.1961	0.0000
Adult plaice	*Pleuronectes platessa*	*3.26*	0.3500	0.6000	4.3350	*0.9881*	0.1384	0.2030
Sandeels	Lesser sand eel *(Ammodytes tobianus)*	*3.28*	2.4294	1.7400	9.1600	*0.4000*	0.1900	0.0000
Seabream	*Spondyliosoma cantharus*	*3.30*	0.1000	1.7420	11.3530	*0.1604*	0.1534	0.0200
Gurnards	Atlantic red gurnard *(Aspitriglia cuculus)*, Tub gurnard *(Chelidonichthys lucernus)* and Grey gurnard *(Eutrigla gurnardus)*	*3.30*	0.4100	0.7300	4.7540	*0.4775*	0.1536	0.0810
Scad	*Trachurus trachurus*	*3.32*	0.2300	0.6500	6.0250	*0.6002*	0.0581	0.0650

Juvenile plaice	*Pleuronectes platessa*	*3.32*	0.1500	1.3000	8.2180	*0.3625*	0.1582	0.0000
Cuttlefish	*Sepia officinalis*	*3.37*	0.3100	3.5000	15.0000	*0.5603*	0.2333	0.0110
Juvenile sole	*Solea solea*	*3.40*	0.0600	1.3000	10.1260	*0.5480*	0.1284	0.0000
Adult seabass	*Dicentrarchus labrax*	*3.42*	0.0680	0.5400	3.2880	*0.9811*	0.1642	0.0360
Sprat	*Sprattus sprattus*	*3.53*	0.7620	1.8520	10.1630	*0.2782*	0.1822	0.0030
Juvenile cod	*Gadus morhua*	*3.54*	0.1030	2.2680	6.0640	*0.2353*	0.3740	0.0000
Mackerel	*Scomber scombrus*	*3.55*	0.9520	0.6850	6.0470	*0.8877*	0.1133	0.2150
Rays and dogfish	Thornback ray *(Raja clavata),* Painted ray *(Raja microcellata),* Cuckoo ray *(Leucoraja naevus),* Spotted ray *(Raja montagui),* Starry ray *(Raja radiate),* Undulate ray *(Raja undulate),* Nurse hound *(Scyliorhinus stellaris),* Lesser spotted dogfish *(Scyliorhinus canicula),* Portugese dogfish *(Centroscymnus coelolepsis)* and Spurdog *(Squalus acanthias)*	*3.58*	0.1503	0.4400	2.5710	*0.6049*	0.1711	0.0400
Adult cod	*Gadus morhua*	*3.58*	0.2200	1.2170	3.0310	*0.8444*	0.4015	0.1320
Pollack	*Pollachius pollachius*	*3.60*	0.1090	0.9200	3.3200	*0.9989*	0.2848	0.0300
Herring	*Clupea harengus*	*3.62*	4.7960	1.0400	4.6000	*0.4171*	0.2261	0.8570
John Dory	*Zeus faber*	*3.62*	0.0300	0.8620	4.6870	*0.1194*	0.1839	0.0020
Squid	Veined squid *(Loligo forbesi)* and European squid *(Loligo vulgaris)*	*3.64*	0.4800	3.1500	15.0000	*0.4600*	0.2100	0.2300
Juvenile whiting	*Merlangius merlangus*	*3.64*	0.1150	2.1360	10.9540	*0.1634*	0.1950	0.0000
Hake	*Merluccius merluccius*	*3.83*	0.0970	0.6600	3.6150	*0.8886*	0.1826	0.0020
Adult whiting	*Merlangius merlangus*	*3.88*	0.7000	1.8000	4.7090	*0.9202*	0.3823	0.3660

Large demersal bottom fish	*Ling (Molva molva), European conger eel (Conger conger), Greater weaver (Trachinus draco), Greater porkbeard (Phycis blennoides), European eel (Anguilla anguilla), White anglerfish (Lophius budegassa), Black scabbarfish (Aphanopus carbo), Garfish (Belone belone), Atlantic pomfret (Brama brama), Patagonian toothfish (Dissostichus eleginoides), Orange roughy (Hoplostehus atlanticus), Spotted scorpionfish (Scorpaena plumier) and Anglerfish (Lophius piscatorius)*	*3.89*	1.5970	0.5780	4.3850	*0.3429*	0.1318	0.2740
Seabirds	Petrels, pelecaniformes, eiders, gulls, terns and auks	*3.92*	0.0018	0.4000	66.6410	*0.0000*	0.0060	0.0000
Toothed cetaceans	Common dolphin *(Delphinus delphis),* Harbour porpoise *(Phocoena phocoena),* Bottlenose dolphin *(Tursiops truncates)* and Minke whale *(Balaenoptera acutorostrata)*	*4.24*	0.0159	0.0980	16.8910	*0.0000*	0.0058	0.0000
Seals	Grey *(Halichoerus grypus)* and Harbour seal *(Phoca vitulina)*	*4.38*	0.0002	0.4700	15.7520	*0.0000*	0.0298	0.0000
Sharks	Tope *(Galeorhinus galeus),* Starry smooth-hound *(Mustelus asterias)* and Smooth-hound *(Mustelus mustelus)*	*4.50*	0.3070	0.1900	2.3700	*0.8915*	0.0802	0.0520
Discards		*1.00*	2.3940	-	-	*0.5511*	-	0.0000
Detritus		*1.00*	25.0000	-	-	*0.3936*	-	0.0000

Table S5.3 Economic data (relative profitability) for the English Channel fleets based on data reported in Villanueva *et al.*, (2009) and Daskalov *et al.*, (2011).

Fleet	**Fixed cost (%)**	**Sailing cost (%)**	**Profit (%)**
Beam trawl	35.30	58.70	6.00
Demersal otter trawl	43.10	44.00	12.90
Dredge	31.70	52.40	15.90
Pelagic trawl	39.90	40.00	20.10
Hooks and lines	21.00	11.80	67.20
Nets	48.00	37.70	14.30
Traps and pots	31.90	46.20	21.90
Seine [a]	39.90	40.00	20.10
Other [b]	31.90	46.20	21.90

[a] Cost data for seine fleet based on costs used for pelagic trawl.

[b] Includes both active and passive fishing gears.

Table S5.4 Distribution of the functional groups as assigned to habitat types in the eastern English Channel.

Functional groups	All habitats	Offshore[a]	Intermediate[b]	Coastal homogenous[c]	Coastal heterogeneous[d]
Phytoplankton	+				
Phytobenthos	+				
Scallops		+	+	+	
Suspension feeders		+	+	+	
Deposit feeders			+	+	+
Herbivorous zooplankton	+				
Oysters				+	+
Benthic omnivores		+	+	+	
Whelk			+	+	
Crabs		+			+
Commercial crabs			+	+	+
Small demersal fish			+	+	+
Shrimps and prawns			+	+	+
Carnivorous zooplankton	+				
Other flatfish			+	+	+
Mullet			+		
Carnivorous megabenthos		+	+	+	
Lobster				+	+
Dab			+	+	+
Pilchard			+		
Adult sole			+	+	+
Small Gadoids	+				
Juvenile seabass				+	+
Adult plaice			+	+	+
Sandeels				+	
Seabream			+		
Gurnards		+	+		
Scad			+	+	
Juvenile plaice				+	+
Cuttlefish			+	+	
Juvenile sole				+	+
Adult seabass			+	+	+
Sprat					+
Juvenile cod			+		
Mackerel			+	+	
Rays and dogfish		+		+	+
Adult cod			+		
Pollack			+		
Herring					+
John Dory			+		
Squid		+	+	+	
Juvenile whiting				+	+
Hake			+		

Adult whiting			+	+	+
Large demersal bottom fish		+	+		
Seabirds	+				
Toothed cetaceans		+	+		
Seals			+	+	+
Sharks		+			
Discards	+				
Detritus	+				

[a] Offshore community **-** characterised by: hard sediment types, high salinity and warm temperatures, strong tidal currents and greater depths; and associated with: Elasmobranches (sharks, skates and rays), and poor cod. Relatively lower diversity than in coastal areas. [b] Intermediate community – characterised by: coarse sand sediment types; and associated with: Pelagic (sardine, mackerel) and demersal (dragonets, gurnards, red mullet) species, comparable diversity to offshore community. [c] Coastal homogenous community – characterised by: fine sand sediment types, low salinity and temperature, shallow waters and weak currents; and associated with: squids, pelagic (sardine, mackerel, anchovy) and demersal (black seabream, sandeels, red mullet) species, with higher levels of diversity than the offshore and intermediate communities. [d] Coastal heterogeneous community – characterised by: heterogeneous sediment types (from mud to coarse sands); and associated with: pouting, poor cod, and sole preferential of many flatfish species, highest levels of diversity.

Table S5.5 Dispersal rates and feeding and predation risk parameters of functional groups in Ecospace.

Functional groups	Original base dispersal rate (km/year)	Calibrated base dispersal rate (km/year)	Relative dispersal in bad habitat	Relative vulnerability to predation in bad habitat	Relative feeding rate in bad habitat
Phytoplankton	29	29	2	2	0.05
Phytobenthos	29	29	2	2	0.05
Scallops	5	5	2	2	0.05
Suspension feeders	29	29	2	2	0.05
Deposit feeders	29	29	2	2	0.05
Herbivorous zooplankton	29	29	2	2	0.05
Oysters	5	30	2	2	0.05
Benthic omnivores	29	29	2	2	0.05
Whelk	5	30	2	2	0.05
Crabs	5	5	2	2	0.05
Commercial crabs	20	20	2	2	0.05
Small demersal fish	78	78	2	2	0.05
Shrimps and prawns	29	29	2	2	0.05
Carnivorous zooplankton	29	29	2	2	0.05
Other flatfish	75	75	2	2	0.05
Mullet	157	157	2	2	0.05
Carnivorous megabenthos	30	30	2	2	0.05
Lobster	20	12	2	2	0.05
Dab	75	75	2	2	0.05
Pilchard	157	157	2	2	0.05
Adult sole	78	78	2	2	0.05
Small Gadoids	157	157	2	2	0.05
Juvenile seabass	110	110	2	2	0.05
Adult plaice	75	10	2	2	0.05

Sandeels	50	30	2	2	0.05
Seabream	157	157	2	2	0.05
Gurnards	157	20	2	2	0.05
Scad	1000	300	2	2	0.05
Juvenile plaice	75	10	2	2	0.05
Cuttlefish	141	141	2	2	0.05
Juvenile sole	75	75	2	2	0.05
Adult seabass	157	150	2	2	0.05
Sprat	78	150	2	2	0.05
Juvenile cod	110	75	2	2	0.05
Mackerel	235	235	2	2	0.05
Rays and dogfish	157	157	2	2	0.05
Adult cod	196	100	2	2	0.05
Pollack	157	157	2	2	0.05
Herring	157	150	2	2	0.05
John Dory	157	157	2	2	0.05
Squid	141	141	2	2	0.05
Juvenile whiting	75	75	2	2	0.05
Hake	196	196	2	2	0.05
Adult whiting	157	157	2	2	0.05
Large demersal bottom fish	157	157	2	2	0.05
Seabirds	275	275	2	2	0.05
Toothed cetaceans	974	974	2	2	0.05
Seals	275	275	2	2	0.05
Sharks	275	275	2	2	0.05
Discards	10	10	2	2	0.05
Detritus	29	29	2	2	0.05

Table S5.6 Defining fisheries in Ecospace as assigned to habitat types and MPAs in the eastern English Channel.

Functional groups	**All habitats**	**Offshore**	**Intermediate**	**Coastal homogenous**	**Coastal heterogeneous**	**No-take MPA**	**Limited-take MPA**
Beam trawl		+	+		+		
Demersal otter trawl	+						
Dredge			+		+		
Pelagic trawl	+						+
Hooks and lines			+				+
Net				+	+		+
Traps and pots		+			+		+
Seine	+						+
Other			+	+	+		