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Conservation des habitats marins soumis à des usages multiples: Méthodes, objectifs et contraintes pour l'optimisation d'un réseau d'Aires Marine Protégées en Manche Orientale

Juliette Delavenne

Ecole doctorale: Science de la Matière du Rayonnement et de l'Environnement

Unités de Recherche : Laboratoire d'Océanologie et de Géosciences et Laboratoire Ressources Halieutiques IFREMER centre Manche Mer du Nord

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Paul Marchal	Directeur de thèse
Carlo Rondinini	Rapporteur
Philippe Koubbi	Rapporteur
François Gauthiez	Examinateur
Bernard Drobenko	Examinateur
Sandrine Vaz	Encadrant scientifique
Robert J. Smith	Encadrant scientifique
Jean-Claude Dauvin	Co-Directeur de thèse

Conservation of marine habitats under multiple human uses. Methods, objectives and constraints to optimize a Marine Protected Areas network in the eastern English Channel

Résumé

La Manche orientale représente une zone économique importante qui supporte diverses activités anthropiques comme le tourisme, le transport maritime et l'exploitation de ressources vivantes ou minérales. De plus, cette région possède un riche patrimoine biologique illustré par sa grande diversité d'habitats. Les Aires Marines Protégées (AMP) sont souvent évoquées comme un instrument de gestion permettant d'aménager l'exploitation durable de ces ressources marines, dans le cadre d'une gestion écosystémique intégrée et responsable. Si les Etats ont pour obligation de créer des réseaux d'AMPs dans leurs eaux nationales, chacune d'elles est souvent localisée au cas par cas. Afin de coordonner la mise en place des différents réseaux d'AMPs, une démarche de planification spatiale systématique de la conservation est de plus en plus encouragée. Cette démarche a pour but de proposer un réseau d'AMP qui soit cohérent, même dans un contexte transfrontalier, comme c'est le cas en Manche orientale. Les travaux de recherche menés lors de cette thèse apportent ainsi une contribution scientifique à la mise en cohérence de l'aménagement des activités anthropiques avec les objectifs de conservation de l'écosystème marin de Manche orientale.

Dans le cadre d'une approche de conservation intégrée, toute la biodiversité de la Manche orientale doit être représentée. Pour cela, en complément des typologies benthiques existantes dans la zone, une typologie des masses d'eau a été proposée et validée avec différents jeux de données d'espèces pélagiques.

Marxan et Zonation, deux logiciels largement répandus en planification de la conservation ont été comparés dans le processus de conception du réseau d'AMP en Manche orientale. La conclusion a été que Marxan serait le logiciel utilisé pour la suite des analyses. En effet, ce logiciel est conçu pour atteindre clairement les cibles de conservation, ce qui facilite l'interprétation des résultats. L'utilisation de Marxan requiert cependant de fixer des objectifs de conservation précis permettant une représentation satisfaisante de la biodiversité à conserver. Pour cela, des cibles de conservation par habitat ont été calculé en utilisant les relations aire-espèces par habitat. Les cibles de conservation par habitats sont influencées par divers paramètres tels la taille de l'échantillon ou le niveau de précision de l'habitat. Ici, il a été démontré qu'en plus de la quantité de données disponible, leur qualité et leur pertinence par rapport à l'habitat considéré affectait les résultats. De plus, il apparait que des AMPs conçues pour conserver les habitats benthiques ne seraient pas vraiment appropriées pour conserver la biodiversité de la colonne d'eau sus-jacente.

Puis une étape essentielle de planification de la conservation a été réalisée à travers une analyse des lacunes (gap analysis) à l'échelle de la Manche orientale. Elle a permis de montrer que le réseau d'AMP existant atteint les cibles de conservation calculées dans cette thèse et qu'il couvre 33% de la Manche orientale. Il faut toutefois noter que l'étude des possibles lacunes au niveau de la gestion des AMPs n'a pu être réalisée de façon approfondie car la majorité de ces AMPS ne possèdent pas encore de plan de gestion défini.

Finalement, l'influence de l'intégration des activités humaines dans le processus de conception du réseau d'AMP a été explorée grâce à l'utilisation de données d'effort de pêche et de données de débarquements. De plus, d'autres informations sur le trafic maritime, les extractions de granulats marins et les potentielles zones d'éoliennes en mer ont été ajoutées pour prendre en compte la totalité des usages et réglementation qui génèrent des contraintes spatiales en Manche orientale.

Dans le contexte du développement de la stratégie pour le milieu marin et de réseaux d'aires de conservation dans les eaux européennes, les différentes analyses et études menées au cours de cette thèse contribuent à optimiser le développement d'un réseau d'AMPs en Manche orientale en testant et appliquant des techniques novatrices encore peu utilisées en milieu marin.

Mots clés : Manche orientale, Aires Marines Protégées, planification spatiale, biodiversité, Marxan, typologie d'habitat, cibles de conservation, usages anthropiques.

Abstract

The eastern English Channel is a significant economic area that supports a number of human-based activities, such as tourism and recreational activities, international ports and shipping, and the extraction of both living and mineral resources. In addition, the region supports a number of important marine biological features and large habitat diversity. Marine Protected Areas (MPAs) are increasingly used as a management tool to foster a sustainable exploitation of marine resources in an ecosystem based management framework. All European countries have a legal obligation to develop MPA networks in their national waters. However, there has to date been only limited attempts to coordinate the design and positioning of such networks at an international level and the use of a systematic conservation planning approach is now recommended. This process aims to propose a coherent MPA network, even in a trans-boundary context as in the eastern English Channel (EEC). The studies conducted in this thesis contribute to the scientific knowledge needed to support both anthropogenic activities and conservation objectives in the eastern English Channel.

The representation of the whole biodiversity of the eastern English Channel is important in a context of an integrated conservation approach. With this objective, to complete the existing benthic typologies, a pelagic typology was produced and validated with various pelagic species distribution data to ensure that the total biodiversity of the eastern English Channel would be considered.

Marxan and Zonation, two widely used conservation planning software packages that provide decision support for the design of reserve systems were compared in the MPA network design process in the EEC. It was found that Marxan was most suitable for subsequent analyses in this thesis because it found reasonably efficient and clear solutions to the problem of selecting a system of spatially cohesive sites that met a suite of biodiversity targets, and the results were easily interpretable. However, Marxan needs ecologically relevant conservation targets to ensure a good representation of the biodiversity to conserve. Thus, habitat conservation targets were calculated using habitat-specific species area relationships. Habitat targets are influenced by various parameters such as sample size or the precision level of the habitat classification used. It was demonstrated here that data quantity, but also data quality and relevance to the considered habitat were affecting the results. Moreover, it was found that MPAs based only on benthic habitats may not be fully appropriate to conserve pelagic habitats.

Then, as it is an essential step in a conservation planning approach, a gap analysis was realized at the scale of the EEC. The currently proposed network met conservation targets proposed in this thesis and was found to cover 33% of the EEC. However, a correct assessment of management gaps was not possible as a major part of these MPA do not have management rules yet.

Finally, the influence of the human activity data on the MPA design process was studied using landings and fishing effort data. Other information on maritime traffic, aggregate extraction or offshore windmills zones, and on-going MPA projects were also added to consider the whole set of uses and regulations that generate spatial constraints in the eastern English Channel.

In the context of implementing the Marine Strategy Framework Directive and of developing conservation areas in European waters, the different studies conducted during this thesis may help inform conservation managers in the EEC by testing and applying novel methodologies which are still seldom used in the marine realm.

Keywords: Eastern English Channel, MPA, Spatial planning, biodiversity, Marxan, habitat typology, conservation targets, human uses.

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Abbreviations

CBD: Convention on the Biological Diversity

CGFS: Channel Ground Fisheries Survey

CHARM: CHannel integrated Approach for Marine Resources Management

COP: Conference Of the Parties

EBM: Ecosystem Based Management

EEC : eastern English Channel

EMS: European Marine Site

EU: European Union

EUNIS: EUropean Nature Information System

IBTS: International Bottom Trawl Survey

ICES: International Council for the Exploration of the Sea

MCAA: Marine And Coastal Act

MCZ: Marine Conservation Zone

MPA: Marine Protected Area

MSFD: Marine Strategy Framework Directive

MSP: Marine Spatial Planning

NE: North East

OSPAR: OSlo-PARis

PA: Protected Area

PU: Planning Unit

SAC: Special Area for Conservation

SAR: Species Area Relationship

SPA: Special Protected Area

UNESCO: United Nations Educational Scientific and Cultural Organization

VMS: Vessel Monitoring System

VPUE: Value Per Unit of Effort

Introduction

L'Homme dépend des océans pour un grand nombre de services écosytémiques comme la production de nourriture pour des millions d'individus ou la régulation du climat (Costanza et al., 1997 ; Worm et al., 2006). Cependant, la biodiversité marine doit faire face à un nombre croissant de pressions comme la surexploitation des stocks halieutiques, la pollution, l'extraction de matières minérales, le nombre croissant d'espèces introduites et d'espèces invasives, la dégradation d'habitats et le changement climatique (Bostford et al., 1197 ; Eastwood et al., 2007 ; Ormerod, 2003 ; Roberts, 2003 ; Worm et al., 2006) (Figure 1). Les activités humaines et les pressions qu'elles engendrent ont un effet de plus en plus délétère sur le fonctionnement de l'écosystème marin global (Diaz et al., 2004 ; Sarkar et al., 2006). De plus, une importante perte de biodiversité se surajoute à la perte de services écosystémiques (Worm et al., 2006). Cinq extinctions de masse ont déjà eu lieu sur Terre, qui étaient toutes caractérisées par la perte de 75% des espèces vivantes sur une période de moins de deux millions d'années (Barnosky et al., 2011). Certains scientifiques évoquent maintenant l'imminence d'une sixième extinction, qui pourrait être causée par les activités humaines et leurs influences sur les écosystèmes (Barnosky et al, 2011 ; Hugues et al., 1997).

Les extinctions d'espèces en milieu marin sont plus difficilement observables qu'en milieu terrestre. Elles sont toutefois nombreuses, et des pertes de populations, d'espèces et de groupes fonctionnels, ont été observées dans des écosystèmes tels que les récifs coralliens ou les communautés de poisson côtiers (Jackson et al., 2001 ; Worm et al., 2006).

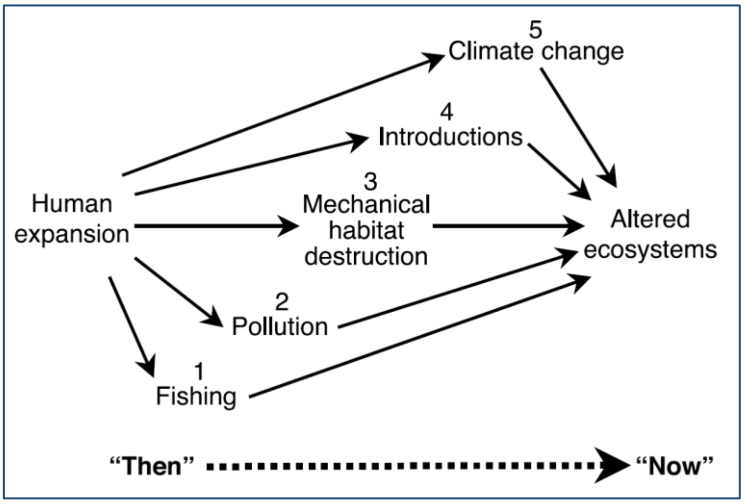


Figure 1 : Séquence historique des perturbations humaines qui affectent les écosystèmes côtiers (d'après Jackson et al., 2001)

L'intensité des activités humaines varie géographiquement, en fonction de l'océan ou de la mer considérée. La région de la mer du Nord, et spécialement la Manche, font partie des domaines maritimes les plus impactés dans le monde (Halpern et al., 2008). La Manche Est est une importante zone économique, scène de nombreuses activités comme le tourisme, la plaisance, le fret, l'extraction de granulats, l'exploitation d'énergies renouvelables et l'activité la plus dominante, la pêche (Carpentier et al., 2009). Les activités de pêche jouent un rôle majeur dans la dégradation des écosystèmes marins, et leurs impacts ont toujours été les premiers à avoir été mis en évidence (Jackson et al., 2001).

Afin de gérer au mieux les multiples activités et de maintenir les services socio-économiques tout en préservant la biodiversité, les nombreux impacts de l'Homme sur les écosystèmes doivent être atténués, et cela peut passer par la mise en place de mesures de gestion écosystémiques (Fraschetti et al., 2011 ; Halpern et al., 2010). En 2005, près de 200 scientifiques se sont réunis et se sont mis d'accord sur une définition de l'approche écosystémique appliquée au milieu marin : « l'approche écosystémique correspond à une approche intégrée qui considère l'écosystème dans son ensemble en prenant en compte l'Homme. L'objectif est de maintenir l'écosystème dans des conditions propices à la production et permettant la résilience afin qu'il soit à même de fournir les services écosystémiques voulus et requis par l'Homme » (McLeod et al., 2005 ; Rosenberg and McLeod, 2005). Cette approche se doit d'être basée sur les meilleures données scientifiques disponibles, et doit être mise en place avec tous les acteurs de la zone concernée. L'approche écosystémique est maintenant recommandée dans des textes de loi internationaux et régionaux, et sa mise en place constitue par exemple l'un des objectifs principaux de la Directive-Cadre Stratégie pour le Milieu Marin (EC, 2008).

Dans ce contexte, le projet multidisciplinaire INTERREG franco-anglais CHARM (CHannel integrated Approach for Marine Resources Management) a été développé afin de produire de l'information scientifique pour les gestionnaires impliqués dans la conservation et l'utilisation des écosystèmes marins de Manche Est (Carpentier et al., 2009). Le projet de recherche va de la description de la Manche, de ses ressources vivantes et de ses activités de pêche à la description des cadres législatifs français et anglais. Ces informations ont ensuite été intégrées à travers diverses approches, telles que la modélisation d'habitats, la description des réseaux trophiques de la Manche, et des analyses exploratoires concernant la mise en place d'une approche systématique de planification spatiale de la conservation (Margules and Pressey, 2000). Cette thèse abonde cette dernière thématique, et aborde différents aspects de l'approche systématique de planification spatiale de la conservation :

- ✓ Disponibilité des données et influence sur la conception d'un réseau d'Aires Marine Protégées (AMP).
    - o Quelles sont les données sur la biodiversité disponibles en Manche Est et quels indicateurs de biodiversité devraient être utilisés (Margules and Pressey, 2000)? Comment le choix de données de biodiversité peut-il influencer la création d'un réseau d'AMPs à travers le processus de calcul des cibles de conservation (Metcalfe et al., 2012 ; Rondinini and Chiozza, 2010) ?
    - o Quelles données économiques devraient être utilisées dans le cadre de la création d'un réseau d'AMPs (Naidoo et al., 2006) ? Comment ces données peuvent-elles influencer la conception du réseau d'AMPs (Carwardine et al., 2009 ; Klein et al., 2009 ; Richardson et al., 2006)?
- ✓ Dans quelle mesure les réseaux d'AMPs français et anglais en Manche Est réussissent-ils à conserver la biodiversité marine (Scott et al., 1993) ?
- ✓ Quels outils d'aide à la décision sont les plus appropriés pour guider la conception des réseaux d'AMPs (Carwardine et al., 2007 ; Moilanen et al., 2009c) ?

Introduction

Mankind depends on oceans for important and valuable ecosystems services, such as provision of food for millions of people or climate regulation (Costanza et al., 1997; Worm et al., 2006). However, marine biodiversity is under increasing pressure due to a diverse range of threats, including overexploitation by fisheries, marine pollution, mineral extraction, introduced and invasive species, habitat destruction and global climate change (Botsford et al., 1997; Eastwood et al., 2007; Ormerod, 2003; Roberts, 2003; Worm et al., 2006) (Figure 1). Human activities and pressures continue to have a detrimental effect on the global marine ecosystem functioning (Diaz et al., 2004; Sarkar et al., 2006). In addition to a degradation of ecosystem services, oceans have suffered a large loss of biodiversity (Worm et al., 2006). The Earth already experienced five large mass extinctions, characterized by a loss of about 75% of living species in a period of less than two millions years (Barnosky et al, 2011). Some scientists point at the risk of entering a sixth mass extinction, caused by human activities and their influence on ecosystems (Barnosky et al., 2011; Hughes et al., 1997).

Although marine extinctions are more slowly uncovered, compared to those of the terrestrial realm, particular ecosystems such as coral reefs or coastal marine fish communities are losing populations, species or functional groups (Jackson et al., 2001; Worm et al., 2006).

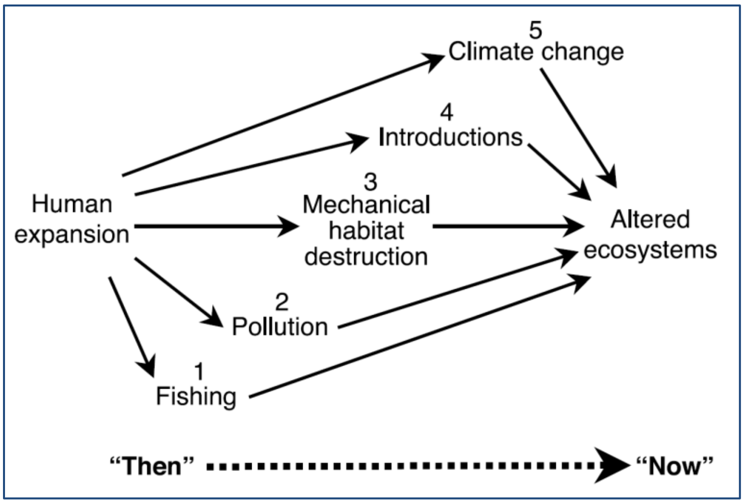


Figure 1: Historical sequence of human disturbances affecting coastal ecosystems. (from Jackson et al, 2001)

The intensity of human activities varies across oceans and seas worldwide. Thus, the North Sea region and especially the English Channel appears to be one of the most highly impacted maritime domains in the world (Halpern et al., 2008). The eastern English Channel is an important economic area for various activities such as tourism, leisure, shipping, aggregate extractions, renewable energy production and most of all fishing (Carpentier et al., 2009). Fishing activities have been claimed to play a central role in the degradation of marine ecosystems, always preceding other human disturbance (Jackson et al., 2001).

To deal with multiple activities and maintain socio-economical services while preserving biodiversity, the various human effects on marine ecosystems need to be mitigated. This can be achieved by implementing an ecosystem-based management (EBM) strategy (Fraschetti et al., 2011; Halpern et al., 2010). In 2005, over 200 scientists agreed a definition of EBM for the oceans: "*EBM is an integrated approach to management that considers the entire ecosystem, including humans. The goal of EBM is to maintain an ecosystem in a healthy, productive and resilient condition so that it can provide the services humans want and need.*" (McLeod et al., 2005; Rosenberg and McLeod, 2005). Such a management approach should be based on the best available science, and should be implemented in collaboration with all concerned stakeholders. The EBM has started to be recommended in international and region policy. In particular, EBM is now a central objective of the Marine Strategy Framework Directive (EC, 2008).

Within this context, a multidisciplinary INTERREG French-English project, CHARM (CHannel integrated Approach for marine Resources Management), was developed to provide scientific information for managers involved in the conservation and utilisation of marine ecosystems in the eastern English Channel (EEC) (Carpentier et al., 2009). This research project encompassed the description of the EEC environment and its living resources, as well as the fishing activities and the French and English legal policy framework. This information was then integrated using a range of approaches, such as population distribution modelling and prediction, description of the trophic network of the EEC and exploratory analyses for applying a conservation planning approach (Margules and Pressey, 2000). This thesis belongs to the latter research topic, and aimed to explore different aspects of the systematic conservation planning. These were:

- ✓ Data availability and its influence on the Marine Protected Areas (MPA) network design.
    - o What biodiversity data are available in the EEC and what biodiversity surrogate should be used (Margules and Pressey, 2000)? How does the choice of biodiversity data influence MPA network design in the conservation target setting process (Metcalfe et al., 2012; Rondinini and Chiozza, 2010)?

  - What socio-economic data should we use in the MPA network design process (Naidoo et al., 2006)? How does socio-economic data influence MPA design (Carwardine et al., 2009; Klein et al., 2009b; Richardson et al., 2006)?
- ✓ How well do French and English MPA networks conserve important marine biodiversity features in the EEC (Scott et al., 1993)?
- ✓ What computational tool is the more appropriate for helping to inform MPA network design (Carwardine et al., 2007; Moilanen et al., 2009c)?

1. Chapter One. Context

1.1 Marine Protected Areas

1.1.1. Definition and origins of Marine Protected Areas

The need to devise methods to conserve and manage the marine environment and resources became apparent during the 1950s and early 1960s. Thus, the first world conference on National Parks (1962) considered the need for protection of coastal and marine areas (Kelleher and Kenchington, 1992).

At the same time, improvements in technologies for the exploration of the seabed for mineral resources and for the exploitation of fishery resources led to large international debates about sea management, sovereignty and jurisdictional rights of nations to the seabed beyond the 3 nautical miles (nm). The third United Nations conference of the Law of the Sea finally enabled nations to take measures (including fishing regulations and protection of living resources) beyond their territorial waters to the 200 nm limit. Since then, various international treaties and conventions have addressed aspects of marine conservation in the context of fisheries, pollution, shipping and science (Cf. section 1.2). In the 1970s IUCN (International Union for the Conservation of Nature) started to produce guidelines for Marine Protected Areas implementation (Kelleher and Kenchington, 1992). In 1970, 118 MPAs existed in 27 nations, increasing to 430 in 69 nations in 1985. Eventually, by the 1990s, almost every countries with a shoreline had at least one MPA implemented (Agardy et al., 2003; Kelleher and Kenchington, 1992).

The term Marine Protected Area is loosely defined as in-water designations, coastal management units that include terrestrial and marine areas, strictly protected areas, or any kind of marine managed areas (Agardy et al., 2003). Each agency or country has its own definition but two core objectives have motivated the establishment of most marine reserves: conservation and sustainable provision for human use (Agardy et al., 2003; Roberts et al., 2003a; Wood et al., 2008). Quoting the IUCN definition as one of many examples, a MPA may be "*Any area of inter-tidal or sub-tidal terrain, together with its overlying water and associated flora, fauna, historical, or cultural features, which has been reserved by law or other effective means to protect part or all of the enclosed environment*" (Kelleher and Kenchington, 1992). The variety of existing definitions and objectives has resulted in a profusion of specific terms to describe various types of MPAs and protection levels: marine park, marine reserve, fisheries reserve, closed area, marine sanctuary, marine and coastal protected areas, nature reserve, ecological reserve, replenishment reserve, marine management area, coastal preserve, area of conservation concern, sensitive sea area, biosphere reserve, 'no-take area', coastal park, national marine park, marine conservation area and marine wilderness area, which were reviewed in Agardy et al. (2003).

In addition to the development of these recent conservation goals, in some places, closed areas for fishing management have been used for centuries, such as in Pacific islands (Gell and Roberts, 2003; Roberts, 2007). Even in France, some areas were prohibited to trawling at the beginning of the 19th Century in front of Marseille to benefit adjacent fisheries. At this time, the fishery scientist Marcel Hérubel was the first to advocate the use of closure areas, or "cantonnemenent" in French, to manage fisheries stocks (Roberts, 2007). Research on Marine Protected Areas is quite recent though, after scientists "rediscovered" the benefits of protecting areas from fishing during the 1970s (Roberts, 2007). Therefore, marine conservation lags several decades behind the land-based conservation movement (Agardy, 1994) which started in the late 19th century in USA with a movement motivated by a desire to preserve sites based on aesthetic criteria and to ensure the survival of threatened species (Whittaker et al., 2005).

1.1.2. MPAs benefits

MPAs can help achieve a broad range of objectives, such as biodiversity conservation and management of human uses including fisheries (Halpern, 2003). The limitations of authorized uses in a MPA help to preserve its biodiversity and maintain the associated ecosystem services (Claudet et al., 2011; Lubchenco et al., 2003). Without fishing activities, fishing mortality is eliminated. The abundance and biomass of commercial species is expected to increase in the short term, while habitat quality could also be improved (Claudet et al., 2011; Lester et al., 2009). In the medium to long term, MPAs would be expected to foster the recovery of populations' size and age structure to pre-exploitation levels and lead to an increase in spawning activities, restoration of trophic cascades, enhancement of species diversity and richness, and the preservation of genetic diversity (Claudet et al., 2011; Gladstone, 2007; Roberts et al., 2005).

MPAs can benefit adjacent fisheries through two phenomena: spillover (net emigration of adults and juveniles) and export of eggs and larvae (Gell and Roberts, 2003; Halpern, 2003). However, the majority of the studies highlighting these positive effects were undertaken in warm waters and coral reefs areas (Bohnsack, 1998; Gell and Roberts, 2003), and the benefits of MPAs in temperate waters with highly mobile species is still questioned (Gell and Roberts, 2003; Kaiser, 2005). However, a review of observed effects of towed-gear exclusion MPAs on fisheries in temperate soft-bottom ecosystems by Goñi et al (2011) identified studies where MPAs had a real positive impact for fishers and local fisheries, with increased catch rates and/or increased sizes of targeted fish. Though, MPAs considered in this review, as in most of empirical studies alike, are strict reserves or "no-take" areas.

However, even if species do not spend their entire life cycle within a MPA, such a reserve can provide valuable protection for fish species at vulnerable stages (e.g., juveniles, spawners), or during migrations (Blyth-Skyrme et al., 2006; Gell and Roberts, 2003). However, such MPAs often result in partial or seasonal fishing closures and in partial protection areas, so their benefits are generally more limited (Lester and Halpern, 2008), or even non-existent (Di Franco et al., 2009), compared to those of no-take areas. Even worse, the significant increase in fishing efforts in adjacent buffer zones can result in an overall negative impact of the MPA (Claudet et al., 2008).

For example, the 'plaice box", a 38 000 km² off the Danish, German and Dutch coasts, was designed to protect juvenile fish for the majority of the commercial fish in the area such as plaice (*Pleuronectes platessa*), sole (*Solea solea*) and to a lesser extent cod (*Gadus morhua*) (Piet and Rijnsdorp, 1998). Seasonal areas closures were introduced in 1989 with fishing restrictions. Its first goal was to reduce discard mortality but no positive effects on targeted species could be measured (Piet and Rijnsdorp, 1998), although the lack of representative reference areas has hampered the evaluation of the plaice box (Pastoors et al., 2000). Moreover, such fishing closures have been shown to displace trawling effort to adjacent areas, thereby threatening non-targeted species, including benthic communities (Dinmore et al., 2003), which are highly sensitive to bottom trawling (Kaiser et al., 2000). Hiddink et al (2006) found that fishing closures could have either positive or negative effects on benthos depending on the fishing activity regulation outside the closure areas.

Partial or seasonal closures and their related effects such as fishing effort reallocation, show that in the fisheries management context, MPAs should be part of a wider ecosystem-based management approach, including marine spatial planning (MSP) (Stelzenmüller and Pinnegar, 2011). MSP is a management tool which aims to develop comprehensive marine spatial plans that are typically implemented through zoning approaches, regulations and consent systems (Douvere, 2008). Local fisheries management measures should be integrated as a component of a large-scale management system, as well as marine spatial plans. That large-scale management system would provide an appropriate platform for managers to make decisions on MPA implementation, of course, but also for other management measures such as fishing effort restrictions, and for stakeholders to get involved in the decision-making process.

Moreover, to consider MPAs as a component of MSP would allow MPA monitoring to be a part of ecosystem monitoring programs, consistent with recommendations by the Marine Strategy Framework Directive (Cf. section 1.2.2) (Douvere, 2008; Halpern et al., 2010; Stelzenmüller and Pinnegar, 2011). Thus, for efficient monitoring, MPAs should have precise and defined goals. Clearly defined management objectives are central to successful MPA networks (Agardy et al., 2003), these should be set to build working collaboration between scientists, local communities, users and management authorities (Agardy et al., 2003; Margules and Pressey, 2000).

1.2. Marine Conservation policy

Many marine conservation management measures result from supra-national decisions or recommendations, which are integrated in the European Union (EU) policy and transcribed at the national level. Marine Protected Areas (MPAs) always arise as a possible management and conservation tool (Roberts et al., 2005; Wood et al., 2008). In 2004, 0.5% of the ocean surface was part of an MPA, and this proportion increased to 1.17% in 2010 (IUCN, 2010). This increase, although far from enough to reach the international recommendations, is the result of regional and national actions to develop MPAs. This percentage should have increased even more, as EU countries committed to develop their MPAs networks by 2012.

In the eastern English Channel, an MPA network exists composed of French and English MPAs, which have different conservation objectives and corresponding legislation. Their origins and related legal frameworks are described here.

1.2.1. Policy at a global level

The consideration of Marine Protected Areas in international texts first occurred in 2002 in the plan of implementation of the World Summit on Sustainable Development (Johannesburg, South Africa) which committed to establish a representative global network of MPAs by 2012 (United Nations, 2002). Then in 2003, at the 5th World Park Congress, the recommendation was made to " greatly improve the marine and coastal areas managed in MPAs by 2012; these networks should include strictly protected areas that amount that amount to at least 20%-30% of each habitat" (Wood et al., 2008).

The Convention on Biological Diversity

At the international level, the United Nation Environment Programme (UNEP) created in 1988 a working group to explore the need for an international convention on biological diversity. The Convention on Biological Diversity (CDB) was open for signatures at the United Nations conference on environment and development (the "Earth summit") in Rio in 1992, committing 193 parties (192 countries and the EU). The primary objectives were the conservation of biodiversity, the sustainable use of its components and the fair and equitable sharing of benefits arising from uses of genetic resources (CBD, 1992).

In 2004, the seventh meeting of The Conference Of the Parties (COP), the governing body of the Convention, adopted the Program Of Work On Protected Areas (POWPA) with the objective to establish and maintain, by 2010 for terrestrial areas, and by 2012 for marine areas, "*comprehensive, effectively managed and ecologically representative systems of protected areas*" (CBD, 2004), with a target of "*at least 10% of each of the world's ecological regions effectively conserved*" (CBD, 2004). Then, in 2009, at the tenth meeting of the COP, in Nagoya (Japan), the parties adopted a revised strategic plan for biodiversity (CBD, 2010) which among other decisions established a conservation target of 17% of terrestrial and inland water areas and 10% of marine and coastal areas (CBD, 2010).

The RAMSAR Convention

The RAMSAR convention was open for signatures in 1972 in Ramsar (Iran) and has been effective since 1975. It consists of a set of member countries and aims to maintain their wetlands of international importance. RAMSAR sites are present both in France and in the United Kingdom, RAMSAR sites protecting intertidal or subtidal habitats and species are considered as MPAs.

1.2.2. Policy at the North East Atlantic level

The OSPAR convention

The convention for the protection of the marine environment of the North-East (NE) Atlantic was open for signatures in 1992 and replaced the previous Oslo Convention (to control pollution from dumping) and Paris Convention (to control pollution from land-based sources and conserve the marine environment of the NE Atlantic). It entered into force in 1998 with 16 contracting parties (15 countries and the EU) (Ardron, 2008a; O'Leary et al., 2012; OSPAR, 1992). Under its five NE Atlantic environment strategies, the OSPAR convention is promoting the implementation of an ecosystem-based approach. Within this context, in 2003, OSPAR ministers adopted recommendations 2003/3 or the establishment of a MPA network of well managed MPAs in the NE Atlantic by 2010 (OSPAR, 2003). It was decided that MPAs should not just be situated in national waters but also in waters beyond jurisdiction. Thus, in 2010, OSPAR Ministers established the world's first network of MPAs on the high seas (O'Leary et al., 2012). They declared six protected areas covering 286,200 km² of the NE Atlantic (Figure 2). Within national waters, each state proposed MPAs to the OPSAR commission, most of which are part of the already existing Natura 2000 MPA network (Ardron, 2008b). The OSPAR commission also produced a list of threatened species and habitats in the NE Atlantic and produced various recommendations for the management of the region (OSPAR, 2008).

Acting under the overarching legal framework of the United Nations Convention on the Law of the Sea (UNCLOS, Montego-Bay, 10 December 1982), OSPAR is an example of regional seas cooperation whereby States can collectively decide to adopt measures to protect the marine environment (O'Leary et al, 2012). This framework makes OSPAR a suitable institution on which to rely for trans-boundaries approaches, facilitating an ecosystem approach and relevant marine planning.

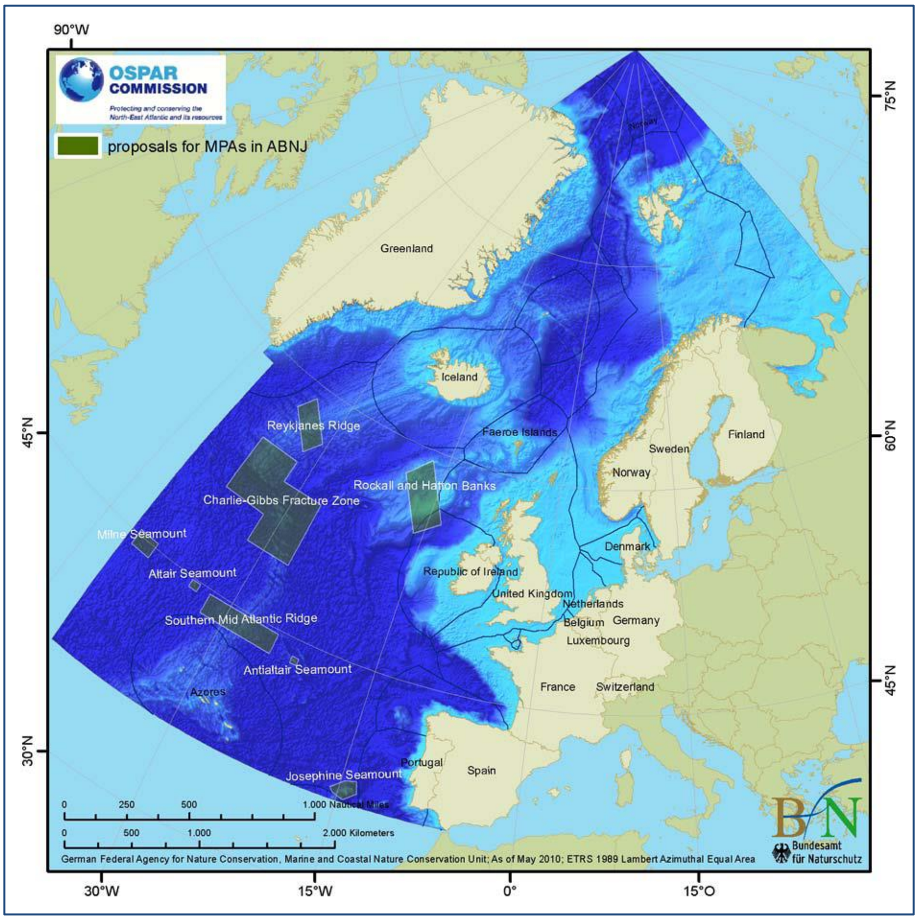


Figure 2: Distributions of OSPAR High Sea MPAS across OSPAR regions. (source: OSPAR commission, 2011 status report on the OSPAR network of MPAs)

The Marine strategy Framework Directive

At the EU level, the (2008/56/EC) Marine Strategy Framework Directive (often referred to as the Marine Directive or MSFD) was adopted on 17th of June 2008, with the aim to protect more effectively the marine environment across the European Union (EC, 2008). It is the first EU legislative instrument relative to the protection of marine biodiversity with the final objective to achieve Good

Environmental Status (GES) in the EU's marine waters by 2020: "*The environmental status of marine waters where these provide ecologically diverse and dynamic oceans and seas which are clean, healthy and productive*" (article 3). It also recalls the CBD objectives encouraging countries to develop MPA networks in their waters. The MSFD promotes the development of marine Natura 2000 areas to reach the CBD objectives (Vong, 2010) and its 6th article encourages regional cooperation and thus, trans-boundaries approaches.

Member states were required to adopt the proposed marine strategy which was due to be transposed to national legislation by the 15th of July 2010 (Cf. section 1.2.3).

Natura 2000 directives

The establishment of a Natura 2000 protected areas network is not marine specific. Being part of the CBD, EU heads of state and governments have made a commitment '*to halt the loss of biodiversity by 2010*'. Within the EU, the terrestrial Natura 2000 network of protected sites represents about 26 000 sites covering almost 1 000 000 km² (European Environment Agency, 2006; Opermanis et al., 2012). The sites are designated for European habitats and species of conservation concern, as listed in the EU (79/409/EEC) Bird and (92/43/EEC) Habitat directives (EC, 1979, 1992).

On 22nd of July 2002, the 6th environmental action Program of the European community defined objectives and priority areas for action on nature and biodiversity, which included seeking to "*further promote the protection of marine areas, in particular with the Natura 2000 network as well as by other feasible Community means*". As a consequence, all European countries need to fulfill their international obligations and to apply the (92/43/EEC) Habitat and (79/409/EEC) Bird Directives to the marine environment. The marine part of the Natura 2000 network had to be an overall part of the European ecological network. Both France and UK proposed Natura 2000 sites for habitats and species listed on the habitat directives and for species listed on the bird directives. The selected sites are named European Marine Sites (EMS) and more specifically, Special Area of Conservations (SACs) for the ones under the Habitat Directive and Special Protected Areas (SPAs) for the ones under the Bird Directive.

SACs and SPAs should be managed to protect the designated features from any damaging activities. Site selection should not be motivated by economic or social potential impacts, but only based on scientific grounds (JNCC and Natural England, 2010).

1.2.3. Integration of the international and EU policy in French and English policy

International policy becomes active when parties sign a convention, and European policy must be transposed into national policy to be applied (Vong, 2010). In France, the MSFD was declined in the "Law Grenelle 2" and in UK in the "Marine Strategy regulations" in 2010. The Natura 2000 directives were more complicated to transpose and their principles can be found in various legal acts.

Both countries developed their own policy about marine conservation and management. In France, the 'Law Grenelle 2" planned that in 2020, 20% of the national waters would be part of a MPA and the half would be "No-take areas". Then, the law of the 14th of April 2006 applied the notion of Marine Protected Area arising from international commitments and European directives into its own law. This law created the "Agence des Aires Marines Protégées" (AAMP), a national agency which aims to contribute to public policies for the creation and development of the MPA network. It also defined 6 types of MPAs (Table 1), one of which was new, the Natural Marine Park. The other recognized types of MPAs are: Natural Reserves with a maritime section, European Marine Sites (Natura 2000), National Parks with a maritime section, maritime sections of Maritime Public Domain (MPD) being managed by coastal conservation and biotope protection orders with a maritime section. Then, in a regulation from the 3rd of June 2011, France integrated few more MPAs in the list to officialise the international recognition of MPAs: are concerned the RAMSAR sites, the World Heritage Sites (UNESCO) and the Man and Biosphere Sites (UNESCO).

In the UK, the "Marine And Coastal Act 2009" (MCAA) represents a multi-sectorial approach to marine policy and created a new type of MPA: the Marine Conservation Zones (MCZ). Five other MPAs are defined by the UK policy (Table 1): European Marine Sites, Sites of Special Scientific Interest (SSSIs) or Areas of Special Scientific Interest (ASSIs), the RAMSAR sites, the Scottish MPAs and the Northern Ireland MPAs. JNCC and Natural England share the management of the MPAs and have divided the UK waters in four regional projects to develop the MPA networks. In the EEC, the Balanced Seas project was in charge of identifying and recommending MCZs to the English government.

Table 1: Conservation objectives of the different kind of MPAs in UK and France (adapted from Vong, 2010). BPO is for Biotope Protection Order, NR is for National Reserves, NP is for National Park, MPD id Maritime Public Domain, NMP is for Natural Marine Park, EMS is for European Marine Sites, MCZ is for Marine Conservation Zones, SSSI for Sites of Special Scientific Interests. French MPAs are in blue, English MPAs are in orange and EMS can be either French or English.

	Conservation Status				
	listed species and/or habitats	**unlisted species and/or habitats**	**Ecosystems (environnement and ressources)**	**landscapes**	**promotion of a sustainable use**
BPO (marine part)	**X**	**X**			
NR (marine part)	**X**	**X**			
NP (marine part)			**X**		
MPD (marine part)			**X**	**X**	
NMP	**X**	**X**	**X**	**X**	**X**
EMS	**X**	**X**	**X**		**X**
MCZ	**X**	**X**	**X**	**X**	**X**
RAMSAR sites (marine part)			**X**		
SSSI or ASSI (marine part)		**X**	**X**	**X**	

Comparing French and English MPA types, we found that RAMSAR sites exist in France but were not recognized as MPAs in France till June 2011 (Vong, 2010), National Parks are present in UK but do not contain marine areas and Natural Reserves exist in the UK too but the only one which had a marine part was converted to a MCZ (Vong, 2010).

The French “Natural Marine Park” (Parc Naturel Marin) initiative is adapted for large marine areas in the 12 nautical miles limit. It aims to conserve and develop knowledge about marine ecosystems, to promote sustainable development of marine activities and to conserve cultural heritage. A Natural Marine Park should also participate to reach the Good Environmental Status as requested by the MSFD. In the EEC, the marine park initiative was launched the 19th of February 2008 and should soon lead to the creation of the “parc naturel marin des Estuaires Picards et de la mer d’Opale”. Marine Conservations Zones (MCZs) are a type of MPA aiming to protect nationally important marine wildlife, habitats, geology and geomorphology to follow Natural England and JNCC’s Ecological Network Guidance (ENG) (JNCC and Natural England, 2010). The identification of the sites has been done using a stakeholder engagement process through the establishment of four regional MCZ projects. In the study area, the balanced seas MCZ project was responsible for identifying MCZs in southeast England. Within the proposed 30 MCZs in the balanced seas study area, Reference areas (RAs) are proposed; they represent the highest level of protection and aim to encompass the specific features and the habitat that need to be included in the MPA network in regard to the ENG.

1.2.4. European and National Marine Protected Areas in the eastern English Channel

Not all the described MPAs types are present in the eastern English Channel. Some very coastal MPAs may be not presented in the map (Figure 3), such as the RAMSAR site of the bay of Somme. Concerning the EMS, on the UK side a distinction is made between the coastal and the offshore Special Areas of Conservation. The so-called "UK marine SACs or SPAs" on the map were terrestrial SACs or SPAs with a marine component. The "offshore pSAC" is a potential marine SAC with a perimeter that was proposed to the European Commission the 30th of August 2012. It is also important to notice that the presented Natural Marine Park may not be the definitive one as this perimeter is still under discussion. Many MPAs are found to overlap; this matter will be investigated in the **Chapter Four** as well as the characteristics of these MPAs in term of their representativity of the biodiversity of the English Channel.

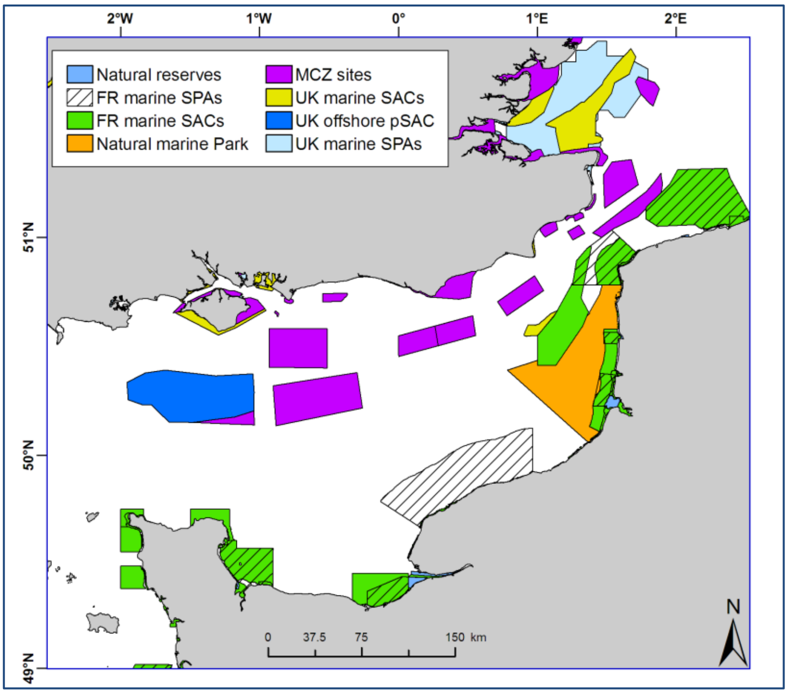


Figure 3 : The actual MPA network in the eastern English Channel

1.3. Conservation planning

1.3.1. Systematic conservation planning definition

Despite the increasing popularity of MPAs as management tools, until recently, decision on the design and locations of a majority of MPAs largely resulted from political or social processes (Halpern, 2003; Margules and Pressey, 2000). In parallel to this expert approach, another process was developed about twenty years ago: "systematic conservation planning".

Systematic conservation planning consists of using specific protocols to identify networks of PAs to be conserved in order to protect biodiversity (Margules and Sarkar, 2007). The key concept underpinning systematic conservation planning is complementarity (Margules and Sarkar, 2007), which means that the biodiversity of a set of locations is a non-additive property of these locations. In other words, the contribution of a given location to the overall representativeness of a PAs network depends less on the local richness of the location than on the extent to which the biodiversity features complement (are different from) those already protected at other locations (Ferrier, 2009; Justus and Sarkar, 2002; Kirkpatrick, 1983). This principle of complementarity requires a dynamic process for PAs selection, because each time a location is selected or added to the existing PA network, the complementarity has to be recalculated for all the other locations (Margules and Sarkar, 2007; Possingham, 2009; Wilson et al., 2009).

Systematic conservation planning can be described as a nine stages process as described in Margules and Sarkar (2007) (Box 1). The detailed investigation of some of these stages is the focus of this thesis and these particular stages will be more precisely described in the following sections.

Box 1: Systematic conservation planning (adapted from Margules and Sarkar (2007) and Margules and Pressey (2000))

(1) Identify stakeholders for the planning region

(2) Compile, assess and refine biodiversity and socio-economic data for the region

- Compile available geographical distribution data on biotic and environmental parameters at every level of organization. Compile Remote-sensing data and survey data at the species level if possible.
- Compile available socio-economic data
- Assess conservation status for biotic entities (rarity, endemism, endangerment)
- Assess the reliability of the data, in particular critically analyze the process of data selection
- Correct for bias and model distributions

(3) Identify biodiversity surrogates for the region

(4) Establish Conservation goals and targets

- Set quantitative conservation targets for species or other features (for example, at least three occurrences of each species or 10% of each habitat surface).
- Set quantitative targets for minimum size, connectivity or other design criteria
- Set precise goals for criteria other than biodiversity including socio-political criteria

(5) Review the existing conservation area network

- Measure the extent to which conservation targets and goals are met by the existing conservation areas
- Identify the imminence of threat to under-represented features

(6) Prioritize new areas for potential conservation action

- Using principles such as complementarity, rarity and endemism, prioritize areas for their biodiversity content
- Starting with the already existing conservation areas network, repeat the process of prioritization to compare results
- Incorporate socio-economic criteria such as various costs
- Incorporate design criteria such as connectivity, size etc.
- Alternatively, carry out the last three steps using different optimal algorithms

(7) Assess prognosis for biodiversity within each newly selected area

- Assess the likelihood of persistence of all biodiversity surrogates in the new conservation areas
- Assess vulnerability of a potential conservation area to external threats

(8) Implement a conservation plan

- Decide on the most appropriate and feasible legal mode of protection for each conservation area
- Decide on the most appropriate and feasible mode of management for each conservation area
- Decide on the relative timing of conservation management when short-term actions are not feasible

(9) Periodically reassess the network

- Decide on indicators that will show whether goals are met
- Periodically measure the indicators
- Return to stage (1)

1.3.2. Biodiversity surrogacy

Biodiversity is mainly described at three levels of biological hierarchy: genes (alleles), species and ecosystems. Because of the complexity of biodiversity, surrogates such as sub-sets of species or habitats types have to be used as a measure of biodiversity (Margules and Pressey, 2000). Moreover, the locations or occupied areas of these biodiversity surrogates have to be detailed enough to be mapped. It is unlikely that it will ever be possible to measure the true variation of biodiversity within or between regions and the effectiveness of surrogates is in consequence not really determinable (Wilson et al., 2009). The choice of surrogates can always be criticized but results most of the time more from data availability than any other factor (Margules and Pressey, 2000).

1.3.3. Target setting

The overall goal of MPA networks (conserving the biodiversity, maintaining ecological processes or managing fish stocks for example) reflect political or institutional priorities and result in the various conservation objectives described in section 1.2.1 (CBD, 2004; Rondinini and Chiozza, 2010; Kelleher and Kenchington, 1992; Smith et al 2009). However, these goals have to be translated into quantitative targets for operational uses (Margules and Pressey, 2000; Rondinini and Chiozza, 2010; Desmet and Cowling, 2004; Pressey et al, 2003). Some targets were advised by international conventions like the Convention on Biological Diversity which recommended conserving 10% of each ecological region (CBD, 2004). But these policy-driven targets have been criticized for their lack of ecological credibility (see review by Carwardine et al, 2009). In response to these criticisms, some researchers started to develop quantitative methods for setting targets (Rondinini and Chiozza, 2010).

Setting species targets often uses techniques based on population viability estimates (Justus et al., 2008; Rondinini et al., 2006). Quantitative methods to define percentage area targets for habitat types are reviewed in Rondinini and Chiozza (2010) and are based on five broad principles: (1) Species-area relationship with a fixed percentage target across habitat types, (2) habitat-specific species-area relationship (Desmet and Cowling, 2004) which is the method used in this thesis and which is described in **Chapter Three**, (3) heuristic principles which can be used when quantitative data lack but most of the time depend on subjective decisions, (4) trade-off target size with reserved area which does not use any biodiversity data and (5) spatially-explicit Population Viability Analysis (PVA). As for the biodiversity surrogate choice, availability of data as well as the broad MPA conservation goal to reach most often conditions the choice in the target setting method.

1.3.4. Computational tools

Systematic conservation planning is usually implemented with software tools using georeferenced data sets and area selection algorithms. The **Chapter Five** of this thesis will compare Marxan and Zonation software. Here, a description of these two software and a brief overview of some others is given.

Marxan software

Marxan stands for "Marine Reserve design using eXplicit ANnealing". It is the "marine" version of Spexan (Spatially EXplicit ANnealing) which was developed to meet the need of the Great Barrier Reef Spatial Planning Authority. They are both based on a program called SIMAN developed at the University of Adelaide (Australia). Marxan was developed in 2000 by Ian Ball and Hugh Possingham of the University of Queensland (Australia) (Ball and Possingham, 2000; Possingham et al., 2000). Marxan is a conservation planning software which uses a heuristic algorithm of the form "simulated annealing". A heuristic algorithm is an algorithm that should find a feasible solution which is close to the optimal solution (Haight, 2009). Thus, it will identify "almost optimal" combinations of Planning Units (PUs), and then propose priority areas for conservation (each solution suggested is called a "portfolio") (Ball et al., 2009; Smith et al., 2006). Marxan is the most widely used decision-support tools in conservation planning. In 2009, 17,100 individuals working in 1,200 organizations from more than 100 countries have used Marxan for marine and terrestrial conservation planning (Ball et al, 2009).

Marxan is a command line program written in the C programming language, but simple and user-friendly interfaces have been created. In this dissertation, the CLUZ (Conservation Land-Uses Zoning) interface was used (Smith, 2004). CLUZ is an extension of the software GIS ArcView version 3.2 (ESRI) which can export files in the Marxan format starting from files in ArcView format. Moreover, the use of a GIS interface makes it possible to work with graphic layers of the data rather than with raw data (matrices).

The basic principle of Marxan is called the "**minimum set approach**". The conservation features must be represented a certain number of times to reach the target decided by the user. They are included as constraints in the process. To reach these targets, the program will select the minimal number of sites while seeking to produce the least expensive solution (or portfolio) possible. The cost of the portfolio is calculated with three elements found in its "objective function" (1) which Marxan intends to minimize.

$$\sum_{PUs} Cost + BLM \sum_{PUs} Boundary + \sum_{\cos t value} SPF * Penalty \tag{1}$$

Where "*Cost*" is the cost of the planning units (a cost is allotted to each *PU* in equation (1)). "*Boundary*" is the boundary cost, which measures, in a portfolio, the length of the sides of the PUs that are shared with unselected units. That means that a "sparse" portfolio is more expensive than an aggregate one. That constitutes a representation of connectivity, the importance of which can be modified by the user through the "*BLM*" (Boundary Length Modifier) parameter. Increasing BLM also inflates the cost of having a sparser portfolio. This is the second term of the equation (1). "SPF" is for Species Penalty Factor and is allotted to each conservation feature. "*Penalties*" represent target penalties: when a target is not reached, Marxan allots an additional cost to the portfolio. By giving a very high penalty to each feature, we ensure that all the targets are met. This is the third term of the equation (1)

Marxan works in an iterative way, testing alternate selections of planning units and preferentially keeping the lower value. There are two standard Marxan outputs. First, the best solution file lists the portfolio with the lowest cost from all the good protected areas networks generated. Second, the summed solution file records the selection frequency for sites across all the good protected areas networks generated.

Marxan is continually developing and improving. Marxan with Zones, an extension of the Marxan software, was recently developed. It has the same functionality as Marxan but it is also able to set priorities for multiple zones, i.e. PAs with different levels of protection (Ball et al., 2009; Watts et al., 2009). Marxan with Zones was applied in a context of marine zoning dealing with fisheries and conservation in California (Klein et al., 2010), and it is currently used in the eastern English Channel.

In 2011, another improvement of Marxan was developed with Zonae Cogito which combines Marxan and all its possibilities with a MapGIS interface. This means that users do not need to rely on commercial GIS software to produce Marxan inputs (Segan et al., 2011).

Zonation

Zonation was developed by Atte Moilanen and the Metapopulation Research team at the University of Helsinki (Finland). It is also a spatial planning software but it is more adapted to large-scale studies (Moilanen, 2005).

The basic principle of the program is to provide a hierarchical prioritization of the landscape based on the biological value of each cell on the studied zone (hierarchical meaning that the “5 best percent” are included in the “10 best percent” etc.). Zonation also uses the principle of complementarity (Moilanen, 2007). To be operational, the software requires grid data with exactly the same space extent and resolution, as the planning units of Zonation are represented by the cells of the grids. This often requires some interpolation or other type of model estimation of surrogates prior to zonation analyses.

The algorithm used is different from Marxan’s, and it can be characterized as an accelerated reverse stepwise heuristic process based on the principle of the “**maximum cover approach**”. Instead of adding, progressively, interesting cells, the process is done by allotting a certain ecological value to the entire zone. The program then removes cells which are the least important to preserve this total ecological value. This approach thus supports connectivity and Zonation has various cell removal rules. It is possible to add cost information in Zonation thanks to a specific option which allots a cost to each cell. The meta-algorithm is as followed from (Moilanen et al., 2009a):

1. Select all cells. Set rank=0
2. Repeat until no cells remain in the landscape
   2.1 Calculate marginal values from all remaining cells that are situated at the edge of the landscape. Mark the marginal value of cell by $\delta_{i.}$
   2.2 Remove the cells having lowest δ_i scores. When removing each cell, store the rank, and set rank = rank+1 and update the representation levels of features. Return to 2.1 if they are any cell remaining in the landscape.

Zonation outputs include three components: (1) a ranking of conservation priority through the landscape which is the used output in the Chapter Three, named the “hierarchical ranking output”; (2) curves describing the performance of species at different levels of landscape removal and (3) species-specific information about habitat quality in the selected top fractions of landscape (Moilanen et al., 2009a).

C Plan

C-Plan was created in 1995 and was the first interactive software system dealing with irreplaceability as a measure of the likelihood of needing a site within a planning region for achieving targets (Pressey et al., 2009). C-Plan has a GIS interface and specific functions to produce different types of outputs such as maps displaying sites and their irreplaceability or their contributions to the targets and other graphical information about individual sites. Since 2003, there is an interactive link between Marxan and C-Plan which means that C-Plan can help running Marxan analyses.

ConsNet

The ConsNEt portal is a web-based resource for conservation planning which allows users to run two software packages: ResNet and MultCSync (Ciarleglio et al., 2009; Sarkar et al., 2009). These software packages enable the selection of conservation areas networks, analysis of biodiversity surrogates, establishment of connectivity between selected areas and incorporation of socio-political criteria (Sarkar et al, 2009). It was developed at the University of Texas at Austin. ResNet can solve both minimum set problems and maximum cover ones.

1.3.5. Where are we now?

Conservation planning approaches have started to be widely used, both for marine and terrestrial zoning and protected areas network implementations. A number of methodological issues have been addressed in the literature, but others are still subject to investigation. Moreover, conservation planning is connected to other fields of ecological and conservation research (Moilanen et al., 2009b), like species distribution modeling (Elith and Leathwick, 2009), metapopulation biology (Nicholson and Ovaskainen, 2009), and the influence of climate change on biodiversity (Araujo, 2009). Moilanen (2009c) attempted to summarize a state of the art of quantitative conservation research, and he identified several important scientific challenges: e.g., multiple zoning; consideration of dynamics (temporal dynamic, ecological processes) and uncertainty (natural variability or model uncertainty). Climate change and shifting population distributions and socio-political considerations are topics where methods still need to be improved as well as the integration of sociologic and economic considerations and the feedback inherent with such concerns.

1.4. The eastern English Channel

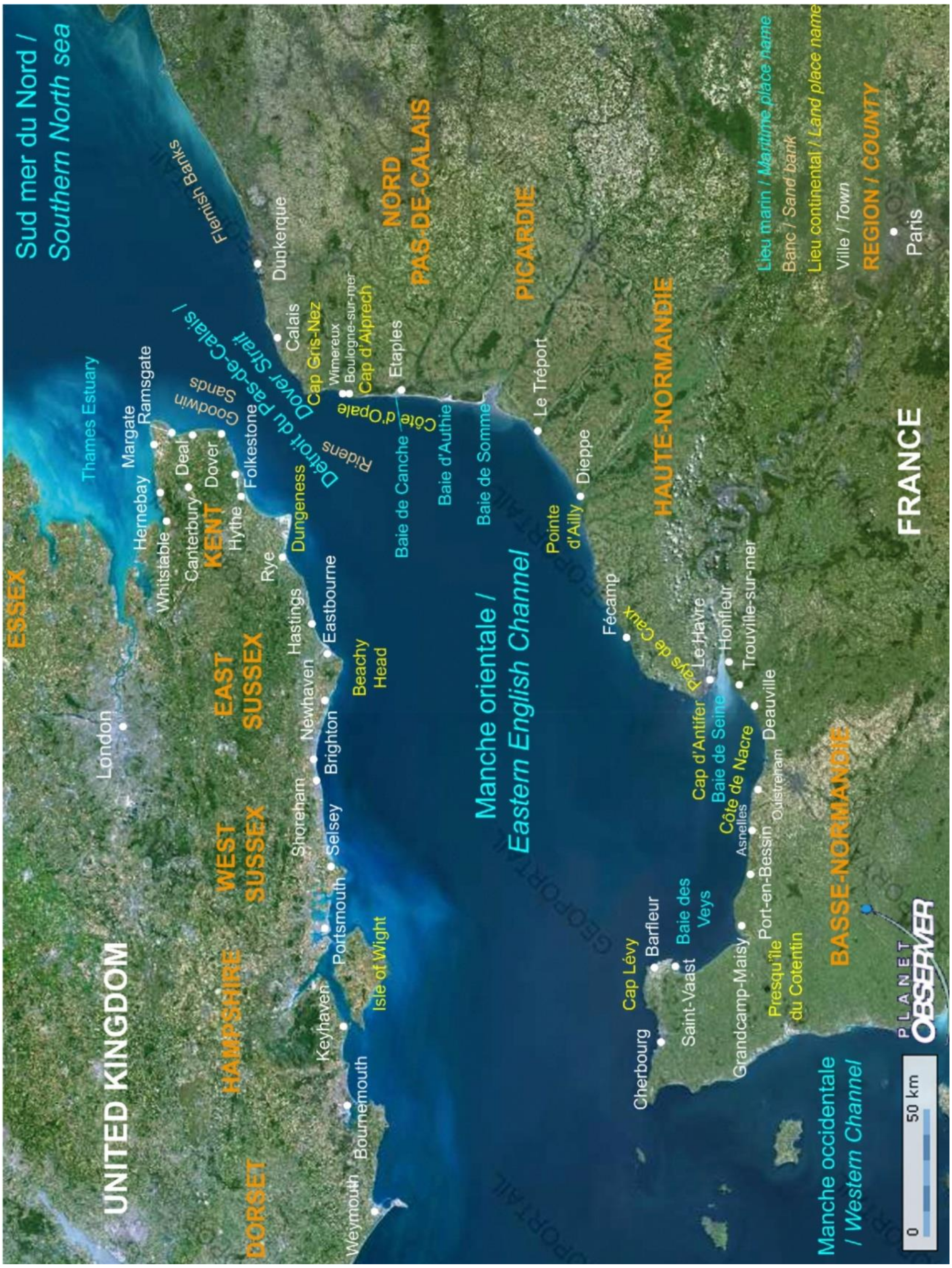


Figure 4: The eastern English Channel and locations names (from Carpentier et al, 2009)

1.4.1. The eastern English Channel environment

The eastern English Channel (EEC) is a shallow epicontinental sea that geographically separates France and the United Kingdom. It is delimited by the Cotentin peninsula in the west and the Dover Strait to the east and is about 35 000 km² (Figure 4). Ecologically, it can also be described as a biogeographical transition zone between the Lusitianian (to the south) and boreal (to the north) provinces.

The Channel is a megatidal sea where tidal currents are dominant and structure the sediment distribution (Larsonneur et al., 1982; Reynaud et al., 1993). Tidal range is always greater than 5 m on the French coast but is closer to 2 m on the English side (around the Isle of Wight) (Dauvin and Lozachmeur, 2006). The study of the residual tidal currents highlights marked retention, dispersion and advection areas (figure 5). In the Dover strait, the residual tidal circulation contributes up to 30% to the total flow rate (with an average of 120 $m^3.s^{-1}$) entering the North Sea. The shear bedstress (Figure 6) resulting from these tidal currents creates a sediment succession from gravels and pebbles in areas with strong currents to fine sediments locked in bays and estuaries (Figure 7) (Alridge and Davies, 1993; Dauvin and Lozachmeur, 2006). At its maximum, the EEC is 70 m deep in the trench running through the center of the Channel; it then becomes progressively shallow toward the east with a depth of 40 m in the Dover Strait (Carpentier et al, 2009).

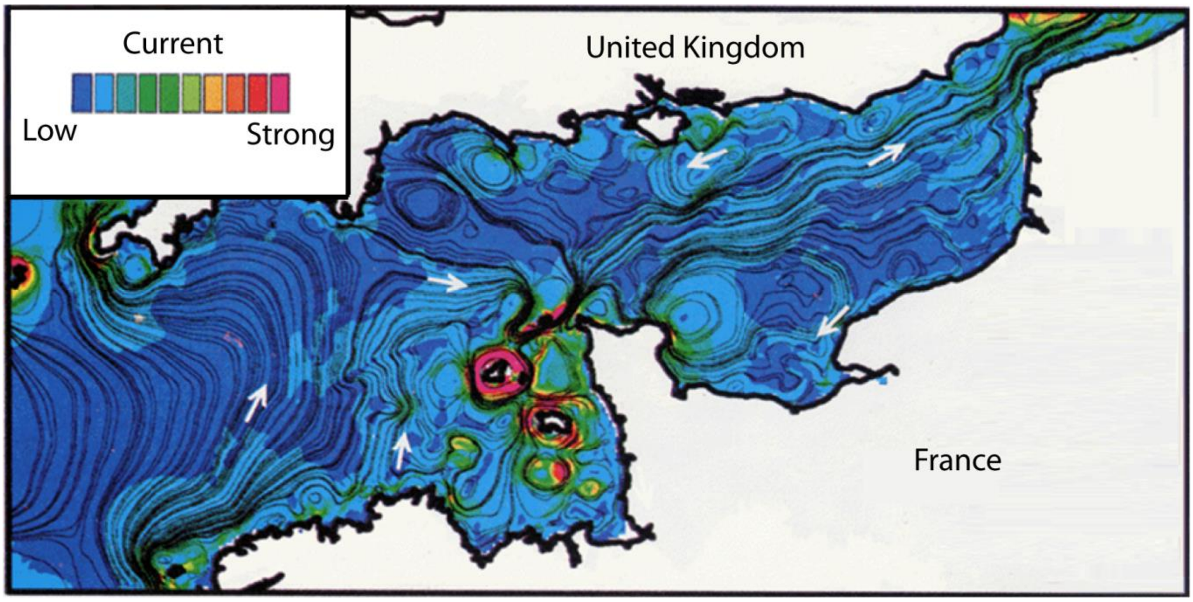


Figure 5: residual tidal currents (after Salomon and Breton, 1991)

Due to the low bathymetry and the strong currents the water column is well-mixed. The shallow inshore areas show large seasonal temperature variations compared to the deeper offshore waters, which remain more stable. This leads to an inversion of the temperature gradient with coastal waters colder than offshore waters in winter and conversely in summer. This pattern is also largely influenced by large freshwater inputs along the French coast with a resulting reduced salinity (Carpentier et al., 2009). This special configuration on the French side of the EEC is referred to as "the costal flow", which circulates from the Seine estuary to the Dover Strait (Brylinsky and Laguadeuc, 1990).

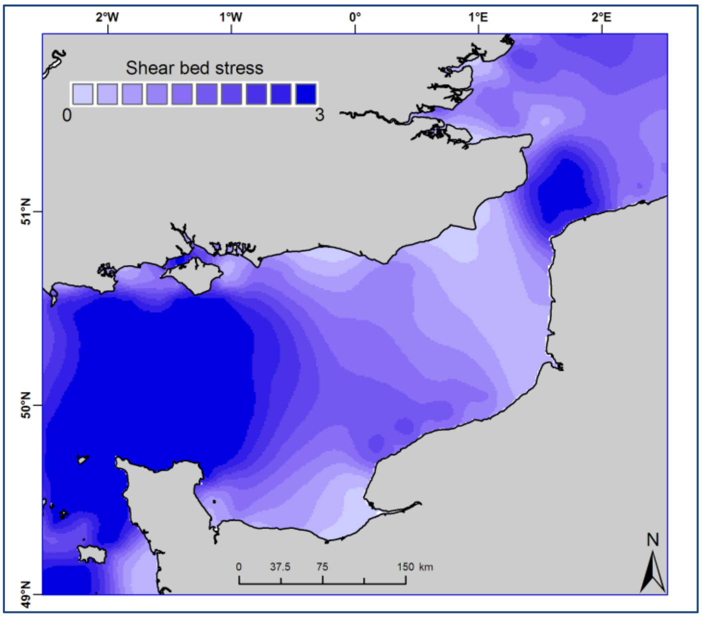


Figure 6: modelled Shear Bed stress

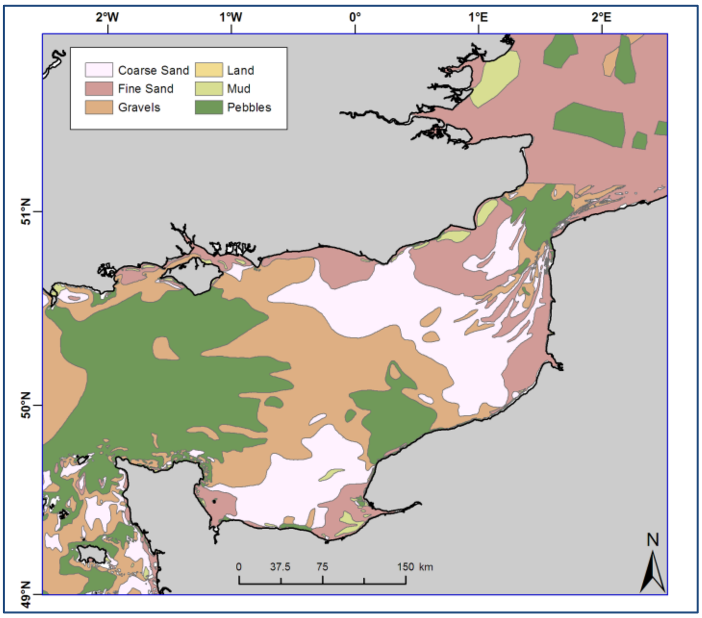


Figure 7: Seabed sediment types extracted from a digital version of the sediment map of the English Channel developed originally by Larsonneur et al (1982)

Currents also contribute to the turbidity of the coastal waters by favouring the re-suspension of mineral and organic particles and the diffusion of continental intakes. This turbidity limits the light penetration into the water column and this may sometime limit the otherwise important primary production (Cabioch et al., 1977; Foveau, 2009).

1.4.2. Habitats in the eastern English Channel

Benthic habitats

Appropriate knowledge on benthic habitats is a prerequisite to an efficient management of marine regions (Coggan and Diesing, 2011). Seabed habitat mapping lags behind terrestrial mapping by several decades, mostly due to the difficulty and cost of sampling (Coggan and Diesing, 2011). To provide a common European framework to produce these habitats maps, the EUNIS (EUropean Nature Information System) now exists (European Environment Agency, 2006). Marine habitats are grouped in the hierarchical EUNIS classification scheme, based on studies led by the JNCC (Connor et al., 2004), OSPAR parts and ICES (International Committee for the Exploitation of the Sea) working groups (Lozach, 2011). In **Chapter Three, Four and Six**, I will make extensive use of the benthic habitat map produced within the EUNIS framework by Coggan and Diesing (2011) (Figure 8). The EUNIS classification scheme has different levels of precision, in the EEC, rocky habitats are classified up to the third level and soft sediment habitats to the fourth level i.e. more precisely (Coggan and Diesing, 2011). This map was produced based only on environmental data. The introduction of biological descriptors (species composition) could lead to finer levels of description (level 5 and 6) (Lozach, 2011). However, the authors compared their work to the faunal associations described by Holme (1966), and found good matches. In the eastern English Channel, benthic communities closely match the sediment structures (Carpentier et al., 2009).

Other benthic classifications of the area exist. In 1995, San-Vincente añorve produced a map comprising five benthic communities associated with different types of sediments (Carpentier et al., 2009; San-Vicente Añorve, 1995). This map is the one used in the **Chapter Five** of the thesis because when this study was produced the EUNIS map produced by Coggan and Diesing in 2011 was not yet available. In contrast to the EUNIS map, this benthic habitats map represents benthic species communities. The EU Habitat Directive (1992) also provided a classification of benthic habitats (Glemarec and Bellan-Santini, 2005) and over the nine described marine habitats, five are found in the EEC (Dauvin and Lozachmeur, 2006).

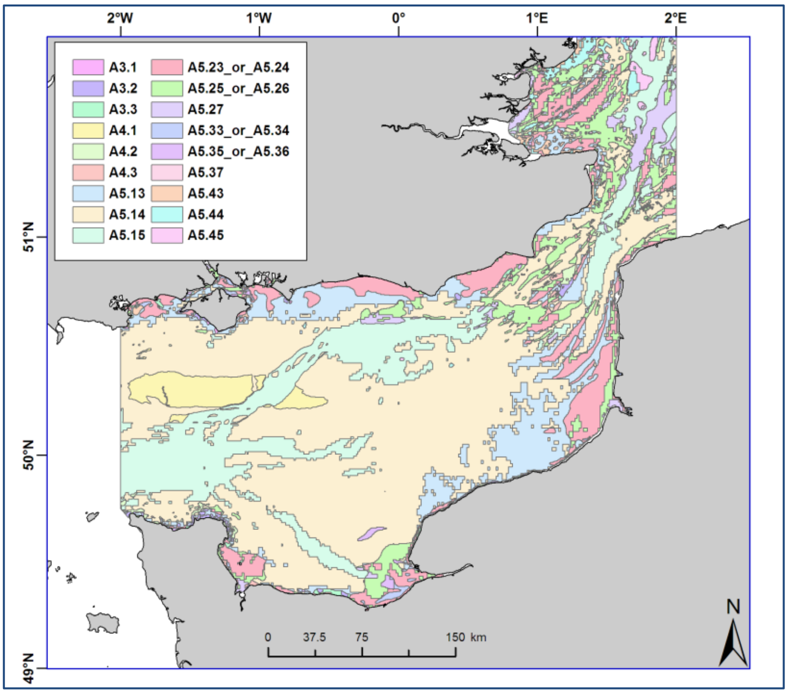


Figure 8: Modelled EUNIS map for the eastern English Channel and a part of the North Sea. Rocky habitats are described to level 3 and sedimentary habitats are described to level 4. Data from Coggan and Diesing (2011)

Fish communities

The fish communities of the EEC were described by Vaz et al (2007) from Channel Ground Fisheries Survey (CGFS) data. The CGFS has operated each October in the area since 1988. A total of 84 species were considered, including two cephalopods and three macro-invertebrates. It appeared that the fish communities were strongly spatially structured (Figure 9) into four types of assemblages in response to the environment, the hydrology and the sediment types(Vaz et al., 2007). Class 1 (in green in the Figure 9) is an offshore community mainly represented by elasmobranchs species and poor cod and displaying a relatively lower diversity than classes 3 and 4 in coastal areas. This class is characteristic of relatively more oceanic hydrological conditions. Class 2 (in yellow) is an intermediate community represented by both pelagic and demersal species. Class 3 (in orange) is a coastal homogeneous community represented by squids, pelagic and demersal species, with higher diversity levels than the two first classes. Class 4 (in red) is a costal heterogeneous community which is favoured by many flatfish species, and which displays the highest diversity level. (Vaz et al, 2007: Carpentier et al, 2009).

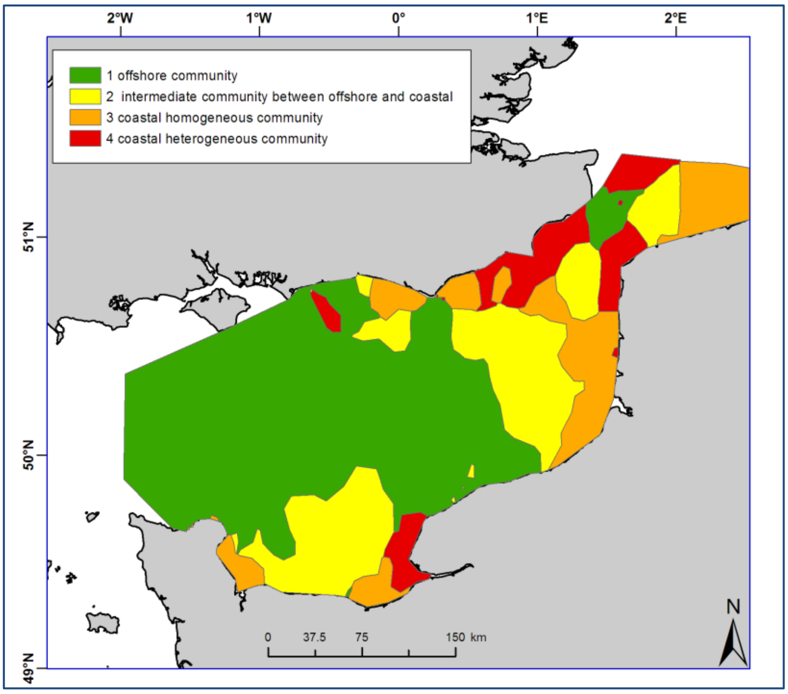


Figure 9: Marine fish communities' classification (from Vaz et al, 2007)

The water column

While the EEC seabed has been studied and categorized in various studies and projects, such as MESH (MESH, 2008) and UKSeaMap (Connor et al., 2006), the water column remains the unexplored part of the EUNIS habitat classification. Within the MSFD framework (EC, 2008), Gaillard-Rocher (2012) produced a pelagic typology of the entire French coast. In **Chapter Two** of this thesis, seasonal water column typologies of the study area will be developed with appropriate parameters at the scale of the EEC. These typologies will be validated by phytoplankton, zooplankton and pelagic fish data derived from sea surveys.

1.4.3 Human uses in the eastern English Channel

In 2008, Halpern and collaborators produced a global map of human impacts on ecosystems (Figure 10). The cumulated impact was calculated from land-based anthropogenic drivers such as nutrient inputs and pollution and ocean-based anthropogenic drivers such as fisheries, shipping or invasive species. Four areas arose as very highly impacted and the North Sea region, especially the Channel, was one of them (Figure 10).

The English Channel is a "congested sea" (Buléon and Shurmer-Smith, 2008) with nearly 500 ships of over 300 tons which enter and leave the Channel every day, 90 to 120 daily rotations by ferries between the continent, the British isles and the Channel Islands, more than 2000 fishing vessels which are licensed in the whole Channel, and 350 000 leisure crafts. In the EEC there are also numerous communication and power cables and gas transport installations. Finally, the EEC is also an important area for extracting living resources (fishing) and mineral resources (aggregates extraction). Moreover, numerous projects exist for deploying sea-based energy production farms with a majority of offshore wind farms and potentialities for wave and tidal power being also explored.

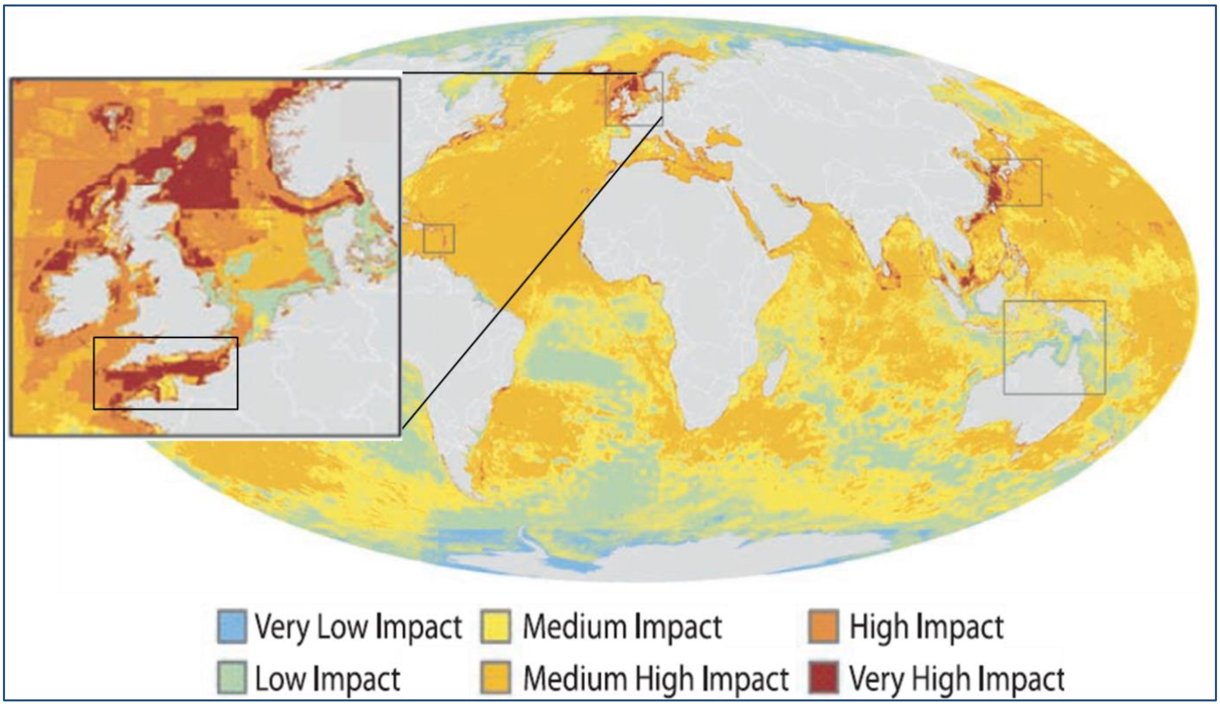


Figure 10: global map of cumulative human impacts across the oceans with a zoom on the highly impacted North Sea region (adapted from Halpern et al, 2008).

The Channel coastal area is also the focus of many human activities. Tourism is an important economic activity in the EEC - the seaside resort concept was literally invented in the Channel during the 19th century (Buléon and Shurmer-Smith, 2008) - and the coastal areas increasingly attract new human uses. A large number of ports are set along the coast, including industrial harbours such as Le Havre and Dunkerque, fishing ports at Boulogne sur Mer (France) or leisure ports at Deauville (France) and Brighton (United Kingdom). Fish and shellfish farming is also an important activity in several areas.

Aggregate extractions

Worldwide, marine sediment have become an increasingly important source of sand and aggregates for the building industry, land reclamation, beach nourishment and coastal defences (Desprez, 2000). The EEC presents a large paleovalleys network, which is now filled with coarse sediments, creating a large aggregate source (Lozach and Dauvin, 2012). The UK and French Governments have approved extraction licenses and some areas where applications were submitted are still waiting for approval (Figure 11). Various past or on-going studies have investigated the impacts of aggregate extraction on the EEC and North Sea environment and biological communities (Barrio Frojan et al., 2011; Desprez, 2000; Lozach, 2011; Lozach and Dauvin, 2012). Common impacts of aggregate extractions are changes in the nature and stability of sediments, alteration of bottom topography, the redistribution of fine particulate material (Desprez, 2000) and screening, which is a phenomena of sediment discharge in the water column (Boyd et al., 2004). These two consequences can impact the surrounding biological communities (Lozach, 2011; Lozach and Dauvin, 2012). The aggregate extraction sites were used as a socio-economic layer in **Chapter Six** as well as windfarm potential sectors (Figure 11).

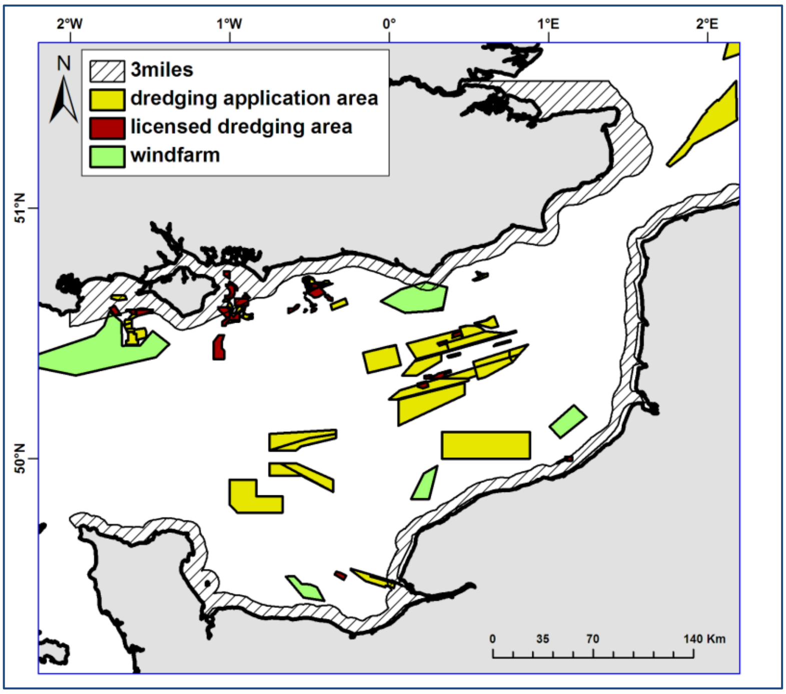


Figure 11: Aggregate extractions sites and windfarm potential areas in the EEC

Fisheries

Fishing is the most prevalent human activity impacting the marine environment (Pauly et al., 2005), and it is of highest importance in the EEC. In 2005, 696 French and English fishing vessels were active in the area and French landings represented 218 M€ (Carpentier et al, 2009). French and English fisheries can be divided into three categories: inshore fisheries (within 12 nm zone), offshore fisheries (outside the 12 nm zone) and mixed fisheries (fishing both inshore and offshore), each fishery can be related to specific fishing gears and target species. The French fishing fleet mainly consists of trawlers, netters, dredgers, polyvalent vessels, and it employs about 2 919 fishers (Carpentier et al, 2009). In the UK, only 57 vessels were reported as landing fish from the EEC in 2005 with a majority of trawlers, netters and potters.

In the UK, France, and all countries neighbouring the North Sea, landings and fishing effort are mainly monitored using sale slips collected in auctions and from mandatory European logbooks for vessels longer than 10m (Carpentier et al, 2009). The NE Atlantic was divided in areas of 1° longitude and 0.5° latitude by the International Committee or the Exploration of the Sea (ICES), the zones are known as “ICES statistical divisions” (Figure 12) and are used to better identify the origins of catches. The landing data will be used in the **Chapter Five and Six**. Moreover, From January 2005, all vessels over 15m in length have been requested to transmit their position at interval of 2h or less (EuropeanCommission, 2003), these satellite-based Vessel Monitoring System data (VMS) will be used to represent fishing patterns.

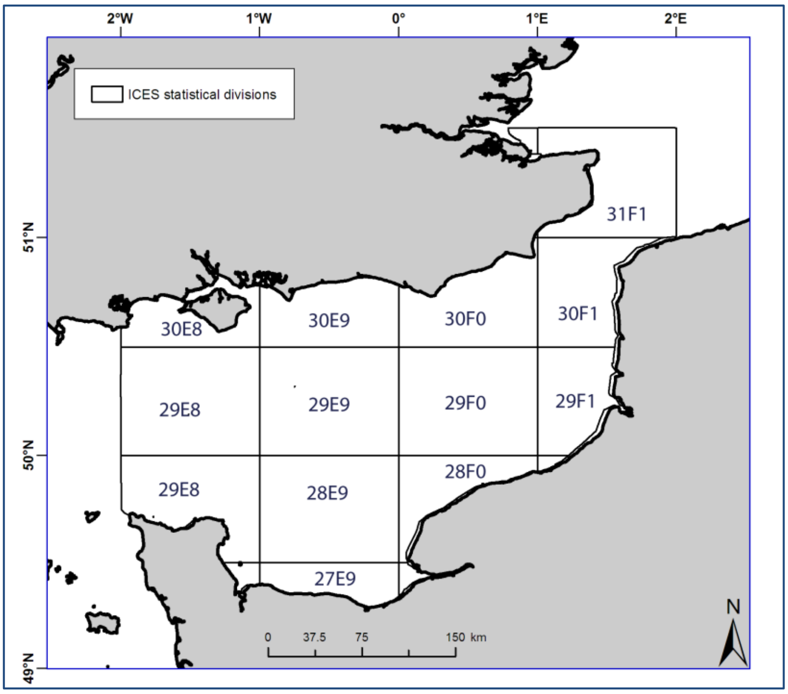


Figure 12: ICES statistical division in the EEC

1.5. Objectives, methods and plan of the thesis

1.5.1. Objectives and plan

In the following chapters, I answered some questions related to the implementation of an ecosystem-based approach to manage the EEC through the implementation of an MPA network at the regional level without consideration for national boundaries. Based on the systematic conservation planning steps described in Box 1, I focused here on five steps corresponding to the five next chapters using various numerical and analytical methods (Table 3)

- ***Steps (1) and (2). Collect existing data and identify biodiversity surrogates for the region.***

- This thesis benefitted from the numerous data produced or gathered during the CHARM project (Carpentier et al., 2009) such as fish communities, benthic communities, various fish and benthic species distributions and preferential habitat. These biodiversity surrogates were used in all the following chapters. Additionally, if in most of marine exercises of conservation planning, only benthic habitats and seabed characteristics were considered, **Chapter Two** created a definition of the pelagic typology of the EEC.

- ***Step (3). Establish Conservation goals and targets.***

Using both benthic and pelagic habitats, quantitative targets were calculated using the Species-Area Relationship method in **Chapter Three.** This chapter also investigated the influence of biological data and biodiversity surrogates on the result.

- ***Step (4) Review the existing conservation area network***

A gap analysis of the existing MPA network at the regional scale was carried out in **Chapter Four.** The MPA network success to reach the habitat-specific targets developed in step (3) was investigated.

- ***Step (5) Use computational tools to prioritize new areas for potential conservation action***

C**hapter Five** compared two decision-support tools used for designing MPA networks and their sensitivity to some parameters such as the target levels and the cost used. In order to facilitate the comparison, fixed targets were used and not the habitat-specific ones calculated in the chapter three.
In **Chapter six,** Marxan was also used to provide information and perform sensitivity analyses about the effect of introducing the local socio-economic data in the study, and especially information about the fishing sector which is of primary importance in the EEC.

Finally, C**hapter Seven** was a general discussion suggesting perspectives with regards to the different findings of the thesis.

1.5.2. Methods

The results presented in the chapter Two to Six are based on different papers either published or in preparation (Table 2). Each chapter focuses on one aspect of the systematic conservation planning process and uses various ecological features and parameters. The chapter Three focuses on setting habitat targets based on the Species-Area Relationship (SAR) approach which lead to variable targets for each habitat. These habitat-specific targets were not used in chapters Five and Six as fixed target for all habitats were preferred in order to identify specific cost influence in the sensitivity analyses. Table 2 summarizes some criteria used for the chapter Two to Six.

Table2: Description of the used parameters in the sensitivity analyses of chapter Two to Six

Chapter	Associated article	Use of a computational tool such as Marxan	Ecological features used	targets	Cost representation
2	**Delavenne, J.,** Marchal, P., Vaz, S. 2013. Defining a pelagic typology of the eastern English Channel. *Continental Shelf Research, 52*, 87-96.	No	Hydrological and physical environment	None	None
3	**Delavenne J**., Vaz S, Metcalfe, K., Smith R.J., Marchal P. Habitat Targets with the Species Area Relationship approach using different biological datasets and typologies. *In prep*	Yes	- Benthic typology from Coggan and Diesing (2011) - Pelagic typology (Delavenne et al, 2013) - Surveys stations	SAR approach	Planning unit area
4	**Delavenne, J**., Vaz, S., Marchal, P., Smith, R.J. A gap analysis of the eastern English Channel. *In prep*	No	- Benthic typology - Pelagic typology - Habitat Suitability Index maps for 8 species under status	- ENG targets -CBD targets - chapter 3 targets	None

5	**Delavenne J**., Metcalfe K., Smith R.J., Vaz J., Martin C.S., Dupuis L., Coppin F., Carpentier A. 2011. 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	Yes	- Benthic communities (San-Vicente Añorve, 1995) -Distribution maps for 8 species (Carpentier et al, 2009) - Physical environment (Carpentier et al, 2009)	- 10% - 30 % - 50%	- Planning Unit area - accessibility cost based on human activities - fishing cost: French landings at ICES square scale
6	None	Yes	- Benthic typology (Coggan and Diesing, 2011) - Pelagic typology (Delavenne et al, 2013)	- 20 %	- fishing effort for France and UK (hours fished) - Value Per Unit of Effort (VPUE) using fishing effort and landings (Fr and UK) - accessibility cost based on other human activities

The different numerical and analytical methods used in the next chapters are summarized in the table 3.

Table 3: the different numerical and analytical methods used in this thesis

Method	Purpose	Chapter
Hierarchical agglomerative clustering	Aggregate points with similar environmental conditions to produce habitat typologies	2
Principal component analysis	Distinguish habitat depending on their environmental conditions	2
ANOSIM (analysis of similarities)	Verify if the biological compositions of the pelagic habitats were substantially different	2

Linear discriminant analysis	Determine the extent to which pelagic habitats were good surrogates for particular biological communities	2
Marxan analysis	Run MPA network scenarios	3; 5; 6
Zonation analysis	Run MPA network scenarios	5
Spearman's rank test	Assess the similarity between Marxan/zonation selection frequency outputs	5; 6
Wilcoxon's sign rank test	Assess the similarity between Marxan best solutions outputs	5
Index of spatial difference (Lee et al, 2010)	Assess the spatial similarity between Marxan selection frequency outputs	3
Specie-Area Relationships equation	habitat conservation targets calculation	3
Minimize Difference Threshold (MDT) transformation	Transform probabilities of presence in presence/absence data	3
Value Per Unit of Effort calculations	Calculate a cost equation for Marxan analyses	6
Indicator kriging and kriging	Produced a continuous pelagic typology combining maps of the 7 classes. Produce probability of presence maps for various species.	2; 5
Geographic Information system	Lead various spatial analyses and use georeferenced data	2; 3; 4; 5; 6

2. Chapter Two

Defining a pelagic typology of the eastern English Channel

Abstract

Classifying marine habitats is a growing research field and is of increasing interest to spatial planners and managers. Most studies predominantly focus on the seabed in order to determine benthic habitat types, and only limited attention has been paid to the water column. Classification projects aim at identifying candidate management units for the application of various European or national regulations such as the EU Water Framework Directive. Here, we propose a seasonal classification of the water column in the eastern English Channel, which we validated using available biological data. For the three tested compartments, phytoplankton, zooplankton and pelagic fishes, the validation results were satisfactory, with recall values The percentage of observations correctly assigned in a given water column class) ranging from 0.5 to 1. This validation was a crucial step to verify that the proposed typologies were ecologically relevant and to use it as a biodiversity surrogates in management and conservation plans. Because management plans are generally set on an annual rather than a seasonal basis, we also produced a "multi-seasonal" typology encompassing seasonal variability, which can be used as an appropriate all year round description of the water column attributes in the eastern English Channel.

Key-words: eastern English Channel, pelagic typology, clustering analysis, seasonality

2.1. Introduction

There has been in recent years an increasing demand, from marine policy-makers and managers, to develop knowledge about the distribution and habitat of marine organisms and thereby improve the scientific basis to ecosystem-based management at international (International Council for the Exploration of the Sea, ICES or OSPAR for OSlo PARis convention initiatives (OSPAR, 1992), regional (Marine Strategy Framework Directive (EC, 2008)) and national levels (Coggan and Diesing, 2011; Costello, 2009). Thus, habitat typologies, which are now commonly used as biodiversity surrogates in ecosystem-based management or conservation planning (Moilanen et al., 2009c), have increasingly been investigated in the last decade in habitat mapping studies.

There is no unique, widely accepted, definition of habitat. Depending on the purpose and the scale of the study, a habitat can be defined by different features such as geography, physical parameters or species composition (Costello, 2009). In this study, habitats are defined according to the EUNIS definition (European Nature Information System): "a habitat is a place characterized primarily by its physical features and secondarily by the species of plants or animals that live there" (Davies *et al.*, 2004). If most of the European continental shelf seabed has already been categorized (Cameron and

Askew, 2011; Connor et al., 2006) there are only few examples of classification of the water column above the seabed, although the pelagic realm represents 99% of the biosphere volume (Game *et al.*, 2009) and represents a distinct, vastly unexplored part of the EUNIS habitat classification.

The study takes place in the eastern English Channel (EEC), which is a very important area from an economic and ecological point of view. It supports spawning grounds and nurseries for various fishes including valuable commercial species (Carpentier et al., 2009; Loots et al., 2010), and it is also a migratory route for fish, birds, and marine mammals. The EEC is also a biogeographical transition zone between the warm temperate Atlantic oceanic system and the boreal North Sea, and it encompasses a wider range of ecological conditions than many other European seas (Carpentier et al., 2009; Dauvin, 2008). In addition, the EEC is an intensively used area, which provides substantial economic return for fisheries, maritime traffic, aggregate extractions and other economic sectors (Buléon and Shurmer-Smith, 2008).

Whereas a seabed typology has already been defined for the EEC based on physical data (Coggan and Diesing, 2011), or benthic communities (MESH, 2008), few attempts have been made to describe its water column (Connor *et al.*, 2006). However, the pelagic realm is of primary importance for many organisms living in the EEC. This applies of course to pelagics, but also to those bentho-demersal organisms that are influenced by both the seabed and the water column, and even to numerous benthic species, which are subject to a pelagic phase during their life cycle. Various studies investigating habitat models for different species, including benthic invertebrates, fishes and cephalopods (Carpentier et al., 2009; Martin et al., 2010; Vaz et al., 2008) showed that proxy of the water masses such as temperature, salinity or turbidity determined to a large extent the distribution of those species habitats. These results suggest that the characteristics of the water column should also be taken into account to develop management and conservation plans that would comprehensively protect the EEC's biodiversity.

In this study, a water column typology of the EEC was developed. This typology encompassed seasonality to take into account the most important part of the temporal variability of the water column. The seasonal typologies obtained in the present work were tested against various biological datasets from autumnal and winter survey data available for these two seasons. This was done to verify to which extent the water masses so defined could discriminate between the different ecological communities. Finally, a "multi-seasonal" pelagic typology was produced that integrated all four season's characteristics into a single, all year round summary typology.

2.2. Materials and methods

2.2.1. The study area

The eastern English Channel (EEC) geography and environmental conditions are described in the chapter one. The EEC is influenced by the Atlantic Ocean to the west and is connected to the North Sea through the Dover Strait. Hence, oceanic waters cross the EEC towards the North Sea under a dominant westerly winds regime. In terms of hydrodynamism, the EEC is characterized by its strong tidal currents and the seabed shear stress resulting from tidal currents on the bottom may be relatively high, especially in narrow areas.

2.2.2. The descriptors of the water column

To produce seasonal pelagic typologies of the EEC, different environmental data layers have been gathered from various sources. The choice of these data was based on their known contribution to the structuration of the different biological communities of the study area, and also on their availability over the whole EEC area. Some data were *in-situ* observations, while others were model-derived. Depth combined bathymetry and hydrodynamic modeled mean sea level to illustrate the average water column thickness at mid-tide for an average tide coefficient. Bathymetric data were derived from SHOM(Service Hydrographique et Océanographique de la Marine) hydrographic charts and the mean sea level was estimated using a hydrodynamic model (Carpentier *et al.*, 2009). Because the EEC is a megatidal area, the strength of tidal currents is an important structuring feature for the water column, and this has been estimated by the shear bedstress which was estimated using a 2-D hydrodynamic model originally developed for the Irish Sea but extended to cover the Northwest European shelf (Alridge and Davies, 1993). Both depth and bedstress were constant physical parameters across seasons. The difference of temperature between the maximum (in August) and the minimum (in February) was added as a third constant parameter because this yearly temperature variation may highlight the “coastal flow” structure which is a particularly structuring feature of this area (Brylinsky and Laguadeuc, 1990; Koubbi et al., 2006). It was calculated with monthly satellite imagery data mean between 1986 to 2006. Four seasonal parameters were used in addition to the three constant parameters detailed above: the surface temperature and KPAR (Photosynthetically Available Radiation indice), which are both derived from satellite imagery data, and bottom salinity and surface current speeds, which both are model outputs. The sea surface temperature (SST) is calculated using the infra-red channels of the AVHRR (Advanced Very High Resolution Radiometer) sensor on-board NOAA (National Oceanic and Atmospheric Administration) satellites platforms (Carpentier *et al.*, 2009). The SST calculation algorithm is described in Walton et al. (1998). The KPAR is the attenuation coefficient of photosynthetically available radiation, and it is correlated to the suspended particulate matter data and Chlorophyll *a*. Both SST and KPAR were available as monthly averages, over the last 21 years. The bottom salinity came from the freely available ICES datacenter, and it was available as monthly averages between 1971 and 2000 (Berx and Hughes, 2009). The surface current speed data have been extracted from the ECOSMO model (Schrum *et al.*, 2006), and these were also available as monthly averages. These four seasonal parameters were available with a 10 km spatial resolution. All these data layers have been resampled on a regular grid of 10 km representing 610 grid cells.

2.2.3. Classification methodology

A Gower dissimilarity coefficient was calculated (Gower, 1971; Legendre and Legendre, 1998) on standardized data and a Hierarchical Agglomerative Clustering (HAC) method was then applied to the resulting dissimilarity matrix. The group average method (or unweighted pair-group method) was used; this technique accounts for group structure and it is reasonably space-conserving. In other words, the probability to be associated to a group is not determined by its size. A typology was produced for each season: spring (March, April, May), summer (June, July, August), autumn (September, October, November) and winter (December, January, February), using the seven parameters described above, the seasons were chosen to coincide with the water column classification work developed in the UKseaMap project 2006 (Connor *et al.*, 2006). Depth, shear bed stress and temperature difference were constant. Seasonal means were calculated for the monthly surface temperature, salinity, KPAR and current speed parameters. In addition, an integrated typology was produced to reflect the multi-seasonal variability performing an HAC on 19 parameters: the three constant parameters, and each seasonal value of the four others.

The classification cut level was obtained by combining two criteria. The optimal number of groups to retain was established by using the Calinsky criterion (Calinsky and Harabasz, 1974; Guidi et al., 2009; Milligan and Cooper, 1985; Smith et al., 2008) which compares the sum of squares of the partition between and within groups. The criterion has been applied to all resulting dendrograms. Then, in order to prevent large numbers of undersized groups with small spatial extent, some clusters were regrouped to the upper cut level to have at least 1% of the observations into each group.

2.2.4. Interpolation and regionalisation of the classification:

The continuous pelagic typologies maps were obtained using indicator krigging of each class on a 1 km² resolution. This interpolation techniques is adapted to nominal variables (Webster and Oliver, 2001) and resulted into a map of occurrence probability of each class at any given location. The maps of the 7 classes were then combined selecting at each location the class with the highest probability of occurrence.

2.2.5 Evaluation of the classification using Principal Component Analysis

In order to verify the representativeness and robustness of the resulting classifications, the results were projected in a Principal Component Analysis (PCA) to verify that each cluster could be clearly distinguished from each other in a reduced ordinated space.

Table 4: Taxa used in the study, originating from the IBTS survey for the winter observations and from the CGFS survey for the autumn ones. Only the 10 more frequent taxa are cited for the zooplankton (35 taxa in total) and the phytoplankton (94 taxa in total). N is the number of samples

Pelagic fish (autumn)	**Pelagic fish (Winter)**	**Zooplankton (winter)**	**Phytoplankton (winter)**
Horse Mackerel (*Trachurus trachurus*)	Horse Mackerel (*Trachurus trachurus*)	Appendicularia	Cryptophyceae spp.
Sprat (*Spratus spratus*)	Sprat (*Spratus spratus*)	Brachyura	Nitzschia longissima
Herring (*Clupea harengus*)	Herring (*Clupea harengus*)	Acartia	Gymnodinium spp
Sardine (*Sardina pilchardus*)	Sardine (*Sardina pilchardus*)	Paracalanus spp.	Paralia sulceta
Northern squid (*Loligo forbesi*)	Northern squid (*Loligo forbesi*)	Pseudocalanus	Thalassiosina spp.
European squid (*Loligo vulgaris*)	European squid (*Loligo vulgaris*)	Temora	Ciliophora spp.
Mackerel (*Scomber scombrus*)	Mackerel (*Scomber scombrus*)	Calanoida	Nanoflagellés spp.
Black bream (*Spondyliosoma cantharus*)	Black bream (*Spondyliosoma cantharus*)	Chaetognata	Skelatoneam costatum
Gilthead seabream (*Sparus aurata*)		Centropaeges	Pleurosigma spp.
Seabass (*Dicentrarchus labrax*)		Clupeidae	Mediophyceae spp.
Anchovy (*Engrausis encrasicolus*)			
N = 138	**N = 61**	**N=18**	**N=88**

2.2.6. Ecological validation of the typology

In order to verify that each seasonal typology had an ecological relevance, the seven water bodies were tested to verify that they were truly different in term of their composition in pelagic fishes or planktonic taxa. Unfortunately, due to limited data availability, only the winter and the autumnal typologies were investigated in this way. Biological datasets used for this validation originated from two scientific surveys: the Channel Ground Fish Survey (CGFS) and the International Bottom Trawl Survey (IBTS). The CGFS covers exclusively the EEC in October since 1988. The IBTS normally occurs in January in the North Sea, but its coverage has been extended to the EEC since 2008. The autumnal typology was tested against the pelagic fishes and cephalopods abundances recorded in the CGFS trawls between 1988 and 2010 (Table 3). Because of the large inter-annual variability in this data set, species abundance data were averaged over time on a 0.1° regular grid. The winter typology was tested against three biological compartments: pelagic fishes and cephalopods (from 2008 to 2011), zooplankton (2008) and phytoplankton (from 2008 to 2010). Since the typology definition was 10 km, the biological stations located less than five kilometers away from the boundary between two classes were removed from the analyses to avoid edge effect. All used taxa are named in table 4.

First, the homoscedasticity hypothesis was tested with the Marti Anderson's method (Anderson et al 2006), which is a multivariate equivalent of Leven's test or homogeneity of variances. Then, to improve the variance homogeneity, all abundance data were log-transformed. Second, an analysis of similarities (ANOSIM) was realized to determine if the biological compositions of the different water bodies were statistically different ($p<0.001$). This analysis consists of looking at the ranked dissimilarities between and within groups (Clarke, 1993). Finally, linear discriminant analyses were conducted to determine the extent to which the defined water bodies were able to reveal differences in biological communities, i.e. if the defined water bodies were good predictors of the biological communities' structure. The discriminant analysis produced contingency tables representing the percentage of correct group allocations, as predicted by the analysis. The significance of the discriminant analysis results was tested by comparing intra-group and total variances with a Bartlett test (Bartlett, 1937).

All the analyses were performed using the R statistical software (http://www.R-project.org/) with cluster and vegan packages.

Table5: Physical characteristics of each typology: mean and standard deviations (sd). a) spring typology, b) summer, c) autumn, d) winter and e) multi-seasons typology.

a)	Depth	Shear bedstress	SST variation	SST	Salinity	KPAR	Current speed
1	10.26	1.03	11.7	8.99	34.6	1.651	0.071
sp	(4.89)	(0.480)	(0.376)	(0.262)	(0.114)	(0.162)	(0.034)
2	24.28	0.843	11.2	9.18	34.3	0.419	0.071
sp	(6.78)	(0.370)	(0.459)	(0.172)	(0.082)	(0.161)	(0.021)
3	58.6	2.93	8.19	9.40	34.8	0.322	0.197
sp	(9.45)	(0.190)	(0.664)	(0.145)	(0.081)	(0.041)	(0.062)
4	44.2	1.38	9.15	9.15	34.5	0.272	0.111
sp	(7.21)	(0.471)	(0.681)	(0.087)	(0.124)	(0.037)	(0.018)
5	18.5	1.19	10.1	8.95	34.8	0.685	0.093
sp	(9.12)	(0.606)	(0.594)	(0.134)	(0.069)	(0.125)	(0.042)
6	35.6	2.75	9.93	8.94	34.7	0.660	0.143
sp	(9.10)	(0.339)	(0.811)	(0.175)	(0.139)	(0.224)	(0.039)
7	27.3	0.950	10.6	8.74	34.6	0.534	0.116
sp	(9.29)	(0.481)	(0.791)	(0.138)	(0.088)	(0.255)	(0.035)

b)	Depth	Shear bedstress	SST variation	SST	Salinity	KPAR	Current speed
1	13.4	0.787	11.7	16.7	34.6	0.844	0.048
su	(5.97)	(0.454)	(0.290)	(0.278)	(0.033)	(0.292)	(0.023)
2	27.8	10.8	11.0	16.2	34.6	0.304	0.074
su	(6.66)	(0.354)	(0.503)	(0.303)	(0.052)	(0.097)	(0.019)
3	60.4	2.94	7.98	14.9	34.9	0.199	0.186
su	(8.11)	(0.145)	(0.398)	(0.201)	(0.021)	(0.017)	(0.069)
4	41.4	1.23	9.17	15.3	34.8	0.208	0.099
su	(9.31)	(0.507)	(0.625)	(0.275)	(0.061)	(0.046)	(0.025)
5	14.0	1.01	10.4	16.1	34.9	0.427	0.099
su	(7.18)	(0.633)	(0.325)	(0.358)	(0.049)	(0.142)	(0.048)
6	41.2	2.90	9.55	9.55	34.8	0.285	0.180
su	(10.9)	(0.186)	(0.510)	(0.510)	(0.054)	(0.058)	(0.071)
7	31.0	2.56	10.6	10.6	34.0	0.375	0.177
su	(8.44)	(0.395)	(0.514)	(0.514)	(0.040)	(0.127)	(0.029)

c)	Depth	Shear bedstress	SST variation	SST	Salinity	KPAR	Current speed
1a	12.3	0.997	11.7	14.7	34.8	1.60	0.083
	(6.95)	(0.451)	(0.354)	(0.357)	(0.136)	(0.285)	(0.038)
2a	25.4	0.899	11.2	15.5	34.4	0.295	0.073
	(6.63)	(0.390)	(0.473)	(0.136)	(00.045)	(0.108)	(0.021)
3a	53.1	2.94	8.54	15.6	34.5	0.324	0.200
	(13.8)	(0.170)	(0.870)	(0.137)	(0.036)	(0.111)	(0.064)
4a	45.2	1.45	9.02	15.6	34.5	0.221	0.115
	(7.18)	(0.472)	(0.682)	(0.086)	(0.048)	(0.034)	(0.016)
5a	17.4	1.59	10.5	15.3	34.5	0.621	0.144
	(7.53)	(0.486)	(0.349)	(0.084)	(0.028)	(0.161)	(0.051)
6a	24.7	0.674	10.1	15.4	34.6	0.445	0.121
	(11.4)	(0.309)	(0.679)	(0.102)	(0.075)	(0.180)	(0.047)
7a	30.1	1.87	10.9	15.3	34.8	0.512	0.125
	(9.96)	(0.700)	(0.660)	(0.123)	(0.012)	(0.184)	(0.044)

d)	Depth	Shear bedstress	SST variation	SST	Salinity	KPAR	Current speed
	11.3	0.977	11.7	7.53	34.6	2.46	0.097
1w	(5.96)	(0.480)	(0.479)	(0.474)	(0.088)	(0.343)	(0.047)
	24.9	0.864	11.2	8.27	34.2	0.396	0.085
2w	(7.04)	(0.417)	(0.458)	(0.237)	(0.112)	(0.220)	(0.027)
	57.1	2.95	8.30	9.89	34.7	0.336	0.214
3w	(10.5)	(0.135)	(0.698)	(0.328)	(0.076)	(0.122)	(0.058)
	45.5	1.42	8.93	9.52	34.4	0.279	0.131
4w	(6.7)	(0.452)	(0.554)	(0.301)	(0.104)	(0.096)	(0.015)
	34.5	2.69	10.2	8.84	34.6	0.838	0.179
5w	(6.40)	(0.424)	(0.450)	(0.268)	(0.104)	(0.360)	(0.046)
	26.2	0.998	10.2	8.83	34.5	0.674	0.129
6w	(10.5)	(0.497)	(0.526)	(0.259)	(0.150)	(0.367)	(0.049)
	20.1	1.54	11.6	7.87	34.7	0.679	0.109
7w	(5.66)	(0.529)	(0.371)	(0.268)	(0.058)	(0.337)	(0.027)

e)	Depth	Shear bedstress	SST variation	SST	Salinity	KPAR	Current speed
1	42.7 (14.8)	1.29 (0.576)	8.99 (0.386)	12.3 (0.154)	347 (0.042)	0.411 (0.122)	0.080 (0.011)
2	8.47 (3.56)	0.874 (0.378)	11.9 (0.121)	11.9 (0.280)	34.6 (0.106)	1.75 (0.222)	0.050 (0.008)
3	30.5 (5.78)	10.4 (0.361)	10.7 (0.586)	12.4 (0.234)	34.4 (0.069)	0.278 (0.079)	0.088 (0.021)
4	15.5 (4.17)	0.538 (0.228)	11.8 (0.143)	12.4 (0.150)	34.3 (0.068)	0.579 (0.216)	0.047 (0.010)
5	47.7 (7.56)	1.89 (0.763)	8.75 (0.614)	12.4 (0.202)	34.6 (0.078)	0.248 (0.050)	0.132 (0.031)
6	17.4 (9.93)	1.63 (0.984)	10.4 (0.470)	12.3 (0.206)	34.7 (0.029)	0.731 (0.128)	0.079 (0.024)
7	61.7 (10.6)	3.00 (2.31)	8.03 (0.597)	12.5 (0.195)	34.7 (0.035)	0.269 (0.034)	0.245 (0.045)
8	27.7 (9.94)	2.31 (0.844)	9.74 (0.645)	12.3 (0.221)	34.7 (0.010)	0.590 (0.126)	0.175 (0.015)
9	16.9 (7.49)	1.17 (0.492)	11.5 (0.404)	11.9 (0.188)	34.7 (0.019)	1.53 (0.319)	0.110 (0.026)
10	31.2 (7.44)	0.617 (0.236)	10.1 (0.672)	12.2 (0.222)	34.5 (0.054)	0.366 (0.115)	0.150 (0.029)
11	14.9 (7.13)	0.478 (0.271)	10.2 (0.258)	12.1 (0.116)	34.6 (0.049)	0.885 (0.251)	0.131 (0.035)
12	37.7 (10.1)	2.56 (0.447)	10.4 (0.369)	12.1 (0.143)	34.6 (0.029)	0.605 (0.278)	0.186 (0.019)
13	27.7 (8.62)	1.65 (0.612)	11.0 (0.649)	12.1 (0.260)	34.7 (0.033)	0.466 (0.011)	0.098 (0.020)

2.3. Results

2.3.1. The seasonal typologies

Four seasonal typologies were obtained from each classification of the seasonal parameters and these were mapped in figure 13. The environmental characteristics of the different water bodies were described in table 2 for all typologies and some are illustrated in figure 14. According to the used method, 7 clusters were found for each seasonal typology. For the typology representing the multi-seasonal variability, it was interesting to have more groups, the Calinski criteria was the best for 16 groups but after regrouping to have more than 1% of the observations in each groups only 13 clusters were found.

In the four seasonal typologies some water masses appeared to be stable and exhibited a certain spatial persistence although they were not identical from season to season (Figure 13 and 14). The types 3sp, 3su, 3a and 3w looked similar by their locations even if their mean values for environmental parameters such as temperature or salinity were different throughout the year. The deepest water bodies of the area always had the strongest bedstress and surface current (Table 5). We can observe that the water types 2sp, 2su, 2a and 2w (French coasts) and the types 4sp, 4su, 4a and 4w (central Channel) were also spatially persistent throughout the year. By contrast, the water bodies associated to the English coasts and the Dover Strait appeared more season-dependent. This is reflected in the multi-seasonal typology (Figure 15), where the stable French coast and central Channel classes were still present, while the English coasts, Dover Strait and estuaries were composed of more classes and were more scattered.

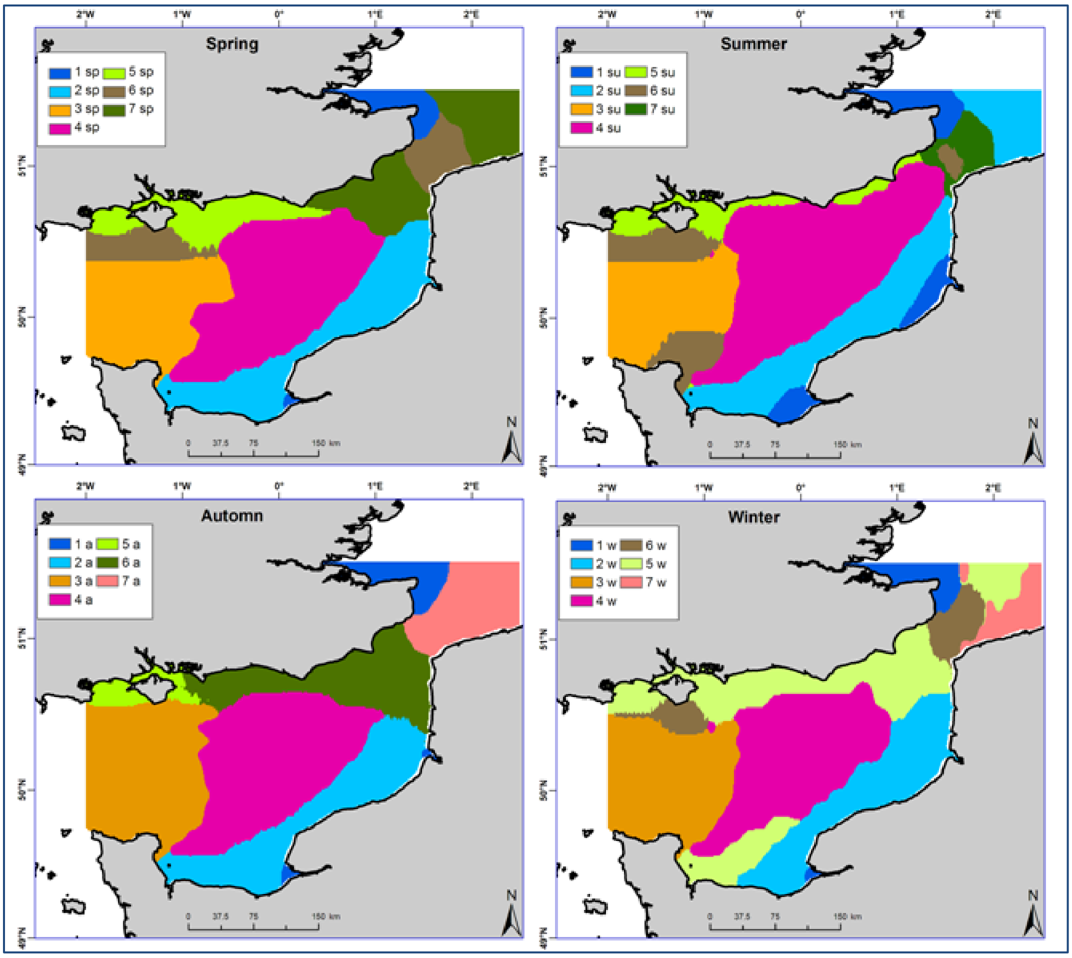


Figure 13: Seasonal typologies. Seven water types are presented each for each season

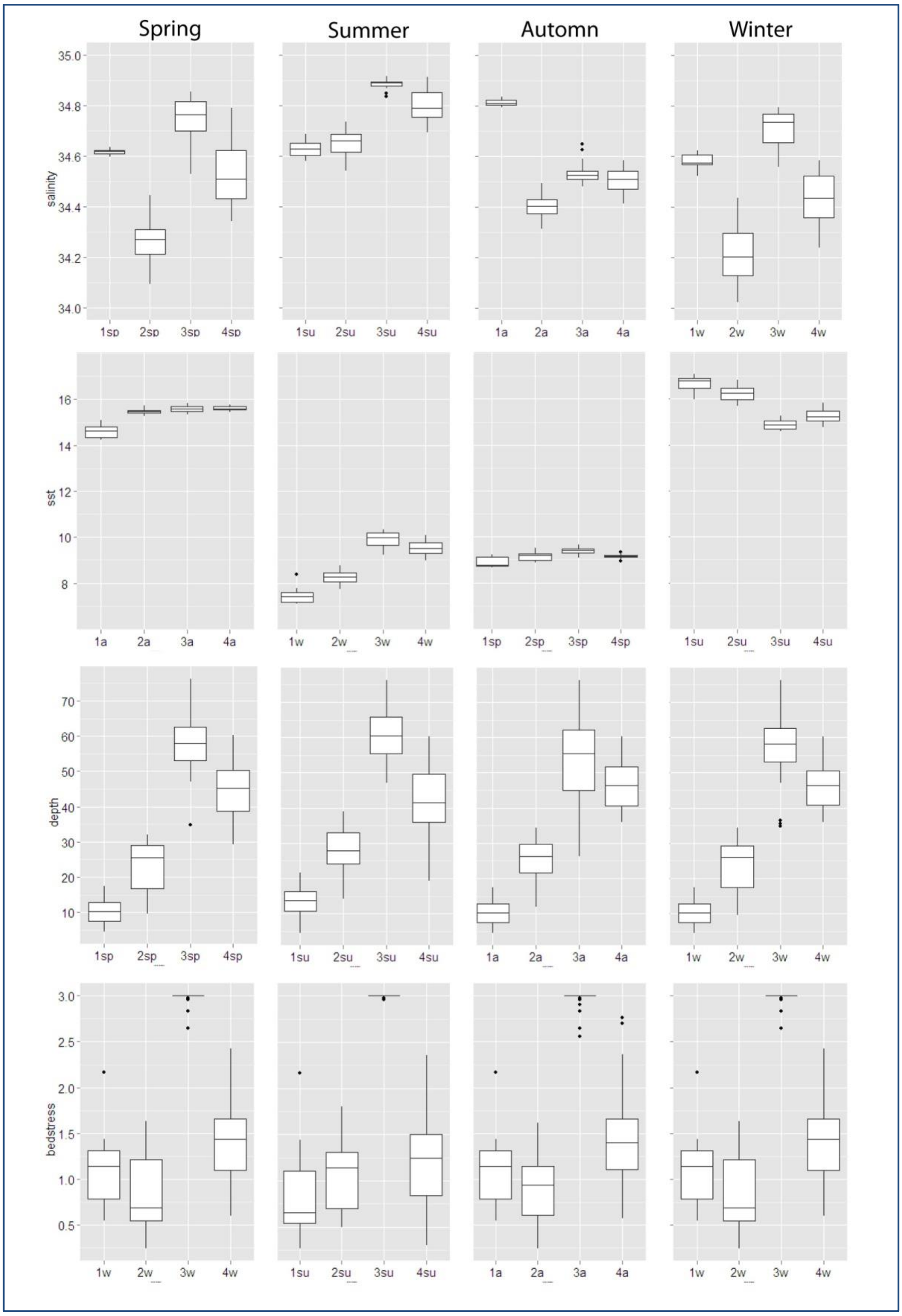


Figure 14: variations of four parameters (Salinity, sst, depth and shear bedstress) during the four seasons in the water types 1(sp, su, a, w),2 (sp, su, a, w),3 (sp, su, a, w) and 4 (sp, su, a, w)

The PCA showed that the 7 water bodies may be clearly discriminated on the first two axes for each seasonal typology (Figure 16). These results also confirmed that it was not necessary to proceed to an ordination of the data, before the realization of a classification. Moreover, more than 50% of the variance was explained on the first axis indicating that a strong environmental gradient strongly structured the data. The water types 1a, 1w, 1sp and 1su were always well separated from the 6 other classes and defined by high kpar values. For the four seasons a gradient from the types 1, 2, 4 and 3 was clearly noticeable along the first axis following a gradient of decreasing temperature variability, and increasing depth, current speed and bedstress. The transition patterns to classes 5, 6 and 7 were more variable and season specific. The figure 14 illustrates the seasonal variation for four of the seven criteria used. Depth, bedstress, sst and salinity were chosen because their contributions to the first two axes of the PCA were the most important. Only the types 1 (sp, su, a, w), 2 (sp, su, a, w), 3 (sp, su, a, w) and 4 are presented because of their relative spatial stability overtime, which is well illustrated by the depth and bedstress plots showing little variations over the four seasons.

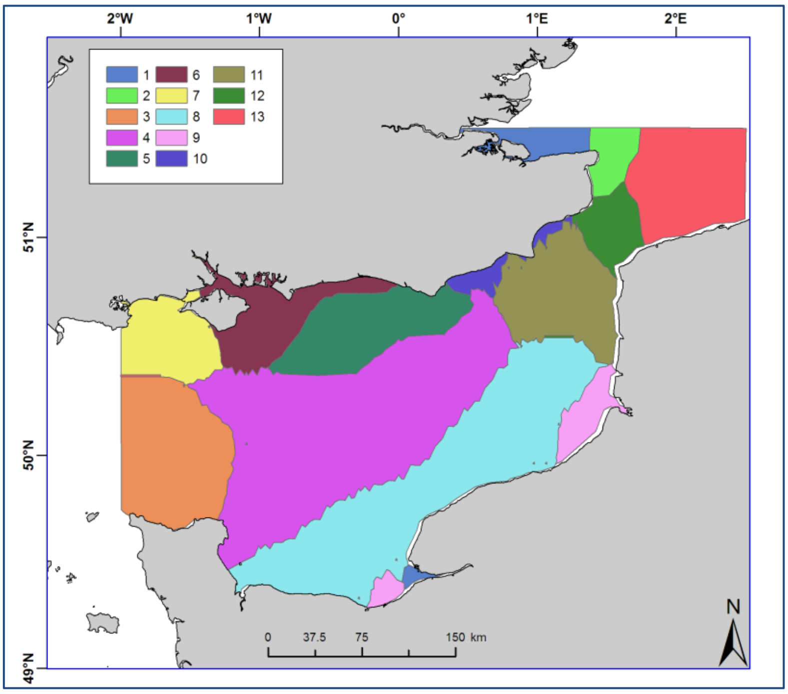
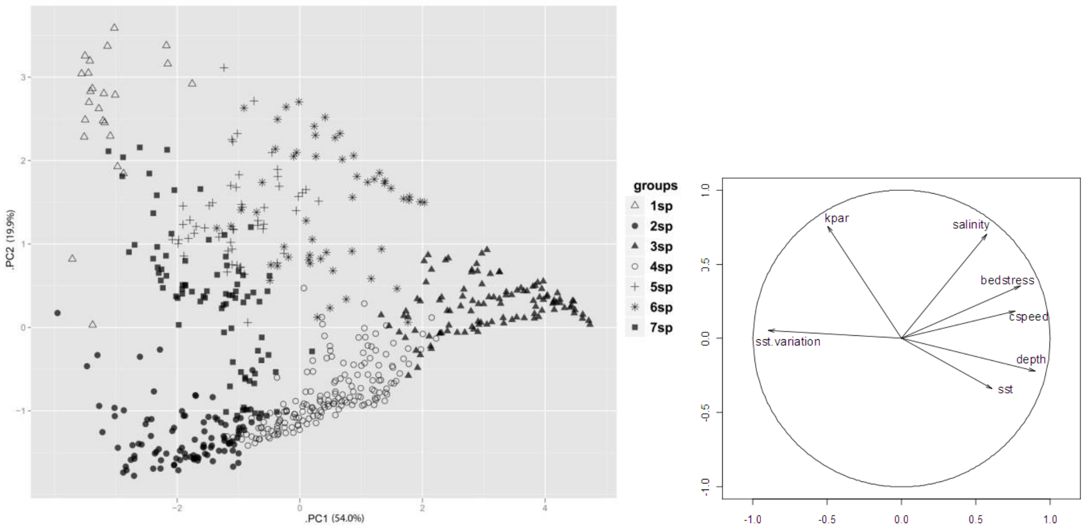
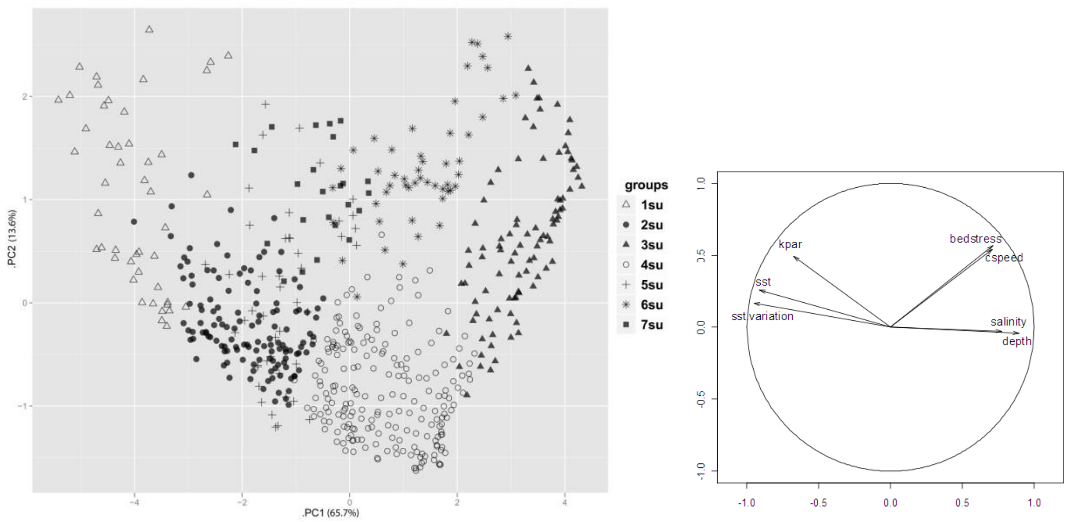


Figure 15: Multi-seasonal typology with 13 water types presented

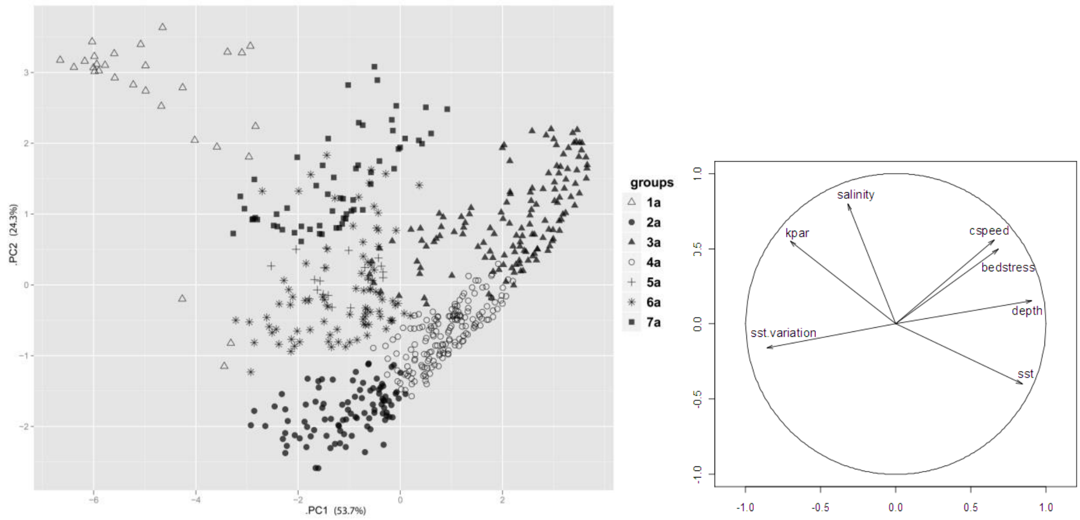
a)Spring



b) Summer



c) Autumn



d) Winter

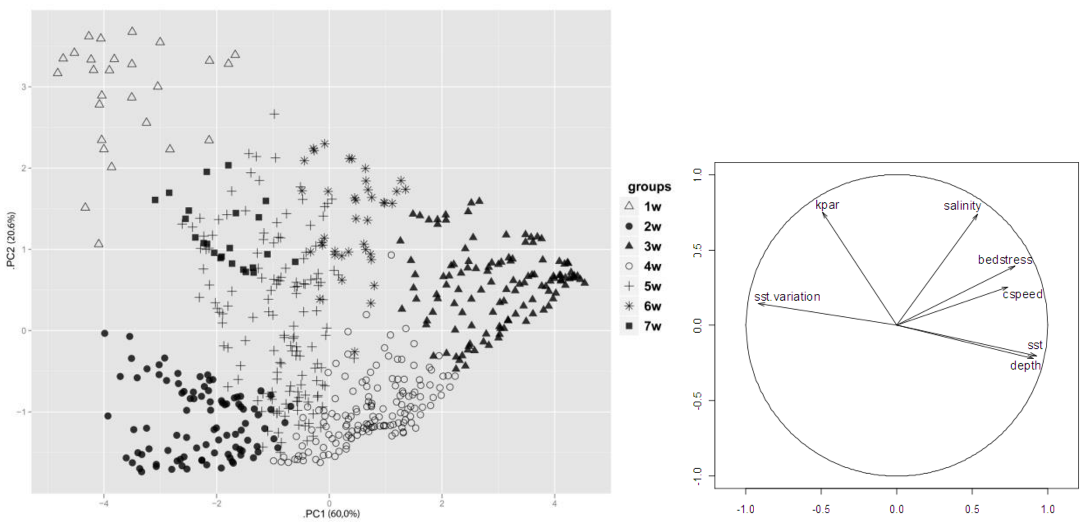


Figure 16 : PCA results for the four seasonal typologies

2.3.2. The biological validation

ANOSIM analyses were conducted for each available biological datasets separately. A significant result meant that the tested biological compositions were different for each defined environmental classes. For the pelagic fishes against the autumnal typology, the ANOSIM statistic, the R value was 0,28 and significant with $p<0.001$. For the winter typology, the ANOSIM were significant for each of the three biological datasets with R equal to 0,27 for the phytoplankton, 0.20 for the zooplankton and 0,26 for the pelagic fishes. This was further confirmed and detailed for each class with the discriminant analyses results (Table 6). For each dataset analysis, a significant Bartlett test indicated that the biological communities were different across classes. The contingency tables (Table 6) showed the quality of the discrimination of each separate class. High recall values were observed for the phytoplankton and zooplankton communities, which were found to be very coherent with the proposed pelagic typologies. The recall is equal to 1 for all classes for the phytoplankton data and do not go below 0.5 for the zooplankton. Moreover, the few misclassified observations are always placed in adjacent classes. Concerning the pelagic fishes the recall ranged from 0.75 to 0.94 and the weakest score corresponds to the class 3w containing 4 observations only out of which 1 was placed in the adjacent class 4w. For the autumnal typology, the available dataset was much larger and variable than those available for the winter validation. Still, the recall ranged from 0,65 to 0.95 which was an unexpectedly good result. Globally the discriminant analyses results had high recall scores and validated the proposed pelagic typologies in winter and autumn.

Table 6: Contingency tables from the discriminant analyses. The rows correspond to observed classes based on the observation spatial location, and the columns to the predicted classes based on the modeled relationships between the biological assemblages structure and the proposed classification. The diagonal indicates the number of observations correctly allocated. The RECALL line shows the percentage of good reallocation (ratio between the points well reclassified compared to the total number of points in the class). Table 3a represents the winter typology tested against the phytoplankton data, Table 3b represents the winter typology tested against the zooplankton data, table 3c is the winter typology tested against the pelagic fish data and the table 3d represents the autumnal typology tested against the pelagic fish data.

a) Predicted/Real	**2w**	**3w**	**4w**	**5w**	**6w**	**7w**
2w	**30**	0	0	0	0	0
3w	0	**6**	0	0	0	0
4w	0	0	**26**	0	0	0
5w	0	0	0	**17**	0	0
6w	0	0	0	0	**3**	0
7w	0	0	0	0	0	**6**
RECALL	1	1	1	1	1	1

b) Predicted/Real	2w	4w	5w	6w	7w
2w	**9**	0	1	0	0
4w	1	**3**	0	0	0
5w	0	0	**1**	0	0
6w	0	0	0	**1**	0
7w	0	0	0	0	**2**
RECALL	0.9	1	0.5	1	1

c) Predicted/Real	2w	3w	4w	5w	6w	7w
2w	**16**	0	0	0	0	0
3w	0	**3**	1	0	0	0
4w	2	1	**16**	1	1	0
5w	2	0	0	**7**	0	1
6w	0	0	0	0	**4**	0
7w	1	0	0	2	0	**4**
RECALL	0.76	0.75	0.94	0.78	0.8	0.8

d) Predicted/Real	2a	3a	4a	6a	7a
2a	**28**	0	3	4	0
3a	0	**19**	0	0	0
4a	7	1	**42**	1	0
6a	7	0	1	**15**	2
7a	1	0	0	0	**7**
RECALL	0.65	0.95	0.91	0.75	0.78

2.4. Discussion

A pelagic typology of the entire French coast was produced by Gaillard-Rocher (2012) to answer the need of the Marine Strategy Framework Directive implementation in the French waters. This typology used the same parameters for the Channel and the Bay of Biscay. Although this first attempt offered a good methodological coherence at the national level, it only resulted in few water classes in the EEC as the strong tidal features of this particular area was poorly described. The study presented here goes a step further, with more detailed and appropriate parameters at the scale of the eastern English Channel, and with a biological validation of the typology. Constructed from both modeled and field data, the proposed seasonal and multi-seasonal typologies were believed to be an appropriate representation of the water masses evolution along a year cycle at the scale of the eastern English Channel.

The seasonal and multi-seasonal typologies reflected the major hydrological characteristics of the eastern English Channel. Some water masses appeared to be very stable during the year, certainly due to the influence of strong structuring physical parameters such as tide driven seabed stress or depth, which were constant throughout the year. The central EEC water body (4sp, 4su, 4a and 4w) was the deepest and looked stable over the year even though its extent could vary. Similarly, the location and extent of water body 3 (sp, su, a and w), which was the one subject to the strongest currents, varied little over seasons. This was also illustrated by the low variation in depth, bedstress and SST variation values in these two water types over the four seasons. These two offshore water bodies also had a less important SST intra-annual variation, their water masses were more directly influenced by oceanic waters from the western English Channel and the Celtic Sea than by costal and terrestrial factors, and in consequence had more stable oceanic parameters. Coastal water types seemed more variable, certainly due to the large influence of season-dependent parameters such as temperature or turbidity. For example, water body 1 represented estuarine waters but although this type contained the Thames and the Seine estuaries all year round, it also contained the Somme estuary and its surrounding area during summer. In summer, the characteristics of this area were more similar to that of the two other estuaries than in the other seasons with higher surface temperature and KPAR. But even if coastal water bodies are more variable it is noticeable that the water along the English coasts and the Dover Strait are less stable than the French coasts which is represented by the water bodies 2 in the four seasons. This is also highlighted by the multi-seasonal typology which exhibited smaller areas close to the English coast and the Dover strait reflecting the stronger variability of these water types along the year.

Accounting for the temporal variability of the water types for management purpose or conservation areas design is not always feasible. Therefore, besides the seasonal typologies, it appeared important to produce this “multi-seasonal” typology, which would integrate the seasonal variability and may be used as an appropriate “all year round” typology of the water types in the eastern English Channel.

The validation of the typologies with biological datasets is a crucial step to verify that the water bodies defined may be used as pelagic biodiversity surrogates for management or conservation purposes (Grantham et al., 2011; Gregr et al., 2012; Snelder et al., 2005) and unfortunately this validation step is almost never done. In the study area, the UK SeaMap project (Connor *et al.*, 2006) constituted a first attempt at defining a pelagic typology at the UK water scale and they validated their seasonal water types with phytoplankton data. However, their validation exercise was built on annual mean distribution data of only 5 plankton indicators, which limited the scope of their conclusions. In this study, the seasonal resolution of the biological datasets used for the validation was consistent with the pelagic typologies. The biological characterization showed that the different water types tested were significantly different from a biological point of view and were also able to discriminate between different biological communities in autumn and winter. Although these taxonomically rich communities are known to be patchy and very variable at a spatial and temporal scale often smaller than the proposed water types or the season (Carpentier et al., 2009; Koubbi et al., 2006), the results of the discriminant analysis confirmed as expected that phytoplankton and zooplankton are very dependent of the water masses and that the proposed typologies were relevant to these taxa.

Even though discrimination results had lower recall values for pelagic fishes than for the other taxa, the results showed that the proposed water masses' typologies were also reasonable predictors for the pelagic fish assemblages which was a comforting result considering the mobility and large distribution pattern of these organisms. It would have been useful to develop this biological characterization for spring and summer but no relevant data were available on these periods for the pelagic biological compartment at the scale of the entire EEC. Moreover, no relevant biological dataset was available all year round to enable the evaluation of the multi-seasonal typology. Finally, besides plankton and fishes data, this study would be complete with a validation with large megavertebrates such as cetaceans or sharks.

The proposed typologies are a first attempt to produce a seasonally pelagic typology at the scale of the eastern English Channel with a EUNIS-like approach. Our results may inform managers responsible of the enforcement of the EU Water or Marine Strategy Framework Directives (EC, 2008), and support marine spatial planning in the eastern English Channel. However, it still has to be stressed that even though the produced typology may be used as a preliminary pelagic biodiversity surrogate, further work needs to be done to include explicitly habitats with essential functions such as nurseries or spawning grounds in any management or conservation plan. Therefore, the present maps may be representative of the pelagic biodiversity at the EEC scale but does not comprehensively reflect the whole functional diversity of the area.

3. Chapter Three

Habitat Targets using the Species Area Relationships with different biological datasets and typologies.

Abstract

One of the key steps of systematic conservation planning is to produce a list of conservation features such as species, habitats and ecological processes and then setting quantitative targets for the minimum amount of each feature which needs to be protected from human pressure. Target-setting approach for habitats often relies on policy-driven targets which have been criticized for their lack of relevance. Species Areas Relationship (SAR) is increasingly used to set more scientifically defensible habitat targets. However, habitat targets based on SAR were shown to be influenced by various parameters such as sample size or the level of habitat classification used. In the present study, two benthic species datasets from the eastern English Channel, sampled with different gears and representing different biologic compartments, were used to estimate the influence of the biological dataset used on the produced SAR habitat targets. Original or interpolated observations were also used to explore the influence of data density and sparseness on the results. Finally, two habitat typologies were used to explore how the choice of habitat description could influence marine protected areas (MPA) networks design. We found that besides the sample size, the quality of these samples and their relevance to the considered habitat was significant. The use of interpolated data was also an effective and precautionary way to circumvent the spatial variations of sampling effort. Moreover, the use of different typologies to design PA networks led to different patterns of priority areas so a MPA network solely designed with benthic habitats as biodiversity surrogates may not be fully representative of the whole biodiversity of a marine region. Using the scientifically defensible SAR habitat-targets, combining different datasets depending of their representativeness of habitats and taking into consideration different type of biodiversity surrogates may constitute a real improvement in MPA network design.

Keywords: English Channel, conservation targets, marine protected areas, species-area relationships, habitat typology.

3.1. Introduction

Marine Protected Areas (MPAs) are increasingly used worldwide to manage the use of marine resources and mitigate impacts on biodiversity and ecological processes (Halpern and Warner, 2002; Smith et al., 2009; Wood et al., 2008). Systematic conservation planning is a commonly used approach for prioritising where new MPAs should be located, as it allows planners to identify MPA networks that meet conservation goals whilst minimizing adverse socio-economic impacts on stakeholders (Margules and Pressey, 2000). One key step in the conservation planning process is to set appropriate conservation targets (Carwardine et al., 2009; Knight et al., 2006). Planners sometimes use broad conservation targets that reflect political or institutional goals, such as conserving 10% of each habitat (CBD, 2004) but these policy-driven targets have been heavily criticized for their ecological irrelevance (Svancara et al., 2005). Instead, it is much better to use quantitative or data-driven targets to produce a more effective conservation planning process (Carwardine et al., 2009; Rondinini and Chiozza, 2010).

Systematic conservation planning often focuses on protecting species, communities, ecological processes or habitat types (Pressey et al., 2003; Rondinini and Chiozza, 2010). Setting species targets often uses techniques based on population viability estimates (Justus et al., 2008; Rondinini et al., 2006), but targets for habitat types are commonly based on expert opinion or policy recommendations (Rondinini and Chiozza, 2010; Smith et al., 2009). So, to develop more scientifically defensible habitat-specific targets, researchers developed a method based on the species-area relationship (SAR) (Desmet and Cowling, 2004). This process uses survey data to determine the proportion of each habitat type needed to conserve a user-specified percentage of the associated species (Desmet and Cowling, 2004). This methodology was applied in South Africa for vegetation types and for benthic habitats in English waters as part of the UK Government's Marine Conservation Zone (MCZ) project (JNCC and Natural England, 2010; Rondinini, 2011a). In addition, a study by Metcalfe et al (2012) used the SAR-based approach to develop benthic habitat targets for the eastern English Channel (EEC).

This research by Metcalfe et al (2012) used an extensive macrobenthic dataset from the EEC and EUNIS benthic habitat types and found that habitat targets based on SAR could be influenced by the sample size, the choice of richness estimator and the level of habitat classification. Based on this, they argued that more research was needed on how to minimize the impacts of data quality on target setting. Here, the present study builds on this work by investigating the impact of sampling on the definition of habitat targets in the eastern English Channel. In addition, the impact of the biodiversity surrogate on setting targets was investigated by also calculating targets for pelagic habitat types. So, the targets produced by Metcalfe et al (2012) for benthic habitats based on grab survey macrofauna presence/ absence data were compared with: (1) benthic target habitats based on raw trawl survey megafauna presence/absence data, (2) benthic target habitats based on interpolated trawl survey megafauna presence/absence data and; (3) pelagic target habitats based on interpolated trawl survey demersal fish presence/absence data. Then, the Marxan conservation planning software was used to compare the results of using benthic and pelagic habitat maps and targets in MPA network design.

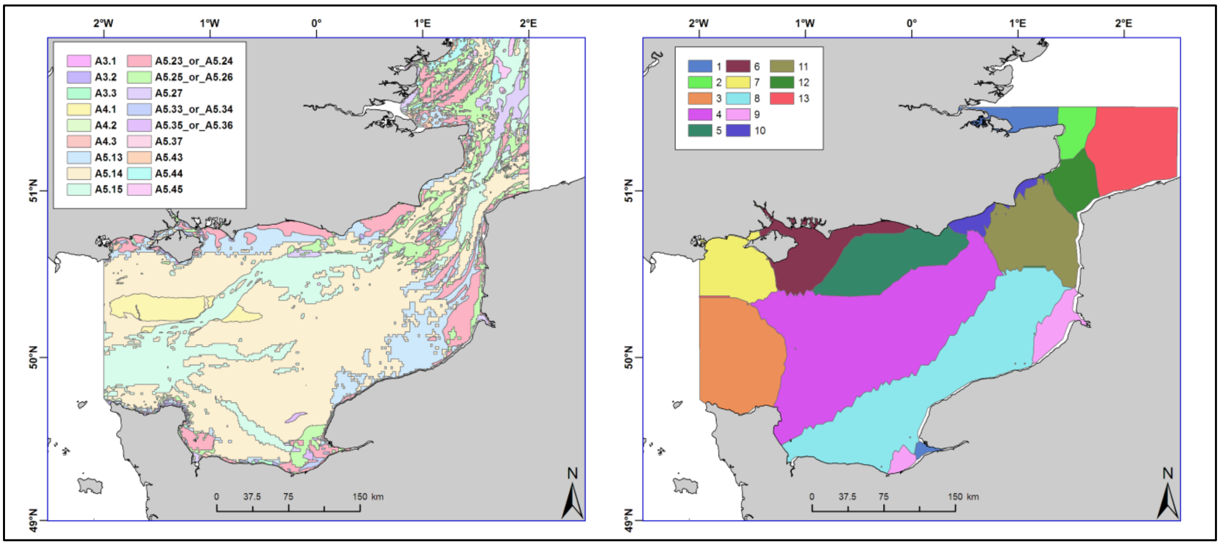


Figure 17: The benthic EUNIS typology (From Coggan and Diesing (2010) on the left) and the pelagic typology from Delavenne et al (in press), on the right)

3.2. Methods

3.2.1. Habitat maps

Two habitat typologies were used in this study (Figure 17). The benthic habitat map which is described in the part 1.4 was modeled from physical characteristics by Coggan and Diesing (2011). This is based on the European Nature Information System (EUNIS) habitat classification hierarchy (European Environment Agency, 2006). Only habitats which can be represented by environmental data are represented, the rocky habitats are therefore modeled to level 3 in the EUNIS hierarchy and sediment habitats which are better known can be modeled to a finer level (level 4) The pelagic habitat map is described in the Chapter Two, it was computed from the classification of several hydrological characteristics of the water column across all and contained 13 habitat types (Delavenne et al., *subm*).

3.2.2. Species data

Three biological datasets representing three distinct biological compartments were used for the different target setting analyses: macrobenthic invertebrates, megabenthic invertebrates and demersal fish data (Figure 18). The macrobenthic species data were collected during surveys carried out between 1985 and 2007, providing data from 1314 sampling stations and 731 species (Metcalfe et al., 2012). These samples were collected with a Hamon Grab with the exception of 16 stations which used a Van Veen grab. These types of gear sample seabed surfaces ranging from 0.1 to 0.5 m². Macrobenthic species were sorted and identified after being sieved on a 2 mm mesh (Carpentier et al., 2009). Both the collection (often requiring dedicated survey) and the processing of the samples (very long sorting time and high level of taxonomic expertise required) make such dataset expensive and difficult to obtain.

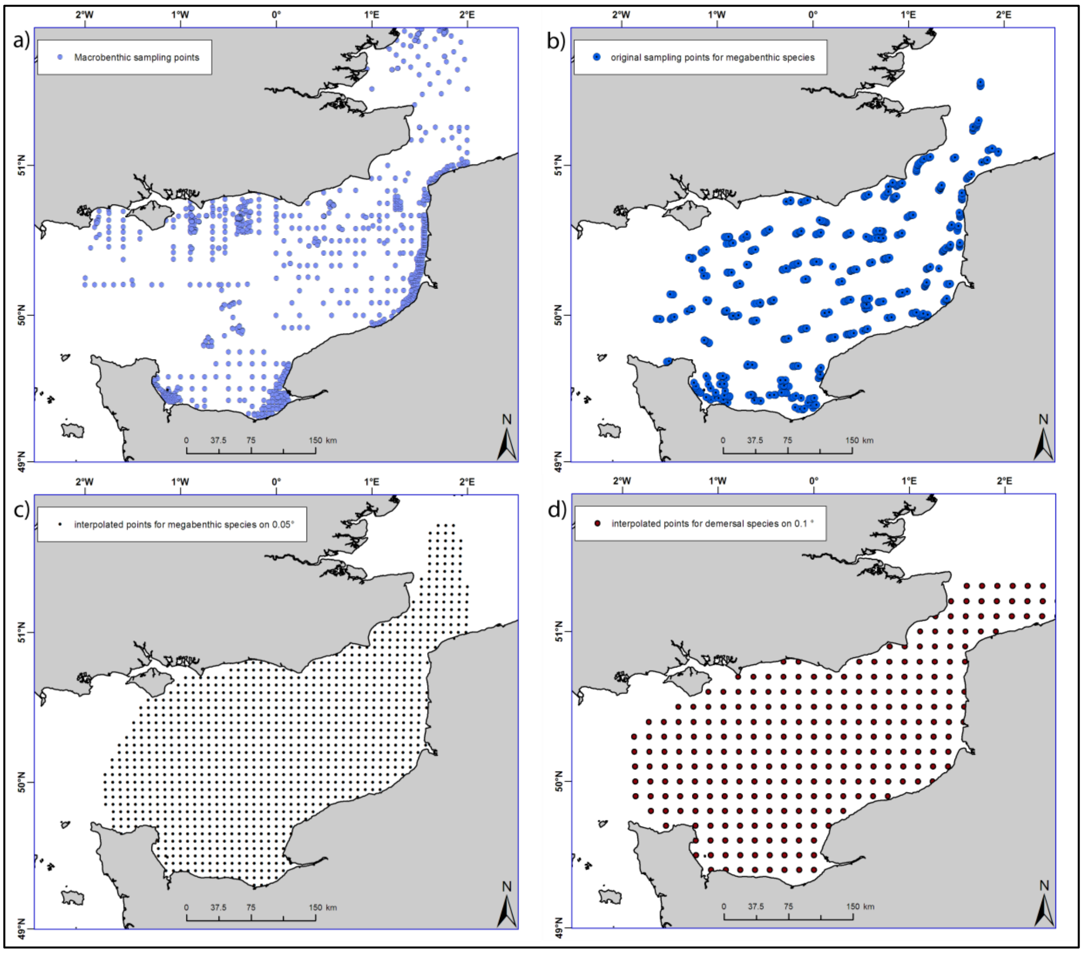


Figure 18: Biological datasets, a) the macrobenthos sampling points, b) the megabenthos sampling points, c) the interpolated megabenthos points, d) the interpolated demersal fish points

The megabenthos data originated from two scientific trawl surveys: the Channel Ground Fish Survey (CGFS) and the International Bottom Trawl survey (IBTS). The CGFS has operated every October in the EEC since 1988 and the International Bottom Trawl survey (IBTS) normally operates in January in the North Sea, but its coverage has been extended to the EEC since 2008. The megabenthic invertebrates (here > 20 mm) were collected opportunistically in the bottom trawl with sampled surface ranging from 20 000 to 30 000 m². The taxonomic identification was normally done on board and required less expertise than would grab samples which made this dataset relatively quick and cheap to acquire. The data used in this study consisted of binary (presence/absence) records for 30 species, recorded from 2006 to 2011 for the CGFS and 2008 to 2011 for IBTS .

The sampling gear mechanisms and sampled surfaces were considerably different between the two datasets. This resulted in important difference in the main collected taxa with a majority of annelids in the macrobenthic species and a majority of crustaceans in the megabentic species. Although grabs are widely used in exhaustive endofaunal benthic studies, the megabenthic species collected in trawls could represent habitats at a much larger scale and be much more integrative than grab data (Ellis and Rogers, 2004). Moreover, they might be much more effective on harder substrates which are not easily sampled with grabs.

Demersal species live and feed on or near the sea bottom which make them equally dependent on the water column and the seabed. Eleven species of demersal fish and one cephalopod were collected during the CGFS survey from 1988 to 2009 and were available as abundance indices: Cod, whiting, red mullet, red gurnard, tope, greater spotted dogfish, lesser spotted dogfish, black bream, spurdog, horse mackerel, poor cod and cuttlefish were chosen as they were believed to be equally illustrative of the hydrological and seabed habitats of the EEC.

3.2.3. Sensitivity analyses

Two complementary sensitivity analyses were run to evaluate the effects of using different biological datasets and habitat typologies on the target-setting results. First, to study the influence of biological datasets on target setting, the benthic habitat typology was coupled to three different datasets: (i) macrobenthos presence/absence data (Metcalfe et al, 2012), (ii) megabenthos presence/absence data (raw data from the survey stations) and (iii) megabenthos probability of presence. Megabenthos probabilities of presence were calculated from megabenthos presence/absence using indicator kriging on a 0.05° latitude mesh grid (Figure 22). Kriging is a geostatistical method to estimate the values of a property of interest (e.g. species density or probability of presence), at non sampled locations. Kriging differs from other interpolators because it uses a model of the spatial auto-correlation pattern of the variable of interest. The interpolation allowed us to produce regularly spaced probability of presence data, but the SAR method for setting targets requires binary observations so this kriged data were converted into presence/absence data for the same regular grid. The "Minimize Difference Threshold" (MDT) transformation was used to find the transformation threshold to convert the probabilities of presence to 0 or 1. The MDT threshold criterion avoids omission errors (false negatives) and is recommended as a precautionary conservation approach (Jiménez-Valverde and Lobo, 2007; Rondinini et al., 2006).

Second, to investigate the influence of the habitat typologies on target setting, we investigated the results of using both benthic and pelagic habitats in the target setting process. We considered the distribution of 12 demersal species and to avoid sampling biases in the spatial coverage across habitats we used interpolated densities on a 0.1° resolution grid which corresponded to the average resolution of the yearly CGFS surveys, in contrast to the 0.05° resolution used for the megabenthos data, which corresponded to the survey resolution when several years were pooled. This resolution choice was done according to the ecology of the two types of species, as megabenthic species have a patchier distribution, more limited mobility and inter-annual variability, and needed to be mapped at a finer spatial scale. Continuous interpolated densities had to be transformed because the SAR method for setting targets requires binary observations. The straightforward transformation into presence/absence was not appropriate due to the smoothing effect of the interpolation which resulted into many observations with very low density values and few absences. Instead, the median of the data distribution was used as a threshold to recode densities into binary data effectively highlighting the most suitable habitat (where the density values were the highest) for each species.

3.2.4. Calculating habitat targets

We calculated habitat targets using the approach developed by Desmet and Cowling (2004). This method is based on the well-established relationship between habitat area and the number of species that can be supported in this habitat. The traditional power model (equation 2) is transformed to estimate the proportion of habitat required to represent a given percentage of species (equation 3)

(2) $S=cA^{z}$

(3) $\text{Log } A' = \text{Log } S'/z$

S' and A' are respectively the proportion of species and area (Desmet and Cowling, 2004; Rondinini and Chiozza, 2010). z describes the slope of the power function, which measures how species richness increases with area (Rosenzweig, 1995). C is a constant scaling factor and can be ignored when comparing proportions or percentages of area and species (Desmet and Cowling 2004, Rondinini and Chiozza 2010). Using this relationship, it is possible to set conservation targets for habitats. The method involves (i) estimating the z value of the species area relationship for a given habitat, (ii) calculating the percentage of the area required to represent a given percentage of species and, (iii) multiplying this percentage value by the total habitat area.

The z value is calculated for each habitat type using equation (4) (Desmet and Cowling, 2004; Neigel, 2003).

(4) $z= (y_2-y_1)/(x_2-x_1)$

where y_2 is the log-transformed total number of species in a habitat type; y_1 is the log-transformed average number of species per survey sample; x_2 is the log-transformed total area of habitat class; x_1 is the log-transformed average sampled area. y_1, x_2 and x_1 are derived from the species dataset. x_1, the sampled area, was estimated from the trawled surface, which is known for each station. To calculate the z value the only unknown part is y_2, an estimation of the total number of species found in a given habitat (Desmet and Cowling 2004). There is no consensus on the best estimator to use when calculating species richness. However, there is agreement that the bootstrap estimator is the most conservative (Colwell and Coddington, 1994; Desmet and Cowling, 2004; Metcalfe et al., 2012). Since we used habitat targets from Metcalfe et al (2012) calculated with bootstrap estimator, the same method was applied to all the other habitat targets set here. Moreover, Metcalfe et al (2012) showed that the boostrap estimator was the one requiring the fewest number of samples to produce stable targets. The z value was then used to calculate the proportion of area required to represent 80% of species for each habitat type with more than three sampling points. The 80% value was chosen to ensure consistency with Metcalfe et al (2012) and with regional conservation projects in the United Kingdom (JNCC and Natural England, 2010).

Depending on the biological datasets spatial cover, some habitat types did not have corresponding observation. In the case of the interpolated megabenthos data, targets were only calculated for those habitats where original (unprocessed) megabenthos data were available, to ensure the ecological relevance of the results and avoid using extrapolated data. However, the study of the effect of these targets at the scale of the EEC required obtaining targets for each habitat types. Desmet and Cowling (2004) extrapolated their z values to other regions without survey data, based on “observed relationship between the calculated z values […] and some remotely measurable land-class properties”. Here, we did not extrapolate the z value but instead extended some of the available computed targets to those habitats with no observation, based on the similarity of their habitat characteristics: for benthic typology, targets for non-sampled regions were defined using those of the upper EUNIS levels (for the A3 and A.4 habitat types). When the upper level target was not appropriate, we used the most conservative targets calculated for similar habitats types at the same EUNIS level. For example, we extended the particularly high A5.1x (coarse sediment) habitat targets to the A5.4x (mixed sediments) habitat that were found at similar depths. For pelagic habitats, we also extrapolated targets for non-sampled habitat classes based both on habitat characteristics or geographic proximity.

3.2.5. Determining the effects of using different habitat targets in Marxan analyses

Marxan is one of the most widely used conservation planning decision-support tool (Ball et al., 2009) and uses a minimum set approach to identify portfolios of planning units that achieves conservation targets at near-minimal cost (Ball and Possingham, 2000). Marxan calculates the cost of a portfolio based on an objective function consisting of: (i) the combined cost of the planning units in the portfolio; (ii) a penalty set whenever a target is not met, and; (iii) a spatial constraint cost reflecting the portfolio's fragmentation level (Possingham et al 2000). The spatial constraint cost is based on the boundary length of the portfolio, as fragmented portfolios have more exposed edges (Ball and Possingham, 2000; Possingham et al., 2000).

A Marxan analysis involves running the software a number of times and producing a near-optimal, but often different, portfolio each time. It then identifies the best portfolio as the one with the lowest cost and produces a selection frequency output by counting the number of times each planning unit appeared in the different portfolios (Ball et al., 2009). Here, we ran six different scenarios in Marxan. Scenarios A1 to A3 were based on the benthic habitat types. Scenario A1 used targets calculated from the macrobenthos dataset (The results were calculated in Metcalfe et al (2012)), scenario A2 used targets calculated with the original (unprocessed) megabenthos dataset and scenario A3 used targets calculated from the interpolated megabenthos. The habitat targets for the macrobenthos, the original megabenthos data and the interpolated megabenthos data are presented in tables 7, 8 and 9 respectively

Scenarios B1 to B3 were based on the habitat targets calculated for the demersal dataset, with B1 using the benthic habitat as the features to conserve, scenario B2 used both benthic and pelagic habitats and scenario B3 just used the pelagic habitat types. To be able to run these scenarios, we first, produced the planning unit theme by creating a series of 10 km edge grid squares using the Repeating Shapes extension in ArcView 3.2 and then, we calculated the area of each habitat class in each planning unit. Some of the planning units overlapped the coastline and so less of their area fell within the planning region. For these overlapping planning units, we calculated their area within the planning region by clipping them with the coastline boundary and used these area values as the basis of the planning unit costs. Thus, planning units that only contained a small amount of the English Channel tended to contain less of each habitat class but also had a lower cost.

After running the six scenarios, the Marxan outputs (best portfolio and selection frequency output) were compared. First, to investigate the impact of the target values and used typologies, we calculated an index of difference in spatial pattern between pair of selection frequency outputs. The proportion of selection frequency in each planning unit was calculated such as the sum in all planning units was 1.0. Between two maps, the absolute differences per planning unit were calculated, summed for the entire study area and then divided by two. This provides an index of spatial difference varying from 0 (identical spatial pattern) to one (maximal difference in spatial pattern) (Lee et al., 2010). A low index indicates similar spatial patterns and conversely, high value indicates marked differences. Three indices were calculated between the three selection frequency outputs produced with the benthic typology (scenarios A1 to A3) and another three were calculated with the demersal species as biological dataset (scenarios B1 to B3). This comparison was done to quantify the effect of data or typology change in the resulting conservation planning exercise. Furthermore, the best portfolios areas were calculated on 10 runs to produce a mean value for each scenario.

3.3. Results

For the macrobenthos dataset, percentage targets range from 6.25% for low-energy circalittoral rock to 24.65% for infralittoral fine sand or muddy sand (Table 7). For the original megabenthos dataset the minimum target is 6.93% for the deep circalittoral sand and the maximum percentage target is 16.4% for infralittoral coarse sediment (Table 8).Concerning the interpolated megabenthos data, percentage targets range from 6.5% also for the infralittoral sandy mud or fine mud to 23.06% for the deep circalittoral sand (Table 9).

The same process was applied to the targets derived from the demersal species dataset (Tables 10, 11 and 12). Habitat targets were calculated when sampling point were available (Tables 11 and 16). For the benthic habitat types, missing values were estimated based on the closest habitat in the EUNIS hierarchy with an available target (Table 10). A habitat target was calculated at the A4 level (9.36%) and applied to the A4.x and A3.x habitats. The A5.3x habitats got the A5.2x targets, and the A5.4x habitats (mixed sediment) were allotted the A5.1x habitats (coarse sediment) targets, based on depth level considerations. In spite of the much lower number of species, targets obtained for demersal species were of the same order than benthic invertebrate derived targets.

Table 7: Habitat-specific survey data, estimation of the total number of species, z-values and proportion (%) of habitat necessary as conservation target for each considered EUNIS habitat type and the macrobenthos dataset (after Metcalfe et al, 2012).

	EUNIS habitat description	Area (km²) of habitat	Number of surveys station	Average area (m²) of samples	Average number of species per sample	Total number of observed species	Bootstrap estimator	z-value	target
A3.1	High energy infralittoral rock	29							
A3.2	Moderate energy infralittoral rock	196							
A3.3	Low energy infralittoral rock	116	11	0.5	10	60	74	0.104	11.68
A4	Circalitorral Rock	1448							
A4.1	High energy circalittoral rock	1149							
A4.2	Moderate energy circalittoral rock	191							
A4.3	Low energy circalittoral rock	108	5	0.5	38	142	178	0.080	6.25
A5.13	Infralittoral coarse sediment	4092	263	0.2	46	971	1079	0.133	18.65
A5.14	Circalittoral coarse sediment	18934	373	0.31	59	1326	1470	0.129	17.84
A5.15	Deep circalittoral coarse sediment	6863	89	0.25	49.7	825	950	0.123	16.38
A5.23 or A5.24	Infralittoral fine sand or muddy sand	3701	288	0.45	18	590	684	0.159	24.65
A5.25 or A5.26	Circalittoral fine sand or muddy sand	3046	165	0.45	18	454	539	0.150	22.63
A5.27	Deep circalittoral sand	886	16	0.28	14	128	160	0.111	13.48
A5.33 or A5.34	Infralittoral sandy mud or fine mud	196	17	0.49	18	139	170	0.113	13.97
A5.35 or A5.36	Circalittoral sandy mud or fine mud	134	11	0.46	26	131	158	0.093	8.98
A4.43	Infralittoral mixed sediments	225							
A5.44	Circalittoral mixed sediments	477	50	0.3	25	245	287	0.115	14.41
A5.45	Deep circalittoral mixed sediments	198	14	0.11	25	164	202	0.098	10.27

Table 8: Habitat-specific survey data, estimation of the total number of species, z-values and proportion (%) of habitat necessary as conservation target for each considered EUNIS habitat type and the original megabenthos dataset.

	EUNIS habitat description	Area (km²) of habitat	Number of surveys station	Average area (**km²**) of samples	Average number of species per sample	Total number of observed species	Bootstrap estimator	z-value	target
A3.1	High energy infralittoral rock	29							
A3.2	Moderate energy infralittoral rock	196							
A3.3	lowenergy infralittoral rock	116							
A4	Circalittoral Rock	1448	6	0.023	5	16	24	0.137	8.06
A4.1	High energycircalittoral rock	1149	4	0.02	5	10	16	0.117	7.73
A4.2	Moderate energy circalittoral rock	191							
A4.3	Low energy circalittoral rock	108	2						
A5.13	Infralittoral coarse sediment	4092	89	0.028	10	29	29	0.083	16.40
A5.14	Circalittoral coarse sediment	18934	241	0.028	9	28	29	0.097	13.85
A5.15	Deep Circalittoral coarse sediment	6863	101	0.028	6	25	29	0.140	9.31
A5.23 or A5.24	Infralittoral fine sand or muddy sand	3701	61	0.028	8	27	28	0.110	12.08
A5.25 or A5.26	Circalittoral fine sand or muddy sand	3046	51	0.029	10	24	28	0.099	12.93
A5.27	Deep circalittoral sand	886	17	0.030	6	13	21	0.147	6.93
A5.33 or A5.34	Infralittoral sandy mud or fine mud	196	4	0.028	8	23	30	0.151	8.35
A5.35 or A5.36	Circalittoral sandy mud or fine mud	134	9	0.028					
A4.43	Infralittoral mixed sediments	225							
A5.44	Circalittoral mixed sediments	477							
A5.45	Deep circalittoral mixed sediments	198							

Table 9: Habitat-specific survey data, estimation of the total number of species, z-values and proportion (%) of habitat necessary as conservation target for each considered EUNIS habitat type and the kriged megabenthos dataset

	EUNIS habitat description	Area (km^2) of habitat	Number of interpo lated points	Average area **(km^2)** of samples	Average number of species per sample	Total number of observed species	Bootstrap estimator	z-value	target
A3.1	High energy infralittoral rock	29							
A3.2	Moderate energy infralittoral rock	196							
A3.3	Low energy infralittoral rock	116							
A4	Circalittoral Rock	1448	36	0.023	10	28	30	0.100	13.46
A4.1	High energy circalittoral rock	1149	26	0.02	8	15	20.5	0.088	12.32
A4.2	Moderate energy circalittoral rock	191							
A4.3	Low energy circalittoral rock	108							
A5.13	Infralittoral coarse sediment	4092	104	0.028	13	28	28	0.065	20.72
A5.14	Circalittoral coarse sediment	18934	550	0.029	12	28	28	0.066	20.38
A5.15	Deep circalittoral coarse sediment	6863	194	0.028	11	28	28	0.078	17.49
A5.23 or A5.24	Infralittoral fine sand or muddy sand	3701	82	0.028	10	28	28.1	0.091	14.97
A5.25 or A5.26	Circalittoral fine sand or muddy sand	3046	68	0.029	12	28	28	0.070	19.20
A5.27	Deep circalittoral sand	886	17	0.03	16	28	28.6	0.059	23.06
A5.33 or A5.34	Infralittoral sandy mud or fine mud	196	3	0.028	5	8	15	0.124	6.51
A5.35 or A5.36	Circalittoral sandy mud or fine mud	134	4	0.028	9	18	27.3	0.138	8.40
A4.43	Infralittoral mixed sediments	225							
A5.44	Circalittoral mixed sediments	477							
A5.45	Deep circalittoral mixed sediments	198							

Concerning the pelagic typology, 3 habitats did not have sampling points (Table 12). Because of geographic proximity habitat 10 was given habitat 11 target (9.64%) and habitat 2 was allotted habitat 13 target (8.09%). Habitat 1 was attributed habitat 9 target (11.4%), since both are estuarine habitats. Targets were more variable than those found for the benthic EUNIS habitats ranging from 8 to almost 30% of habitat surfaces.

Using all targets defined in table 10 and 12, six conservation planning scenarios were run using Marxan. In scenarios A1 to A3, Marxan was run using the benthic typology and habitat targets derived from the three benthic datasets. The selection frequency outputs resulting from these scenarios were mapped (Figure 19). A spatial difference index was calculated between each pair of outputs and was equal to 0.21 between A1 and A2 selection frequency maps, 0.18 between A1 and A3 maps and 0.15 between A2 and A3 maps highlighting relatively low spatial differences resulting from a change of data to define targets. Figure 20 shows that the best solution areas are quite similar for the A1 and the A3 scenarios (covering respectively 6584 and 6956 km² on average). The best solution area using targets derived from scenario A2 was slightly lower, with an average best solution area of 5428 km², due to lower habitat targets.

The table 10 summarizes the benthic habitat targets calculated or extrapolated from the different biological dataset. For the macrobenthos data, habitats without sampling point were allotted the targets of the closest habitat in the EUNIS classification. The A3.1 and A3.2 habitats received the A3.3 habitat target. Similarly, habitats A4.1 and A4.2 were given the A4.3 target. We also attributed to the infralittoral mixed sediment habitat (A5.43) the habitat target of the circalittoral mixed sediment (A5.44) as it was the closest habitat in the EUNIS hierarchy with an available target. Considering the targets based on original and interpolated megabenthos datasets, for habitats A4.x (EUNIS level 3), the habitat target was calculated at the A4 level (EUNIS level 2) and attributed to the habitats without sampling points (8.06% for original megabenthos and 13.46% for the kriged megabenthos). The same A4 target was also applied for the A3.x habitats because they were the closest in the EUNIS hierarchy. In spite of the difference in sampling density and number of species in the difference datasets used, targets were of the same order of magnitude generally ranging between 10 and 20% of habitat surfaces. Targets were found to be the highest for the macrobenthic dataset (especially on soft sediment) and the interpolated megabenthic dataset (especially on coarse sediment).

Table 10: Conservation targets (% of habitat total surface) for each considered EUNIS habitat type. Extended targets for habitat types with no observation are shown with a *.

	EUNIS habitat description	Macrobenthos	Original megabenthos	Interpolated megabenthos	Demersal species
A3.1	High energy infralittoral rock	11.68 *	8.06 *	13.46 *	9.36 *
A3.2	Moderate energy infralittoral rock	11.68 *	8.06 *	13.46 *	9.36 *
A3.3	Low energy infralittoral rock	11.68	8.06 *	13.46 *	9.36 *
A4	Circalittoral Rock	6.25	8.06	13.46	9.36
A4.1	High energy circalittoral rock	6.25 *	7.73	12.32	9.17
A4.2	Moderate energy circalittoral rock	6.25	8.06 *	13.46 *	9.36 *
A4.3	Low energy circalittoral rock	6.25	8.06 *	11.07	9.36 *
A5.13	Infralittoral coarse sediment	18.65	17.29	20.72	15.12
A5.14	Circalittoral coarse sediment	17.84	16.40	20.34	17.77
A5.15	Deep circalittoral coarse sediment	16.38	13.85	17.42	16.25
A5.23 or A5.24	Infralittoral fine sand or muddy sand	24.65	9.31	14.97	11.36
A5.25 or A5.26	Circalittoral fine sand or muddy sand	22.63	12.08	19.20	12.52
A5.27	Deep circalittoral sand	13.48	12.93	23.06	12.52
A5.33 or A5.34	Infralittoral sandy mud or fine mud	13.97	6.93	6.50	11.36 *
A5.35 or A5.36	Circalittoral sandy mud or fine mud	8.98	8.35	8.40	12.52 *
A4.43	Infralittoral mixed sediments	14.41 *	17.29 *	20.72 *	15.12 *
A5.44	Circalittoral mixed sediments	14.41	16.40 *	20.34 *	17.77 *
A5.45	Deep circalittoral mixed sediments	10.27	13.85 *	17.42 *	16.25 *

Table 11: Habitat-specific survey data, estimation of the total number of species, z-values and proportion (%) of habitat necessary as conservation target for each considered EUNIS habitat type and the demersal dataset

	EUNIS habitat description	Area (km²) of habitat	Number of interpo lated points	Average area **(km²)** of samples	Average number of species per sample	Total number of observed species	Bootstrap estimator	z-value	target
A3.1	High energy infralittoral rock	29							
A3.2	Moderate energy infralittoral rock	196							
A3.3	Low energy infralittoral rock	116							
A4	Circalittoral rock	1448	6	0.02	5	7	11	0.079	9.36
A4.1	High energycircalittoral rock	1149	6	0.02	5	7	11	0.081	9.17
A4.2	Moderate energy circalittoral rock	191							
A4.3	Low energy circalittoral rock	108	2						
A5.13	Infralittoral coarse sediment	4092	27	0.028	6	12	12	0.065	15.12
A5.14	Circalittoral coarse sediment	18934	148	0.029	6	12	12	0.055	17.77
A5.15	Deep circalittoral coarse sediment	6863	49	0.028	6	12	12	0.06	16.25
A5.23 or A5.24	Infralittoral fine sand or muddy sand	3701	25	0.028	5	11	13	0.083	11.36
A5.25 or A5.26	Circalittoral fine sand or muddy sand	3046	17	0.028	5	12	13	0.078	12.52
A5.27	Deep circalittoral sand	886							
A5.33 or A5.34	Infralittoral sandy mud or fine mud	196	1						
A5.35 or A5.36	Circalittoral sandy mud or fine mud	134	1						
A4.43	Infralittoral mixed sediments	225							
A5.44	Circalittoral mixed sediments	477							
A5.45	Deep circalittoral mixed sediments	198							

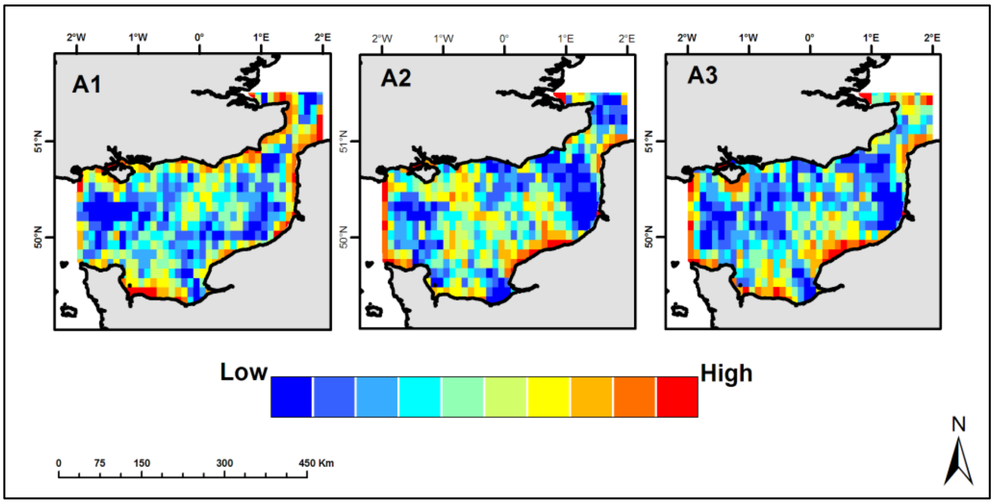


Figure 19: Selection frequency maps using the benthic EUNIS typology and three different benthic datasets. A1) macrofauna dataset, A2) original megafauna dataset, A3) interpolated megafauna dataset

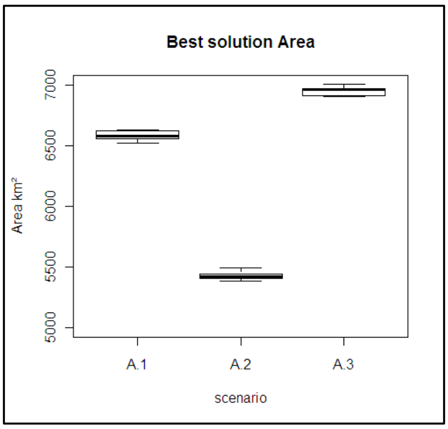


Figure 20: Marxan best solution areasusing the benthic EUNIS typology and three different benthic datasets. A1) macrofauna dataset, A2) original megafauna dataset, A3) interpolated megafauna dataset

Scenarios B1 to B3 were run with habitat targets derived from the demersal species dataset and three sets of habitat typologies (respectively benthic, benthic and pelagic combined, pelagic only). As previously, a spatial difference index was calculated to compare the maps of selection frequency outputs across scenarios (Figure 21). The index was equal to 0.309 when comparing maps resulting from B1 and B3, 0.23 when comparing those resulting from B1 and B2 and 0.258 when comparing maps from B2 and B3. These values revealed low spatial difference resulting from changing habitat typology although these were higher than those found in scenarios A1 to A3. The best solutions areas were relatively similar across the three scenarios (Figure 22), although the area derived from the B3 scenario (5254 km²) was slightly smaller compared to the 5860 and 5873 km² calculated for the B1 and B2 scenarios respectively.

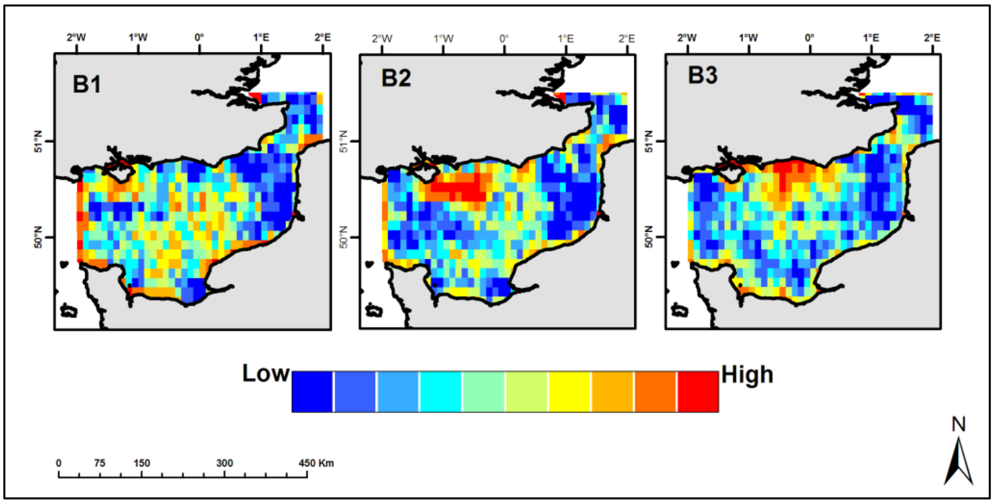


Figure 21: Selection frequency maps using demersal species dataset and B1) the benthic EUNIS typology, B2) benthic EUNIS and pelagic typologies, B3) the pelagic typology.

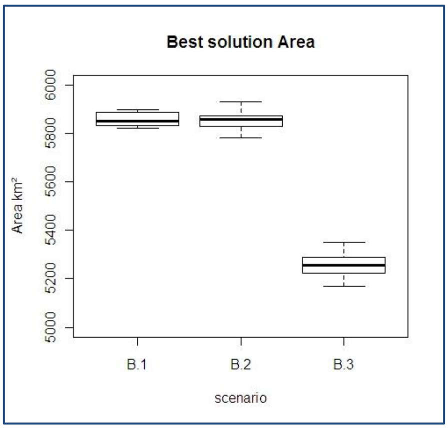


Figure 22: Marxan best solution areas using demersal species dataset and B1) the benthic EUNIS typology, B2) benthic EUNIS and pelagic typologies, B3) the pelagic typology

3.4. Discussion

If various target setting approaches exist in conservation planning (Rondinini and chiozza 2010), the SAR approach is specifically advised in marine conservation planning (Neigel, 2003; Smith et al., 2009). Nevertheless, large inventory data are needed to conduct the process and the quantity and quality of the biological samples have been found to be sources of uncertainties in SAR-based targets (Metcalfe et al., 2012; Rondinini, 2011a; Rondinini and Chiozza, 2010). Metcalfe et al (2012) found that the number of sampling points, the species richness estimator and the level or habitat classification employed all influenced the resulting target. This paper investigated more deeply the effect of the sampling sizes and the quality of the biological data. Moreover, we explored the influence of the varying targets at a spatial scale using both benthic and pelagic habitat types.

Table 12: Habitat-specific survey data, estimation of the total number of species, z-values and proportion (%) of habitat necessary as conservation target for each considered pelagic habitat type and the demersal dataset. Extended targets for habitat types with no observation are shown with a *

	Area (km²) of habitat	Number of surveys station	Average area (m²) of samples	Average number of species per sample	Total number of observed species	Bootstrap estimator	z-value	target
1	767							11.4 *
2	663	2						8.19 *
3	3830	26	0.028	5	9	11	0.066	13.0
4	10330	82	0.028	6	12	12	0.071	13.81
5	2723	23	0.028	8	12	12	0.034	28.74
6	2011	11	0.028	8	11	12	0.036	26.48
7	1818	4	0.028	5	7	11	0.066	11.45
8	7989	65	0.028	5	11	12	0.076	12.36
9	956	9	0.028	4	5	7	0.053	11.40
10	423	2						9.64 *
11	2852	26	0.028	5	11	13	0.098	9.64
12	967	8	0.028	6	8	11	0.052	15.43
13	2926	18	0.028	3	6	9	0.083	8.19

3.4.1. Effects of source data

Evaluating habitat specific targets with the SAR approach relies on the calculation of the z-value. Of the four elements structuring z (equation 4), we only focused in this study on those three terms concomitant to the biological dataset: number of sampling points, total number of observed species and area of samples. We did not do any sensitivity analysis on the estimation of the total number of species found in a given habitat (y_2 in equation 4) and we simply used the bootstrap estimator (Metcalfe et al 2012).

The differences between the habitat targets derived from the three benthic data inventories were investigated. First, it appeared that even if the habitat targets calculated differed, they still laid within the same range of values. It must be noted that in most cases the values obtained in this study were considerably larger than the 10% habitat targets recommended by the CBD (CBD, 2004) and this suggests that the application of the 10% habitat target would fail to represent 80% of the benthic invertebrate species of the EEC, confirming previous studies (JNCC and Natural England, 2010; Metcalfe et al., 2012; Rondinini, 2011a). However, the 10% target from the CBD should be seen as an interim measure in a context where 1.17% of the global ocean is part of a MPA (IUCN, 2010) and where a lot of countries lack data to set SAR-based targets (Metcalfe et al, 2012).

When looking at the sensitivity of habitat targets derived from these three datasets, only focusing on the six habitats for which data were consistently available, we found that targets calculated with the original megafauna dataset (scenario A2) were lower in the coarse sediment (A5.1x) and the sandy (A5.2x) habitats. This may partly be due to consistently less sampling points in the original megafauna dataset (n = 579) than in the macrofauna dataset (n = 1314). The number of sampling points seemed to globally influence the z-value and the target, which bears with the results from previous studies (Metcalfe and al 2012, Rondinini 2011). This effect clearly appeared when comparing the A2 and A3 scenarios. Interpolated data (A3), offering homogeneous density and high number of observation (n = 1073) to represent the same species set, always resulted in higher target than those calculated from irregularly spaced, scarcer original data (A2). The interpolation may be considered as a way to overcome some of the sampling biases resulting from uneven spatial distribution of data points. Moreover, the method used to transform probability of presence in a binary presence/absence variable minimized the risk of an omission error. This approach is believed to be precautionary, and it has been recommended for conservation purpose (Rondinini et al., 2006).

However the size of the dataset used to set the target only partly explained our results. In the case of the infralittoral coarse sediment (A5.13) habitat, there was little difference between calculated targets, although there were almost three times more macrofauna than megafauna observations. Moreover, for the same habitat, the interpolated megafauna dataset produced a higher target (20.72%) with only 104 sampling points, than the macrofauna dataset with 263 sampling points. Another such example was that of the deep circalittoral sand habitat (A5.27) where the three datasets had about the same number of sampling points (16 for the macrofauna and 17 for the two others), while the calculated targets ranged from 12.93 to 23.06%. From those two examples, it appears that the quality, and not only the quantity, of the information and its relevance compared to the considered habitat may have influenced target setting to a large extent. Even if macrobenthos data have more sampling points in the A5.13 habitat, the megafauna observation may be more representative of the biodiversity pattern at the habitat scale

Targets derived from megafauna data (also less numerous and diversified dataset) may be in some cases more relevant than those found using the macrofauna dataset. These two datasets represent different biological compartments with different taxonomic composition as they were sampled with different gears. Sampling surfaces are quite different for each sampling point (more than 20 000 m² for the trawl against 0.25m² for the grab) and the trawl may be more efficient than the grab to integrate the biodiversity of a coarse sediment habitat. Conversely, in sandy habitats (A5. 23 or 24 and A5.25 or 26) where grabs might be more appropriate to represent the biodiversity associated to these types of sediments and there were more macrofauna sampling points, calculated targets were higher than those calculated with the interpolated megafauna dataset.

Bias arising from the number of samples was already known and documented (Metcalfe et al, 2012; Rondinini et al, 2006). Thus, a good knowledge of the ecology of the study area is of primary importance to propose habitat targets based on SAR methods, and the quality of the data or its relevance to the considered habitat may be as important as the quantity of samples. Using macrofauna in soft sediments habitats and trawl sampled megafauna in coarse sediment habitats may represent a more appropriate sampling strategy; the ideal situation would be to combine both dataset to maximize the species representation in any kind of habitat.

3.4.2. A MPA network reflecting which diversity?

The use of habitat targets in Marxan analyses showed that the data sources underlying the target setting process may have a strong influence on the outcomes from decision-support tools. The spatial difference index between the selection frequency maps were not very high but revealed that when considering the same conservation features, slight differences in the attributed targets may lead to different conservation priority areas patterns (Delavenne et al., 2012; Warman et al., 2004). This was illustrated with a higher selection frequency of very coastal, well studied, soft sediment areas when considering the macrofauna. It is uncertain whether this difference reflects a real signal that macrofauna diversity is really higher on very coastal areas zones, or whether it results from more effective and over-sampling of those areas compared to others.

The biological dataset used also influenced the Marxan “Best solution” area needed to meet the agreed targets. The Marxan “best solution” area is decreased by 17%, when comparing the runs using the original unprocessed megafauna targets and those using macrofauna data highliting that the number of observation and species considered may biais the proposed conservation plan. The potential MPA network designed with these macrofauna-based targets was already shown to be more than 50% smaller than those designed for the UK MCZ project (Metcalfe et al., 2012; Rondinini, 2011a). The UK MCZ project used benthos samples of all the UK national waters which represented two to three times more samples for each habitat (Metcalfe et al, 2012). Moreover, the MCZ project used all species recorded in the Marine Recorder Database (n = 4879) which is not limited to macrobenthic species (Rondinini, 2011a, b). These results show the importance of having a good knowledge of the biological dataset used for the SAR target setting process, and of their associated uncertainties. Any data-driven approach will still require expert judgment to weight the conclusions.

The other point of this study was to set and compare SAR-targets for benthic and pelagic habitat typologies, using a demersal species dataset. The maximum number of species (12) in the dataset may be considered as too low for an appropriate target setting exercise, and the conclusions derived from this investigation may be criticized and should therefore be considered with care. However, to produce a fair comparison between the pelagic and the benthic realm, species used had to be equally influenced by the two habitat compartments. A lot of benthic species have a pelagic life stage (Pechenik, 1999) and a lot of fish modify their habitat between their different life stages, to grow up, to feed or to reproduce and these different phases can be whether close to the seabed or in the water column (Gibson, 1997; Loots, 2009) but their influence by water column and seabed are segregated in time. Few species can be said to be truly half-way between pelagic and benthic life styles as most species are preferentially living in the water column or on the seabed. The twelve demersal fish distributions used here represented the only available species set equally influenced by both compartments at the same time. Therefore, this study may be considered as an attempt at comparing the influence of the use of either benthic or pelagic typologies in a management context using an ecologically unbiased set of species. MPA networks only based on benthic biodiversity considerations are not believed to be suited to maintain pelagic biodiversity adequately (Game et al., 2009). Although pelagic habitat descriptions are not yet available in many regions, a pelagic typology map was specifically produced in the EEC for conservation purposes (Delavenne et al., *subm*). Patterns of priority areas resulting from the use of different typologies were more marked than when changing biological dataset. More than just resulting from difference in targets, it is a difference in distribution of the targeted conservation features which was highlighted. This result confirmed that depending on spatial distribution of the ecological compartment under consideration, different conservation prioritization may arise.

One of the habitat-specific SAR targets limitation is that it does not explicitly aim at species persistence because no ecological processes are considered (Rondinini et al., 2006). Using both typologies, benthic and pelagic, to set habitats targets in the same conservation planning approach could help to overcome this problem, by using different habitats layers as proxies for key ecological processes occurring in each compartment. Nevertheless, a MPA network may still require some additional design criteria, such as minimum size requirements and distance optimization, to ensure connectivity and persistence (Claudet et al., 2011; Cowling et al., 1999; Smith et al., 2010; Wilson et al., 2009).

3.4.3. Conclusions

SAR-based targets suffer many caveats and pitfalls as shown in various studies (Desmet and Cowling, 2004; Metcalfe et al., 2012; Rondinini, 2011b). Here, we proposed a way to circumvent the problem of sampling paucity and irregular spacing between habitats using interpolated data. The results also showed that the ecological relevance of the samples, depending on the relation between the chosen sampling gear and strategy and the biological compartment sampled, may have a greater influence on the computed targets than the quantity of observations. Though, the macro and megafauna datasets represent different aspects of the benthic biodiversity, they resulted in targets of similar range and the differences found are believed to originate mostly from the way they were sampled and the adequacy of this sampling strategy with each habitat type. For target setting purpose, we recommend using whichever dataset is most adapted to each habitat and, when possible, to combine both datasets for a same location. In the present case, megafauna data that were opportunistically gathered during routine fisheries survey, were both more relevant in coarser sediment areas and cheaper to produce than were data originating from dedicated benthos survey.

Our analyses also showed that designing MPA networks just taking into account the benthic habitats may not be fully appropriate because it does not adequately represent the pelagic diversity and processes. We recommend that both benthic and pelagic compartments be considered to design priority conservation areas. Finally, it is important to note that the conservation areas derived from this study were based on a static representation of the biodiversity fully omitting dynamic ecological processes. For a more comprehensive conservation planning approach in the eastern English Channel, other parameters such as the persistence and the functional diversity could be implemented, when appropriate data become available (Desmet and Cowling, 2004; Rondinini, 2011a).

4. Chapter Four

A gap analysis of the Marine Protected Area network in the eastern English Channel.

Abstract

To fulfil their international obligations both UK and France are implementing Marine Protected Areas networks in their national waters. This process led to two national MPA networks composed of MPAs with different goals and different design strategies. The EEC is shared between English and French national waters and represents a particular ecosystem between the Atlantic and the North Sea, it can be considered as an eco-region. A range of MPA types are met in the EEC, from European Marine Sites which are the marine version of Natura 2000 sites to national initiatives on both sides. It is interesting to consider these MPAs together as one opportunistic MPA network at the eco-region scale which is much more ecologically relevant than working at the jurisdictional scale, and to realize a gap analysis of this network to explore in what extent marine biodiversity is conserved by the network. The global MPA network represents 33% of the study region and success to reach a range of conservation targets for broad scale habitats and species preferential habitats. Thus, no major representation gaps arose from the analysis. However, habitats being covered by one or several MPAs do not mean that they will beneficiate from sufficient conservation. Management rules are not available yet for a majority of MPAs and it is not possible to conclude about potential management gaps in the area.

Keywords: Gap analysis, Marine Protected Areas networks, eastern English Channel, European Marine sites.

4.1. Introduction

Marine Protected Areas (MPAs) are now widely recognized as effective management tools for conserving biodiversity and ecosystem services (Claudet et al., 2011; Halpern and Warner, 2002; Smith et al., 2009; Wood et al., 2008). However, traditional approaches to developing protected area (PA) networks were largely "*ad hoc*" and based on socio-economic or aesthetic criteria, which produced PA networks that failed to adequately represent biodiversity (Margules and Pressey, 2000; Pressey, 1994; Scott et al., 1993). This problem was first quantified in the 1980s and 1990s, when conservationists started to develop geographic information systems (GIS) and large scale vegetation and habitat maps (Scott et al., 1993). This allowed researchers to measure how well biodiversity was represented in existing PAs with the aim of optimizing the locations of future ones (Pressey and Nicholls, 1991; Scott et al., 1993). This new approach was named "gap analysis" and was pioneered in the United States of America (Scott et al., 1993). Moreover, gap analysis became a key part of the systematic conservation planning approach, as it forms an early step in measuring the effectiveness of existing PA networks (Margules and Pressey, 2000; Margules and Sarkar, 2007).

Gap analyses have now been undertaken in a number of countries (Fearnside and Ferraz, 1995; Mills et al., 2011; Oldfield et al., 2004; Opermanis et al., 2012; Powell et al., 2000; Pressey and Nicholls, 1991; Scott et al., 2001) and at a global scale (Andelman and Willig, 2003; Rodrigues et al., 2004). In addition, the importance of carrying out national gap analyses has been recognized by the Convention on Biological Diversity and each signatory has committed to undertake such research (CBD, 1992). Such an approach is particularly relevant for assessing the effectiveness of MPAs in Europe because each country has implemented a range of different conservation strategies. For example, EU members have had to implement a network of European Marine Sites (EMS) since 2002, which are marine versions of the Natura 2000 sites: Special Areas of Conservation (SACs) are the EMS sites based on the Habitat Directive and Special Protection Areas (SPAs) correspond to the Bird Directive. However, recent work from the Baltic sea showed that the SAC network failed to adequately represent four fish recruitment habitats in the Swedish-Finnish archipelago (Sundblad et al., 2011) and this suggests that these EMS systems are unlikely to be sufficient for achieving conservation goals.

This problem with the EMS approach has been recognized by both the French and UK Governments, which is why they are supplementing their SACs and SPAs with new conservation initiatives. More specifically, the French are implementing a number of Natural Marine Parks and the UK is developing a network of Marine Conservation Zones (MCZs) (NaturalEngland and JNCC, 2010)(Figure 23). However, each country used a different approach for identifying and developing these new MPAs and this has particular implications for the eastern English Channel (EEC), where these different types of MPA can be found. Moreover, the EEC is a discrete biogeographical entity (Carpentier et al., 2009) and can be considered as a distinctly functioning eco-region (ICES, 2008). Therefore, there is a real need for a gap analysis at the scale of the EEC, as this would provide important information for managers and follow a more ecologically sound approach than considering French and English waters separately. Moreover, the benthic and the pelagic habitats of the EEC have been recently mapped (Coggan and Diesing, 2011; Delavenne et al., *subm*), as has the preferential habitat of some species of conservation or management concern (Carpentier et al., 2009). Thus, there is a wealth of biodiversity data for the EEC which makes it possible to undertake a regional gap analysis based on both a coarse filter (habitats) and fine filter (species) approach, and so measure biodiversity representation at multiple scales (Dudley and Parish, 2006; Scott et al., 1993).

The original gap analysis approach simply measured the percentage of the distribution of each feature that was protected (Scott et al., 1993). However, more recent work recognized that it was more appropriate to compare protection levels with quantitative targets (Carwardine et al., 2009;

Knight et al., 2006). In some cases, conservation managers rely on politically driven targets exist, such as the 10% coverage of each habitat recommended by the Convention on Biological Diversity (CBD, 2004), but these have been heavily criticized for their lack of ecological relevance (Svancara et al., 2005). Fortunately, research in the EEC has developed habitat targets using the Species-Area Relationship approach (Metcalfe et al., 2012; Rondinini, 2011a) and so these targets can be used as thresholds to assess the efficiency of the MPA network at the eco-region scale.

Therefore, in this chapter I undertook a gap analysis to measure how well the existing MPA network in the EEC meets targets for protecting habitats types and species. In doing so, I adopted the typology developed by Dudley and Parish (2006; Box 2) and undertook a representation and ecological gap analysis based on biodiversity surrogate coverage and a management gap analysis based on the different types of MPA types and management.

Box 2: The three types of gap analysis (after Dudley and Parish (2006))

Representation gaps: there are either (1) no representation of a particular species or ecosystem in any protected area or, (2) there are not enough examples of the species/ecosystem represented to ensure long-term protection

Ecological gaps: while the species/ecosystem is represented in the protected area system, the occurrence is either of inadequate ecological condition, or the protected areas fail to address the movements or specific conditions necessary for the long-term species survival or ecosystem functioning.

Management gaps: protected areas exist but management regimes (management objectives, governance types, or management effectiveness) do not provide full security for particular species or ecosystem given the local conditions.

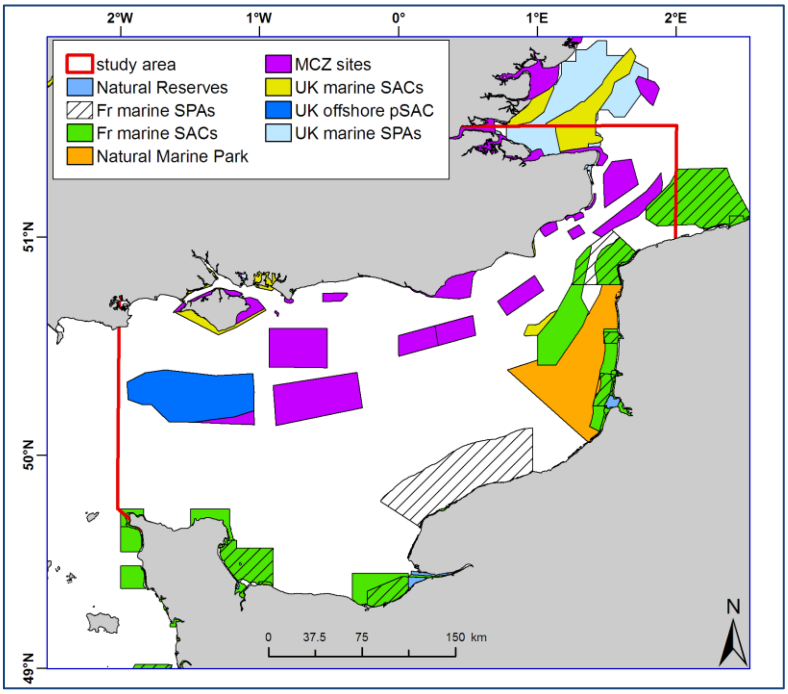


Figure 23: The existing Marine Protected Areas in the eastern English Channel. SACs are Special Areas of conservation; a pSAC is a potential area of conservation; SPAs are Special Protected Areas, and; a MCZ is a Marine Conservation Zone. The study area is represented by the red line

4.2. Methods

4.2.1. Classifying and mapping the Marine Protected Area network

In France, the "Agence des Aires Marines Protégées" (AAMP) is the national agency responsible for the creation and development of the MPA network. In the UK, JNCC and Natural England share the management of MPAs and have divided the UK waters into four regional projects for conservation planning. In the eastern English Channel, the Balanced Seas project was responsible for identifying and recommending MCZs to the UK government (BalancedSeas, 2011).

Five types of marine protected areas are found in the EEC (Figure 23): European marine sites which are named Special Protected Areas (SPAs) and Special Areas of Conservation (SACs) are based on the Habitat (EC, 1992) and Bird (EC, 1979) Directives respectively. A potential SAC (pSAC) is an offshore SAC waiting for approval by the European Commission. National level MPAs are named Natural Reserves and Natural Marine Parks by the French Government and Marine Conservation Zones (MCZs) by the UK Government. In addition, some parts of the MCZ network are defined as "Reference Areas" which have been recommended as zones where human activities are strictly limited (BalancedSeas, 2011). In this analysis I did not include French Natural Reserves because these mainly act to conserve terrestrial habitats and only include estuaries that are not well described by the available species and habitat maps. English SPAs are also very coastal and terrestrial but because of the presence of a significant and large marine SPA in the Thames estuary, they were included in the gap analysis.

4.2.2. Ecological data

Two types of ecological data were used as conservation features in this analysis. The first type was broad habitat maps: the benthic typology was described in the general introduction and the pelagic typology was described in Chapter One.

The second type of ecological data used nine fish preferential habitats (Figure 24) (Carpentier et al., 2009). All the selected species are of conservation interest or under specific management systems in the EEC. Cod (*Gadus morhua*), Spotted ray (*Raja montagui*), thornback ray (*Raja Clavata*) and spurdog (*Squalus acanthias*) are listed as priority species by the OSPAR commission (OSPAR, 2008). Mackerel (*Scomber scombrus*), Sole, (*Solea solea*), plaice (*Pleuronectes platessa*), whiting (*Merlangius merlangus*) and horse mackerel (*Trachurus trachurus*) are species listed in the UK Biodiversity Action Plan. Cod, mackerel, whiting, sole and plaice are also fish stocks under fishing legislation and are evaluated by the International Council for the Exploration of the Sea (ICES). Separate preferential habitats were available for young (< one year old) and older fishes (> one year old) for whiting, plaice and horse mackerel. The preferential habitats for the remaining species were not divided into age groups.

Preferential habitats were produced with generalized linear models (GLM) (Carpentier et al., 2009; Lauria et al., 2011; Martin et al., 2010; McCullagh and Nelder, 1989). GLM is widely used for modeling species distributions using environmental variables (Guisan et al., 2006). It describes the mean response of species abundance according to environmental conditions which were: temperature, salinity, bed shear stress, depth and seabed sediment type (Carpentier et al, 2009). The spurdog analysis was only based on presence/absence data because both low occurrence and abundance was observed. A probability of presence model was produced instead of a preferential habitat. Preferential and probable habitat maps were available as grids of 0.009° resolution and all values were rescaled between 0 and 1 so that each grid node value represented a "habitat suitability index" (HSI) value.

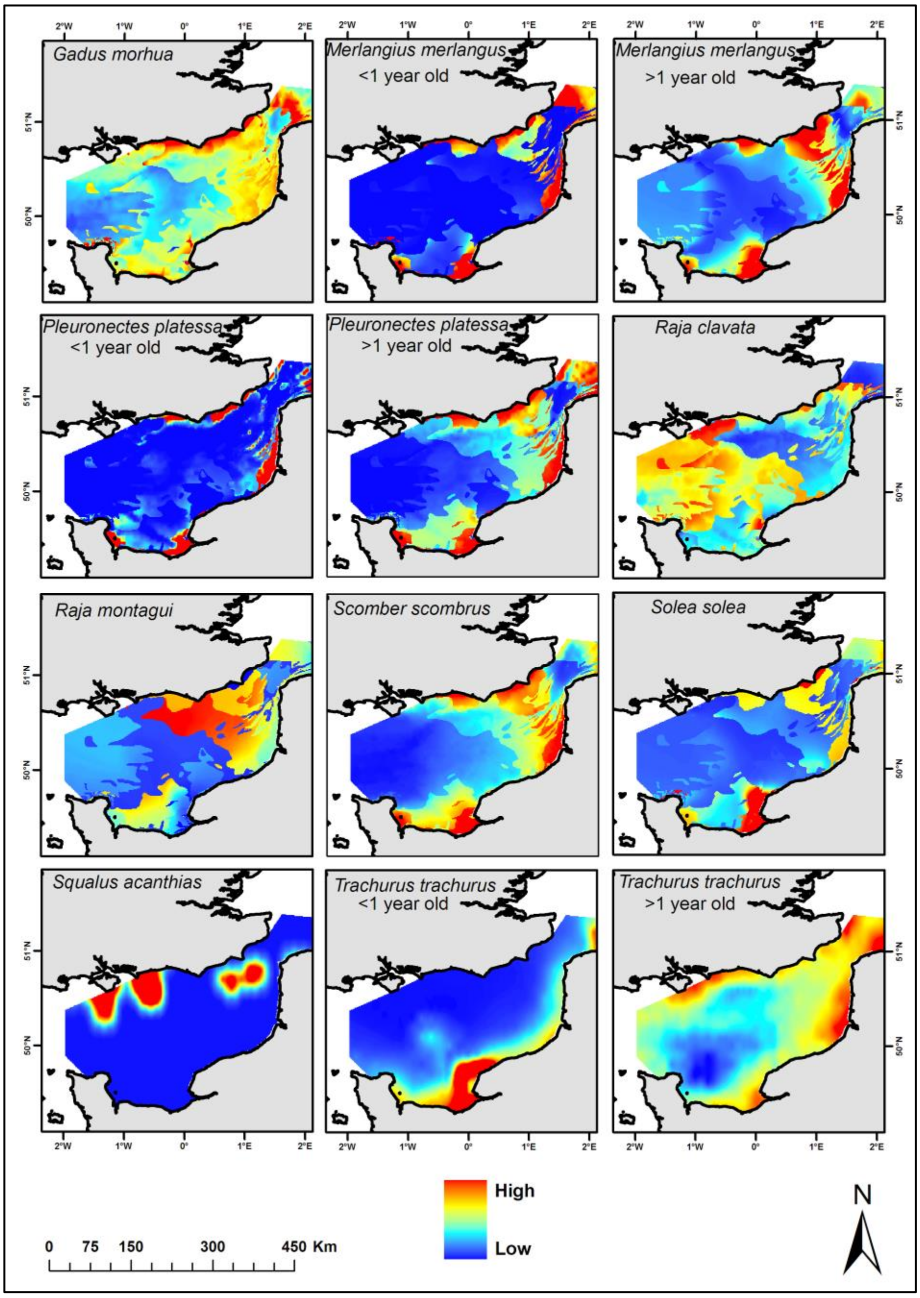


Figure 24: Preferential and probable habitat maps for the nine species and ages.

4.2.3. GIS analysis

All MPA locations and ecological GIS layers were projected using a customized WGS84 Mercator system centred on the EEC. Area and Overlap calculations were done using ArcGIS 9.2 with the analysis tool and the Hawth analysis tool extension. The overlap between each habitat type and each MPA type was calculated in km². For the preferential habitats of the nine species, the Habitat Suitability Index was summed over all the area and summed in each MPA.

4.2.4. Conservation targets

The coverage of each habitat type by the MPA network was compared to quantitative targets calculated with the SAR method (Desmet and Cowling, 2004) in Metcalfe et (2012) and in chapter four of this thesis. These targets appear in the last column of Table 3. For the benthic habitats, the targets for rocky and coarse sediments habitats were calculated with the megabenthos data, whereas the targets for fine sediment habitats were calculated with macrobenthos data. This choice was made because the Hamon grab used to sample macrobenthos is not really efficient in coarse sediments, whereas trawl sampled megabenthos maybe more representative of the habitat. I also measured the extent to which protection levels met targets specified in the Ecological Network Guidance (ENG), as calculated by JNCC (NaturalEngland and JNCC, 2010; Rondinini, 2011a). Here I used the higher ENG habitat targets, which aimed to represent 80% of the species found in each habitat as this was consistent with the targets that were developed in earlier chapters. The ENG targets were calculated for each level 3 EUNIS habitat. I also used the 10% habitat target recommended by the CBD (CBD, 2004).

4.3. Results

4.3.1. The MPA network

The MPA network in the EEC has a combined area of 11 900 km², covering 33% of the 36 108 km² of the study area. The one French Marine Park has an area of 2304 km², whereas the English MCZ network consisted of 23 sites with a median area of 48 km² (Table 13). Natura 2000 areas covered 6815 km² and some of these overlapped with the Marine Conservation zones on the English side and the Natural Marine Park on the French side (Table 14). Some coastal areas were found to have three different designations, such as areas close to the Authie, Somme and Canche estuaries, which were designated as SPAs, SACs and part of the French Marine Park (Figure 23).

Table 13: MPA characteristics. Total area corresponds to the surface of the study area covered by the MPA type. Number corresponds to the number of MPAs patches and the median area is calculated from all the MPA patches for each MPA type. For MPA types, Fr is for French, UK for United Kingdom, SAC is Special Area of Conservation, SPA is Special Protected Area, pSAC corresponds to the potential SAC. MCZ are for Marine Conservation Zones.

MPA type	Total area (km²)	Number	Median area (km²)
Fr SAC	2679	13	142
Fr SPA	3400	12	88
Fr Marine Park	2304	1	2304
UK SAC	509	7	37
UK pSAC	1374	1	1374
UK SPA	555	8	5.2
UK MCZ	3454	23	48
UK MCZ reference areas	139	15	0.38

Table 14: Surface overlap in km² between different types of MPAs

MPA type	Fr Marine Park	Fr SPA
Fr SAC	637	1496
Fr SPA	191	

MPA type	UK MCZ	UK SPA
UK SAC	112	255
UK SPA	49	

4.3.2. Habitats coverage by the MPA network

SACs aim to protect important habitats, as specified in Annexe III of the Habitats Directive. In the EEC, English and French SACs covered between 6 and 82 % of the benthic habitats (Table 15). Coarse sediment habitats were the least well protected (A5.1) and high energy circalittoral rock habitat (A4.1) were the most protected, mostly because of the presence of the Wight Barfleur reef in the potential SAC.

The combined MPA network met all of the SAR-based targets and the 10% CBD habitat target was met for most of the benthic habitat types (Table 16). However, the SAC network only met the SAR-based targets for 11 of the benthic habitats. The ENG habitat targets that were used in the MCZ project were generally higher than those calculated in this thesis (as discussed in Chapter Three) and the MPA network only met targets for 7 of the 10 EUNIS level 3 habitats. More specifically, the network failed to meet targets for low energy infralittoral rock, circalittoral rock and coarse sediment habitats (Table 16).

Table 15: Benthic habitats and their percentage of area within each type of MPA. Results are given at the finest possible EUNIS habitats. Thesis targets are the target calculated in chapter four

	EUNIS habitat description	Total Area (km^2) of habitat	Fr SAC %	Fr SPA %	Marine Park %	UK SAC %	UK pSAC %	UK SPA %	MCZ %	MCZ reference areas %	TOTAL %	Thesis targets %
A3.1	High energy infralittoral rock	14		3.9		67.9			15.7	22.9	78.6	13.5
A3.2	Moderate energy infralittoral rock	150	37.1	8.6	0.03	9.6			7.9		58.0	13.5
A3.3	Low energy infralittoral rock	83	13.8	12.9					1.7		21.7	13.5
A4.1	High energy circalittoral rock	1146		0.03		0.03	81.89		14.4	1.3	96.3	12.3
A4.2	Moderate energy circalittoral rock	181	18.8	3.3		0.12			8.7		29.3	13.5
A4.3	Low energy circalittoral rock	101	8.7	10.7		0.34			0.32		12.9	11.7
A5.1	Coarse sediment	28378	4.1	4.7	3.9	0.8	1.5	0.7	9.0	0.4	25.7	
A5.13	Infralittoral coarse sediment	3901	7.8	22.5	14.3	4.8		3.7	7.6	0.3	52.6	20.7
A5.14	Circalittoral coarse sediment	18458	3.7	1.1	3.0	0.3	1.3	0.2	8.1	0.4	21.3	20.3
A5.15	Deep circalittoral coarse sediment	6019	3.2	4.3	0.1		3.1	0.2	12.9	0.2	21.7	17.4
A5.2	Sand	5604	22.9	13.8	17.5	4.4		4.2	8.0	0.4	47.7	
A5.23	Infralittoral	2848	27.8	17.8	28.8	5.6		4.1	5.4	0.3	58.2	24.7

or A5.24	fine sand or muddy sand										
A5.25 or A5.26	Circalittoral fine sand or muddy sand	2236	18.9	10.5	7.3	3.9	4.8	11.2	0.6	39.6	22.6
A5.27	Deep circalittoral sand	520	12.6	6.3		0.4	1.9	9.1		25.0	13.5
A5.3	Mud	255	43.6	34.8	1.1	3.3	5.9	5.6	0.1	55.7	
A5.33 or A5.34	Infralittoral sandy mud or fine mud	143	52.0	47.8	1.2	2.7	5.2	6.8	0.2	65.7	14.0
A5.35 or A5.36	Circalittoral sandy mud or fine mud	112	32.9	18.2	1.1	4.1	6.8	4.0		42.9	9.0
A5.4	Mixed sediment	196				12.7	57.7	17.5	0.2	65.8	
A5.43	Infralittoral mixed sediments	122				15.5	65.3	21.1	0.4	75.4	14.4
A5.44	Circalittoral mixed sediments	67				7.6	44.0	12.7		49.3	14.4
A5.45	Deep circalittoral mixed sediments	7				12.9	54.3			57.1	10.3

Table 16: Success of the MPA network in meeting the habitat targets. Thesis targets correspond to the ones in the last column of the table 3, calculated at the finest EUNIS habitat level possible. ENG targets are the habitat targets calculated in Rondinini et al (2011) to conserve 80% of the habitat species, calculated for each EUNIS level 3 habitat in the English waters and CBD target is 10% for each habitat.

	EUNIS habitat description	Thesis target	ENG targets	CBD target
A3.1	High energy infralittoral rock	Yes	Yes	Yes
A3.2	Moderate energy infralittoral rock	Yes	Yes	Yes
A3.3	Low energy infralittoral rock	Yes	**No**	Yes
A4.1	High energy circalittoral rock	Yes	Yes	Yes
A4.2	Moderate energy circalittoral rock	Yes	Yes	Yes
A4.3	Low energy circalittoral rock	Yes	**No**	Yes
A5.1	Coarse sediment		**No**	Yes
A5.13	Infralittoral coarse sediment	Yes		Yes
A5.14	Circalittoral coarse sediment	Yes		Yes
A5.15	Deep circalittoral coarse sediment	Yes		Yes
A5.2	Sand		Yes	Yes
A5.23 or A5.24	Infralittoral fine sand or muddy sand	Yes		Yes
A5.25 or A5.26	Circalittoral fine sand or muddy sand	Yes		Yes
A5.27	Deep circalittoral sand	Yes		Yes
A5.3	Mud		Yes	Yes
A5.33 or A5.34	Infralittoral sandy mud or fine mud	Yes		Yes
A5.35 or A5.36	Circalittoral sandy mud or fine mud	Yes		Yes
A5.4	Mixed sediments		Yes	Yes
A5.43	Infralittoral mixed sediments	Yes		Yes
A5.44	Circalittoral mixed sediments	Yes		Yes
A5.45	Deep circalittoral mixed sediments	Yes		Yes

The coverage of the pelagic habitats by the SAC network ranged from 0 for habitat 5 to 47% for habitat 9 (Table 17). The addition of SPAs, MCZ and the Marine Park to the SAC network effectively complemented the representation of all 13 pelagic habitats. This resulted in a good representation of pelagic habitats by the MPA network at the scale of the EEC, with coverage ranging from 11% for habitat 7 to 90% for habitat 9. These results are also higher than the 10% habitat target recommended by the CBD.

Table17: Pelagic habitats and their percentage of area within each type of MPA

	Total Area (km^2) of habitat	Fr SAC %	Fr SPA %	Marine Park %	UK SAC %	UK pSAC %	UK SPA %	MCZ %	MCZ reference areas %	TOTAL %
1	767	11.3	8.6		25.8		75.9	19.0		72.2
2	663				7.48		8.28	28.67	3.56	36.5
3	3830	3.4	0.3			17.9		0.5		21.6
4	10330	2.3	1.6			5.5		12.0	1.0	19.7
5	2723							23.5		23.5
6	2011				9.3		3.2	16.0	0.5	20.2
7	1818				4.4	6.1	0.6	0.6	0.2	10.0
8	7989	9.2	25.7	12.6						41.5
9	956	46.5	31.0	56.9	0.4					79.9
10	423				0.9		0.2	37.6	0.1	38.5
11	2852	33.5	10.6	20.1	2.4			8.0		50.7
12	967	19.5	26.46					24.4		52.4
13	2926	8.5	8.6					5.6		14.6

4.3.3. Species preferential habitats coverage by the MPA network

For the nine considered species, their preferential habitats were well covered by the combined MPA network. It should be highlighted that these should not be interpreted as the percentage area of habitat conserved, as the analysis was based on continuous Habitat Suitability Index scores, rather than a categorical representation. Instead, they should be seen as the relative amount of the overall EEC suitability for a given species. However, the results ranged between 27 to 40% which indicated a good representation of these species preferential habitats (Table 18). The three species that were mapped at different life stages showed different spatial distributions patterns during their lifecycle, but were still well represented by the combined MPA network at each stage.

Table 18: Species habitat suitability indices percentage within each type of MPA

	Fr SAC	Fr SPA	Marine Park	UK SAC	UK pSAC	UK SPA	MCZ	MCZ reference areas	TOTAL
Gadus morhua	4.6	11.2	7.6	7.2	3.1	0.3	9.6	0.4	30.9
Merlangius merlangius < 1 year old	10.6	23.5	13.5	2.1		0.1	14.3	0.5	36.3
Merlangius merlangius > 1 year old	7.8	18.0	11.5	3.4	2.7	0.2	8.8	0.2	31.7
Pleuronectes platessa < 1 year old	10.9	31.3	19.9	3.4	0.5	0.1	11.1	0.3	39.7
Pleuronectes platessa >1 year old	7.6	18.8	12.7	5.0	0.1	0.2	9.1	0.3	31.7
Raja clavata	3.5	8.6	5.6	6.9	4.2	0.3	9.0	0.4	28.3
Raja montagui	4.6	11.6	8.6	4.0	2.5	0.4	8.7	0.2	26.9
Scomber scombrus	7.0	17.9	12.0	6.5	0.7	0.3	7.6	0.2	31.6
Solea solea	6.5	16.0	8.0	7.1	1.7	0.2	9.0	0.3	30.3
Squalus acanthias	4.1	4.3	0.4	2.0	7.5	0.2	16.9	0.1	30.7
Trachurus trachurus < 1 year old	6.6	18.4	8.0	17.0	0.8	0.2	4.6	0.2	35.5
Trachurus trachurus >1 year old	3.7	8.9	6.1	6.8	3.9	0.3	8.8	0.4	28.2

4.4. Discussion

Gap analysis remains at the heart of the systematic conservation planning process and provides a wealth of useful information for decision makers, especially when dealing with international MPA networks. A previous gap analysis was used to inform the MCZ project in English waters (BalancedSeas, 2011). It showed that broad-scale habitat representativeness was good, that many habitat types were found in more than one MPA and that results would be improved by implementing the proposed MCZs in the Balanced Seas region. However, this study had a national perspective and did not include French MPAs, whereas ecosystem-based management principles advise that researchers should work at a regional scale (Halpern et al., 2010; OSPAR, 2006). This study provides such a regional approach and provides data to assess the European Marine Sites network efficiency and measure national contributions to habitat representativeness at the eco-region scale.

4.4.1. The MPA network

The first result of this gap analysis was to highlight that the eastern English Channel has a large Marine Protected Areas network, so that the 81 MPAs have a combined area of 11 900 km² and cover 33% of the study region. This is much larger than that recommended by the Convention on Biological Diversity, which recommends 10% coverage of ecologically representative and effectively managed MPAs by 2012 (CBD, 2004). In 2004, only 0.5% of the ocean surface was covered by MPAs but this increased to 1.17% in 2010, with 4.32% of the continental shelf zone protected (IUCN, 2010). However, a large number of countries, such as UK and France, expanded their MPA networks to meet the 2012 target and so these global levels of protection must have improved.

Looking at these results, the EEC MPA network appeared exemplary for marine biodiversity representation. Moreover, although this study region was limited from the Cotentin peninsula to the south of the Thames estuary, some of the MPA networks considered here extend into adjacent areas and gave added value to the network. This study also illustrated the different national strategies, with the UK creating 23 Marine Conservation Zones covering 3450 km² (31 areas in the whole Balanced Sea area) whereas France decided to create a large Marine Park of 2 304 km² to form a network with existing SACs and SPAs. However, it should be noted that there is still debate over when and how many of the proposed MCZs should be implemented.

4.4.2. How successful is the MPA network at representing biological diversity of the EEC?

The representation of all considered habitats by the MPA network was found to be sufficient. Benthic habitats were all covered by a MPA at the finer possible scale: the sediment habitats were available at a EUNIS level 4 precision whereas the rocky habitats were described to the level 3. In the EEC, the area of these habitats were very different, ranging from 7 km² for the deep circalittoral mixed sediment (A5.45) habitat to 18 458 km² for the circalittoral coarse sediment (A5.14) habitat. While it is generally easier to reach a high percentage of protection for the more restricted habitats, widespread habitats were still well covered by the MPA network. The recent national initiatives really increased the representativeness of the MPA network. If we only considered the SAC network, the coarse sediment (A5.1) habitats did not meet the 20%, although the fine sediment habitats (A5.2 and A5.3) already met the SAR-based targets for benthic and pelagic habitats. The whole network met the ENG targets for 7 of the 10 level 3 EUNIS habitats present in the EEC. In the three habitats where ENG targets had not been reached by the EEC MPA network (two rocky habitats and a coarse sediment one), the ENG target to conserve 70% of the species living in the habitat was about 15%, so the MPA network met the lower ENG targets (NaturalEngland and JNCC, 2010; Rondinini, 2011a).

The MCZ network alone met targets for almost all the described benthic habitats, as it was designed to represent all the EUNIS level 3 habitats (BalancedSeas, 2011). A portion of each of these habitats can also be found in the reference areas, which are meant to provide the highest level of protection within MPAs. Due to its unique location, the French Marine Park failed to represent a part of all the described habitats. However, it increased the total representation of the coarse sediment habitats, which were under-represented by the SACs. The large size of this MPA also makes it important for pelagic and highly mobile species (Gell and Roberts, 2003). Moreover, this area of the EEC is known for its importance as a coastal nursery ground of several commercial fish (Carpentier et al., 2009; Koubbi et al., 2006).

Overall, the soft sediment coastal areas were best protected by the MPA network. This was probably because coastal areas are easier to manage and ensure continuity with the terrestrial protected areas, but offshore areas should not be neglected to ensure that all biodiversity is conserved (Game et al., 2009; Grantham et al., 2011). Fortunately, recent efforts in the EEC have focused on some offshore and coarse sediment areas, such as the Wight Barfleur pSAC, which will be dedicated to conserve this vulnerable habitat.

Pelagic habitats were also well represented in the MPA network. Targets were met for each of the 13 habitats, which was an important result because these water masses have been proved to be suitable surrogates for pelagic biodiversity from phytoplankton to pelagic fishes (Delavenne et al., *subm*) and this water column realm is often missing in marine conservation planning (Game et al., 2009; Grantham et al., 2011). The complementarity of the whole network was revealed by the study of the representation of the pelagic habitats. SACs and SPAs did not reach the 10% target for the pelagic habitats, nor did national networks on their own, but the combined network met the CBD target for all pelagic habitats.

In this study I only considered broad-scale EUNIS habitats but other habitats arose were identified as important conservation features by the OSPAR commission and the ENG (NaturalEngland and JNCC, 2010; OSPAR, 2008). These are sensitive and endangered special habitats, such as littoral chalk communities and intertidal mudflats. The Balanced project concluded that they were well covered by the MCZ network but these are coastal habitats and some are also found on the French side and should be covered by the existing MPAs.

The fine filter approach used habitat suitability indices for nine species and distinguished between juvenile and adult life-stages for three species. Results showed a good cover of their preferential habitats by the MPA network. Cod, Spotted ray, thornback ray and spurdog are listed as priority species by the OSPAR commission (OSPAR, 2008), and results showed that the EEC may provide a good conservation zone for these species, even if the two rays are less well protected than most of the other species. Eggs and larvae were not taken into account in this study although they represent critical life stages for local population renewal. However, important nursery grounds for fish such as plaice or soles are present along the French coast (Carpentier et al., 2009; Koubbi et al., 2006) and were found to be well covered by the Natural Marine Park. However, the MPA network benefits more to coastal species. Offshore species, such as the rays and important commercial species like the red gurnard (*Aspitrigla cuculus*), squid species (*Loligo forbesi and vulgaris*) and the dogfish (*Scyliorhinus canicula*), were not considered in this study. These prefer offshore, coarse sediment habitats (Carpentier et al., 2009) that are the less well represented benthic habitats and so might not be well covered by the MPA network.

In the context of the EEC gap analysis we can conclude that habitats are well represented by the combined MPA network, widely exceeding the CBD target of 10%. Species preferential habitats are well represented too, so there appear to be no big representation gaps for the considered biodiversity surrogates. To assess ecological gaps, we could assume that if every benthic and pelagic habitat is well conserved in one or several MPAs then their ecological functions will be maintained. Moreover, the connectivity between the 81 MPAs appears to be good, as the Balanced Seas project found good connectivity for the benthic habitats in the MCZs (BalancedSeas, 2011) and this study found that every pelagic and benthic habitat was present at least in 2 different MPAs. This connectivity is really important in the temperate marine realm, where most species use a number of different habitats as part of their life cycle (Blyth-Skyrme et al., 2006; Gell and Roberts, 2003; Grüss et al., 2011; Roberts et al., 2003a). However, although a variety of nursery grounds (Carpentier et al., 2009; Koubbi et al., 2006) and spawning grounds (Carpentier et al., 2009; Lelievre et al., 2012) have been partially described in the EEC, they were not explicitly included in the present analyses. In addition, we lack data at the species level on connectivity between the different life stages and trophic relationships, so more studies are needed on larval dispersion to evaluate the rate of passive connectivity between MPAs (Game et al., 2009; Grüss et al., 2011; Roberts et al., 2003a; Shanks et al., 2003). Studying larval dispersion is key for pelagic species but also for benthic species which have a pelagic larval life stage (Ellien et al., 2004; Pechenik, 1999). Thus, future studies of larval dispersal in the region should be extended north, as several commercially important fish in the EEC complete some of their life stages in the North Sea (Loots, 2009; Loots et al., 2010).

4.4.3. How efficient could be MPA network to conserve biological diversity of the EEC?

The results of the gap analysis showed that every biodiversity surrogates are well represented by the EEC MPA network. This is a really interesting and encouraging result for biodiversity conservation in the EEC ecoregion but it does not provide information about potential management gaps (Dudley and Parish, 2006). We know that SACs, SPAs, MCZs, Marine Park, together with French Natural Reserves are managed differently, apply different restrictions on human activities and have different management plans and all of these will influence how well they maintain their biodiversity.

We can investigate this further because each EMS has its own management plan named “Conservation objectives and advice on operations” or “Document d’objectif (DOCOB)”. For SACs, this document advises that any human activity that could have a negative impact on important habitats needs to be assessed, but, at least for now, none of these recommendations has led to any restrictions or bans. The SPAs seek to conserve birds and their associated habitats and the gap analysis found that many of the benthic and pelagic habitats and species only had high representation levels because of the large French SPAs. The question is whether these habitat types and fish species will actually benefit from being within an SPA?

The French Marine Park has not published its management plan yet and it has to be stressed that the mapped Park may not be the definitive Marine Park as the process is still no going. Similarly, the rules for managing the MCZ network have not been specified, although they will contain Reference Areas (RAs) where anthropogenic activities will be strictly limited (BalancedSeas, 2011). Thus, these five types of MPAs are subject to different legislation and possible restrictions, making it difficult to evaluate the management effectiveness of the MPA network in the EEC. This means that, whilst working at the eco-region level might be more ecologically relevant, without a clear cooperation framework between countries, a trans-boundary MPA network might not be optimized, even if it is the primary goal of the Natura 2000 Protected areas network (Opermanis et al., 2012).

A large number of studies have found a positive influence of MPAs on habitats, associated species and on fisheries (Claudet et al., 2011; Gell and Roberts, 2003; Goñi et al., 2011; Vandeperre et al., 2011). All these studies focused on marine reserves *sensus stricto*, no-take areas, where all human impacts are banned. Few studies have looked at partially protected MPAs, but Lester and Halpern (2008) published a review comparing fully versus partially protected areas (areas where just recreational fishing was allowed or just non-destructive fishing gears), and found that partially protected areas confer some ecological benefits compared to open areas but no significant differences in biological parameters, such as species densities or mean sizes, were found between partially protected areas and open areas. Another review by Claudet et al. (2008) showed that no-take areas size had a biological effect: larger no-take areas led to an increase in commercial fish outside; whereas the size of the buffer zone, where some activities were limited, had the opposite impact. Thus, the literature tends to demonstrate that for both conservation and management purposes, no-take areas are the more effective MPA types. This means that real caution is needed when interpreting the results of the EEC gap analysis and more research is needed to estimate how well the different types of MPA meet the different conservation targets.

4.4.4. Conclusions

The EEC MPA network covering has a great potential for conserving important habitats and species and contains no major representation gaps. Moreover, it is likely that if every benthic and pelagic habitat is well conserved in one or several MPAs then their ecological functions should be maintained. However, we lack species-specific data to measure whether this MPA network provides sufficient connectivity to ensure the viability of different life stages and trophic relationships

Ecological considerations would favour the designation of no-take areas within the MPA network. However, social, economic and political sides of the ecosystem-based management also need to be considered (Lester and Halpern, 2008). For now, it is not possible to make conclusions about management gaps in France or the UK, as neither country has made specific conservation or management recommendations for their MPAs. Moreover, the spatial overlap between some MPAs may complicate the definition of the area management. Furthermore, MPAs are not the only new management implementation in the EEC: more aggregate extraction sites have been licensed and more offshore wind farms fields are being built. Thus, a relevant spatial plan must be done to coordinate all these new users together with existing ones, such as fisheries (Buléon and Shurmer-Smith, 2008; Carpentier et al., 2009).

Unlike large federal countries such as USA or Australia, the European Union is divided by many scattered borders and this can be an issue in nature conservation at the EU level (Opermanis et al., 2012). This could be overcome by international cooperation and accords for MPA implementation and common indicators to evaluate MPA efficiency. Policy framework such as the Marine Directive (EC, 2008) or international directives such as OSPAR could be first attempts in this direction.

5. Chapter Five

Systematic conservation planning in the eastern English Channel: comparing the Marxan and Zonation decision-support tools

Abstract

The systematic conservation approach is now commonly used for the design of efficient marine protected area (MPA) networks. Identifying these priority areas often involves using specific conservation-planning software. Several of these software programmes have been developed in recent years, each of which differs in the underlying algorithms used. Here, we investigate whether the choice of software influences the location of priority areas by comparing outputs from Marxan and Zonation, two widely used conservation-planning, decision-support tools. Using biological and socio-economic data from the eastern English Channel, we compared their outputs and showed that the two software packages identified similar sets of priority areas, even though the relatively wide distribution of the habitat types and species considered offered a great deal of flexibility. Moreover, this similarity increased with increasing spatial constraint, especially when using real-world cost data, suggesting that choice of cost metric has a greater influence on conservation-planning analyses than choice of software. However, we found that Marxan generally produced more efficient results and Zonation produced results with greater connectivity, so the most appropriate software package will depend on the overall goals of the MPA planning process.

Keywords: Eastern English Channel, marine conservation planning, Marxan, spatial conservation prioritization, Zonation.

5.1. Introduction

The 1992 Convention on Biological Diversity set the ambitious target of establishing, by 2012, a global system of marine protected areas (MPAs) covering 10% of all marine ecological regions, comprising both multiple-use areas and strictly protected areas. MPAs are increasingly seen as important instruments for conserving biodiversity and maintaining fish stocks (Leathwick *et al.*, 2008), and there is some evidence of their potential benefits in the management of fisheries (Gell and Roberts, 2003; Halpern and Warner, 2002). In its strategy for the marine environment (EC, 2008), the European Commission (EC) is also promoting the idea of marine spatial planning (MSP) to provide a framework to improve decision-making and delivering an ecosystem-based approach to the management of marine activities. MSP is also expected to provide a more transparent process of conflict resolution in situations where there are many demands for the use of marine resources and sea space.

This context has led to renewed interest in developing methods for designing efficient MPA networks (Smith *et al.*, 2009). In particular, it is widely recognized that conservation planners must account for opportunity costs and potential biodiversity loss when designing MPA systems. This has led to the widespread adoption of the systematic conservation-planning approach (Margules and Pressey, 2000; Margules and Sarkar, 2007), which is a target-driven process that aims to identify networks of priority areas for ensuring the representation and long-term persistence of biodiversity (Leslie et al., 2003; Margules et al., 2002). Setting targets helps increase transparency and measure progress, but it also allows socio-economic data to be included in the planning process without influencing or endangering conservation goals. Thus, MPA networks can be designed so that they meet targets, whilst also minimizing impacts on stakeholders and increasing the likelihood of their successful implementation (Knight *et al.*, 2006).

Systematic conservation planning generally involves: (i) producing a list of important species and habitat types, known collectively as conservation features; (ii) setting targets for each of these conservation features; (iii) dividing the planning region into a series of planning units; (iv) calculating the amount of each feature found in each planning unit; (v) assigning a cost value to each planning unit; and (vi) using computer software to identify priority areas for conserving biodiversity, reducing fragmentation levels and minimizing planning unit costs (Moilanen et al., 2009c). A number of conservation-planning software packages have been produced, several of which have been used to design MPA networks (e.g. Fernandes et al., 2005; Klein et al., 2008a; Leathwick et al., 2008; Leslie et al., 2003). However, this has created some uncertainty amongst practitioners about whether the location of the identified priority areas varies with the software used. Here, we investigate this issue by comparing results from Marxan and Zonation, two of the most widely used conservation-planning, decision-support tools (Moilanen et al., 2009c).

Marxan uses a minimum-set approach to identify portfolios of planning units that achieve conservation targets at a near-minimal cost. It does this by first defining the cost of a portfolio as an objective function made up of (i) the combined cost of the planning units in the portfolio, which can be a measure of any aspect of the planning unit, such as its area, the risk of being affected by anthropogenic impacts, or the opportunity costs resulting from protection; (ii) a penalty for each unmet target; and (iii) a spatial constraint cost reflecting the portfolio's fragmentation level (Ball and Possingham, 2000; Possingham *et al.*, 2000). The spatial constraint is based on the boundary length of the portfolio, as fragmented portfolios have more of this exposed edge. Reducing this fragmentation involves adding more planning units to the portfolio, producing more viable, but less efficient, results (Ball and Possingham, 2000).

In contrast, Zonation uses a maximum-cover approach that aims to maximize the conservation benefits for a fixed cost specified by the user by first calculating the marginal loss for each of the cells in the planning region (Moilanen et al., 2005; Moilanen et al., 2009a). It then removes cells one at a time based on maximizing the overall conservation value of the remaining area to produce a conservation-value map based on the hierarchical ranking of the landscape. This conservation-value map then forms the basis of further analyses, and Zonation has a range of options for incorporating connectivity and viability into the prioritization process (Arponen et al., 2006; Cabeza and Moilanen, 2006; Moilanen et al., 2009c).

This means the approaches that underpin Marxan and Zonation are fundamentally different, with Marxan seeking to minimize costs, while meeting specified targets, and Zonation seeking to maximize biodiversity benefits given a specified cost. However, the Zonation outputs can be modified to identify priority areas for meeting specified targets, and this is why marine conservation planners have used both software packages to identify MPA networks based on a target-setting approach (e.g. Klein *et al.*, 2008b; Leathwick *et al.*, 2008). Given these differences, one might expect Marxan and Zonation to identify different sets of priority areas, which could create confusion and doubt about the value of both software packages. Alternatively, one might assume that results should be similar because areas that are needed to meet targets will always be selected, and this was found in earlier work that compared outputs between Marxan and C-Plan, another reserve-system-design tool (Carwardine *et al.*, 2007). In addition, one might expect similar results when using real-world cost data in the analyses, such as information on opportunity costs or threats. This is because these data have a specific spatial pattern within the planning region, and so the same low-cost areas containing important biodiversity tend to be selected (Nhancale and Smith, 2011; Richardson et al., 2006). Therefore, in this chapter, we use data from the eastern English Channel to investigate whether priority-area and conservation-value maps produced by Marxan and Zonation differ and whether this is sensitive to the conservation target and the type of cost metric used in the analysis.

5.2. Material and methods

5.2.1. Mapping the physical data

Five environmental parameters were selected to describe the range of ecosystems found in the eastern English Channel: depth, temperature, sediment type, salinity, and bed-shear stress. Depth combined bathymetry and mean sea level. Bathymetric data were derived from SHOM (Service Hydrographique et Océanographique de la Marine) hydrographic charts, whilst mean sea level (at mid-tide) was estimated using a hydrodynamic model. Temperature and salinity data were measured *in situ* during IFREMER's Channel Ground Fish Survey (CGFS, 1997–2006) and were used to estimate anomalies (observed surface temperature or salinity minus the mean for the area surveyed) and bottom–surface differences. Seabed shear-stress estimates were obtained from a 2D hydrodynamic model originally developed for the Irish Sea, but extended to cover the northwest European shelf (Carpentier *et al.*, 2009). Seabed sediment types were extracted from a sediment map of the English Channel (Larsonneur et al., 1982), in which the original 29 sediment classes were aggregated into the following five broader classes: (i) fine sand, (ii) coarse sand, (iii) fine heterogeneous sandy gravel, (iv) coarse heterogeneous sandy gravel, and (v) pebbles. We used this sediment type map because previous studies in the eastern English Channel showed that benthic invertebrate communities (San-Vicente Añorve, 1995) and fish, cephalopods, and macroinvertebrate species assemblages (Vaz *et al.*, 2007) were related to substrate type

We classified the depth and seabed shear-stress maps into five types based on quantile values. Temperature and salinity exhibited less variation, so we used the same approach to divide these into three types. This classification system produced maps that contained an equal area of each type, so that each broad range of the physical environment would be represented in the final portfolios. However, it should be stressed that this is a preliminary approach which was first undertaken during this thesis. The need for a better classification that takes into account the temporal dynamics and biodiversity value of these different types of physical phenomena arose with this first approach and led to the production of seasonal pelagic typologies (chapter two) and the use of a pelagic and a EUNIS benthic typology in the following chapters.

5.2.2. Mapping the biological data

We used two types of biological distribution data in the analyses: a habitat map based on benthic invertebrate communities to represent broader biodiversity, and eight species-distribution maps to represent "fine-scale" biodiversity patterns (Noss, 1990). We selected these eight additional species because they are economically and ecologically important and ensured the representation of species with offshore and inshore spatial distributions (Table 19). This study aimed at comparing outputs from the different software packages, so it was not deemed necessary to include a large number of species in this exercise.

Table 19: Species used as representative features

Common name	Latin name	Development stage
Herring	*Clupeus harengus*	<1 year old and >1 year old
Cod	*Gadus morhua*	All ages
Tope	*Galeorhinus galeus*	All ages
Veined squid	*Loligo forbesis*	All ages
Plaice	*Pleuronectes platessa*	<1 year old and >1 year old
Spider crab	*Maja brachydactyla*	All ages
Lesser-spotted dogfish	*Scyliorhinus canicula*	All ages
Spurdog, spiny dogfish	*Squalus acanthius*	All ages

5.2.3. Designing the conservation planning system

We produced the planning unit theme by creating a series of 5629 16-km^2 grid squares using the repeating-shapes extension in ArcView 3.2 and then calculated the area of each conservation feature found in each planning unit. Because some of the planning units overlapped the coastline, less of their area fell within the planning region. For these overlapping planning units, we calculated their area within the planning region by clipping them with the coastline boundary and used these area values as the basis of the planning unit costs. Thus, planning units that only contained a small amount of the English Channel tended to contain less of each conservation feature, but also had a lower cost.

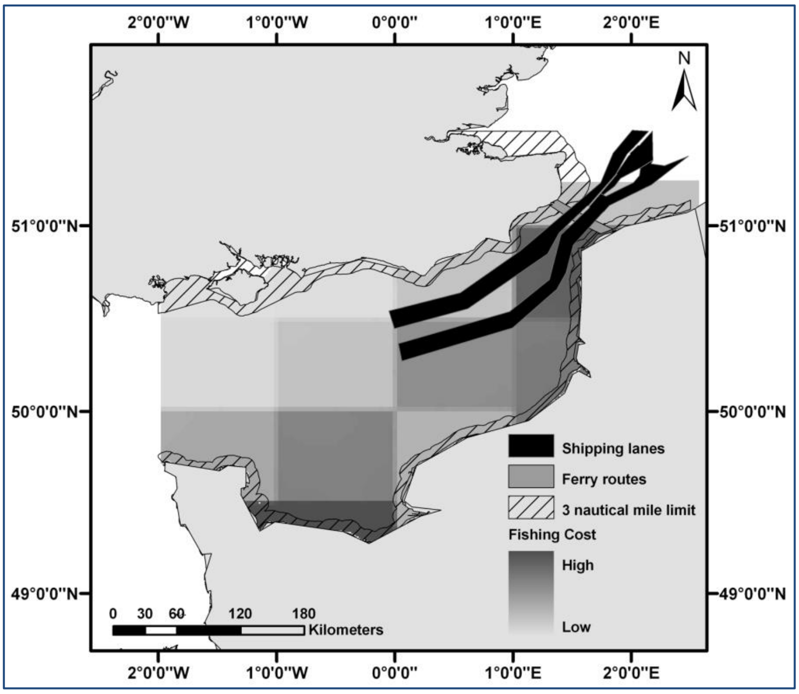


Figure 25: Parameters used to define the planning-unit costs

We carried out nine Marxan and nine Zonation analyses by using three different planning-unit cost metrics and three different sets of targets. Cost metric 1 was "area cost", which was based on the surface area of each planning unit, so that all the planning units had the same cost value apart from those located at the edges of the planning region. Cost metric 2 was "accessibility cost", which was also based on the surface area, but values were reduced by 50% in planning units considered more likely to be suitable for inclusion in a MPA network based on current human activity patterns. Thus, lower values were given to planning units falling within the 3-nautical-mile zone, where trawling is restricted, and within shipping lanes and ferry routes, where fishing pressure is reduced (Figure 25). Cost metric 3 was "fishing cost", which was based on the fishing profitability of the planning unit, using official data from the French Maritime Fisheries and Aquaculture Office (undertaken by the IFREMER-Halieutic information system) that was then modified to weight the costs by the distance to the nearest French port. French vessels dominate fisheries in the eastern English Channel (Martin *et al.*, 2009). The number of vessels vary, but, for example in 2005, 641 French boats and 49 English boats over 10 m were recorded (Carpentier et al., 2009), so this cost variable was only based on data from the French Maritime Fisheries and Aquaculture Office and French ports.

In each analysis, we used the same percentage target for all the habitat types and species, but in the different analyses, we used targets of 10, 30, and 50%. The 10% target has been commonly applied in the literature, but has also been criticized for not being ecologically relevant (Pressey et al., 2003), and the 30% target is currently recommended by the IUCN (IUCN, 2003) and has also been used in previous studies (Klein et al., 2008b). The maximum target of 50% has a stronger ecological basis, but is rarely used in conservation planning because it is assumed to be too politically contentious (Soule and Sanjayan, 1998). However, it should be noted that the English and French MPA agencies have developed or are developing their own targets (e.g. JNCC and Natural England, 2010) and that data-driven habitat target will be developed in the next chapter of this thesis.

5.2.4. Marxan and Zonation analyses

As described above, Marxan and Zonation use different approaches for identifying priority areas and measuring conservation value, so we needed to select methods and outputs that were most comparable. In terms of methodology, this involved choosing the following options in Zonation: (i) the target-based cell removal rule to produce the priority-area map, so that Zonation sequentially removes the lowest-value planning unit from its conservation-value map, as long as that planning unit is not needed to meet the targets for the different features (Moilanen, 2005), and (ii) the boundary-length-penalty (BLP) option, which most closely resembles the BLM factor in Marxan (Moilanen, 2007; Moilanen and Wintle, 2007).

Identifying suitable outputs was relatively straightforward, although it is important to understand the differences in the software packages. A Marxan analysis involves running the software a number of times and producing a near-optimal, but often different, portfolio at the end of each run. It then identifies the best portfolio as the one with the lowest cost, and produces a selection-frequency output by counting the number of times each planning unit appeared in the different portfolios (Ball *et al.*, 2009). In this analysis, we used the best output as Marxan's priority-area map and the selection-frequency output as Marxan's conservation-value map. Thus, Marxan's priority-area map can change between different analyses, and the extent of its near-optimality tends to increase with the number of runs used. Similarly Marxan's conservation-value map can vary between analyses, although these differences tend to be much smaller because each output is based on a number of runs.

In contrast, Zonation produces the same conservation-value map for a given set of inputs, based on the same hierarchical-ranking output, and also produces the same priority-area map for meeting the specified targets. Despite these differences, conservation practitioners use the outputs in similar ways: both priority-area maps show areas that are needed to meet the specified targets, and both conservation-value maps show the relative importance of each planning unit for meeting the conservation objectives.

We undertook nine analyses using Marxan and Zonation to run assessments based on the three different planning-unit cost metrics (Table 20) and the three different targets: 10, 30, and 50%. The Marxan analyses involved running the software 100 times, with each run consisting of one million iterations. After conducting a sensitivity analysis, we used a BLM value of 5 in all three subsequent Marxan analyses, as this best balanced efficiency and portfolio-fragmentation levels (Carpentier et al., 2009; Possingham et al., 2000), and used a target-penalty factor of 100 000 for each conservation feature to ensure that Marxan identified portfolios that met all the targets. The Zonation analyses used the target-based removal rule to identify portfolios that best met the targets, and we selected the BLP value to ensure the lowest boundary length/area value.

We used the Marxan and Zonation conservation-value maps to measure the impact of using different planning-unit cost metrics. We did this by first using a quantile classification in ArcGIS to convert both outputs into maps divided into 10 classes of equal area based on their measure of conservation value. Thus, each planning unit was given a ranking value from between 1 and 10 for both software outputs, and we then used Spearman Rank tests to determine the similarity of the outputs, although we did not record the significance values for these tests because the data were influenced by spatial autocorrelation (Balmford et al., 2001; Nhancale and Smith, 2011). We also investigated the priority-area maps produced by the two different software packages and tested for differences in total area, number of patches, and median patch size using a Wilcoxon Signed Rank test. Finally, we tested whether the priority areas selected by Zonation had higher Marxan conservation-value scores using Mann–Whitney tests.

Table 20: Parameters used in the three sets of analyses

Metric	Marxan BLM value	Zonation BLP value	Cost layer
Area	5.0	0.5	Area of planning unit
Accessibility	5.0	10.0	Area of planning unit, but reduced by 50% for inshore areas and shipping lanes
Fishing	5.0	0.0	French fishermen profitability

5.3. Results

The conservation-value maps produced by Marxan and Zonation were both strongly influenced by cost metric, with similar areas being identified as important (Figure 26). However, important areas were widely scattered when using the area metric, more likely to occur around the coast and the shipping lanes in the Dover Strait when using the accessibility metric, and more likely to occur on the English side of the planning region when using the fishing metric. Using higher targets tended to increase the number of planning units with high conservation-value scores (Figure 26). Zonation outputs generally consisted of more rectangular patches of planning units, whereas the important areas in the Marxan outputs had less regular boundaries (Figure 27; Table 21). The conservation values of the planning units calculated by Marxan and Zonation were correlated and varied with cost metric (Table 22). The results also broadly showed that correlations were higher with increasing conservation targets and when using the fishing-cost metric.

Table 21: Spatial characteristics of the portfolios identified by both Marxan and Zonation based on the three different cost metrics and targets.

Target (%)	Cost metric	Number of patches		Median patch area (km^2)		Total area of portfolio (km^2)	
		Marxan	**Zonation**	**Marxan**	**Zonation**	**Marxan**	**Zonation**
10%	Area	8	6	1315.8	3200.0	11821.4	20047.9
10%	Accessibility	11	6	384.0	3200.0	13380.6	20047.9
10%	Fishing	8	34	509.4	40.0	16334.7	19167.9
30%	Area	8	8	857.3	432.0	27855.0	25514.3
30%	Accessibility	12	3	1038.5	6015.4	26493.6	28695.8
30%	Fishing	19	27	4141.5	32.0	28810.9	31613.9
50%	Area	9	6	15.4	40.0	45333.4	54863.9
50%	Accessibility	6	4	336.0	40.0	47879.8	67583.8
50%	Fishing	9	15	384.0	16.0	21007.8	55823.9

Table 22: Spearman rank correlations of the conservation-value scores produced by Marxan and Zonation based on the three different targets and cost metrics.

Metric	10%	30%	50%
Area	0.270	0.284	0.554
Accessibility	0.249	0.394	0.133
Fishing	0.720	0.830	0.788

In general, the planning units that were identified as part of the Zonation priority-area maps had higher Marxan selection-frequency scores than those planning units that were not selected by Zonation, with the exception of the 10% targets and accessibility cost-metric analysis (Table 23). Marxan generally produced smaller priority-area systems than Zonation ($N = 9$, $Z = -2.429$, $p = 0.015$), but there was no pattern with median patch size or number of patches. There was a linear relationship between priority-area extent and targets, so that priority-area extent ranged between 11 821 km^2 for the 10% target- and area-cost metric Marxan analysis and 67 583 km^2 for the 50% target- and accessibility-cost metric Zonation analysis (Figure 28), but there was no obvious trend with number of patches and median patch size (Table 21).

Table 23: Results from Mann–Whitney tests for the Marxan selection-frequency scores of planning units falling inside and outside the priority areas identified by Zonation.

Cost metric	10%	30%	50%
Area	15.29*	12.97*	34.83*
Accessibility	0.18	20.14*	25.45*
Fishing	35.79*	52.79*	50.29*

*$p < 0.001$

5.4. Discussion

Systematic conservation planning is a widely used approach for designing MPA networks, and most planning assessments rely on computer software to identify priority areas for conservation. These software packages are based on the same principles, but generally use different approaches for measuring conservation value and selecting portfolios of planning units. This has created some confusion amongst conservation practitioners about which software to use and whether this affects the results. Our analysis investigated this issue using the Marxan and Zonation software packages and data from the eastern English Channel. In this section, we discuss whether it is possible to compare the two software packages, given their underlying differences, and go on to discuss how these results are influenced by the application of different cost metrics and targets in the analysis. Finally, we provide suggestions on how practitioners should collect and use data to minimize the influence of these software packages on their results to help produce more relevant results.

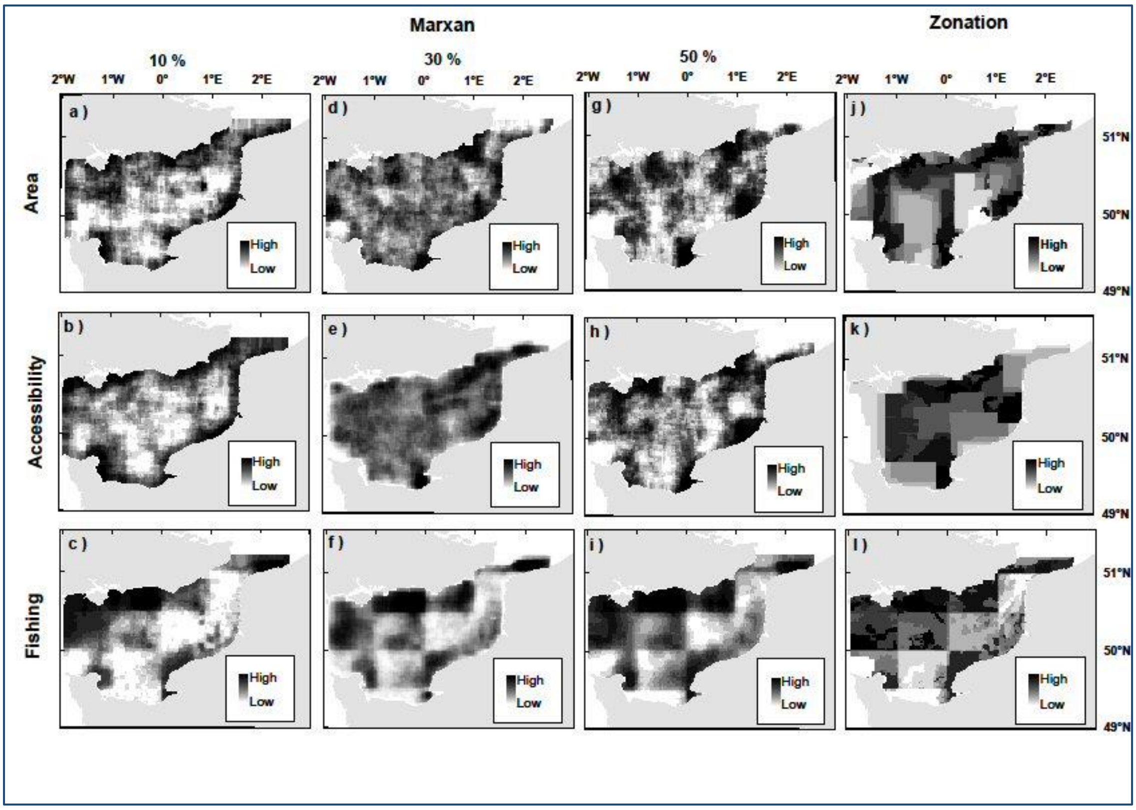


Figure 26: Conservation-value maps for Marxan (a–i) and Zonation (j–l) based on the three different targets and cost metrics. The conservation value for Marxan is based on selection frequency and for Zonation is based on the hierarchical solution output. There are only three maps for Zonation because the hierarchical solution output is a nested output and does not change when using different targets.

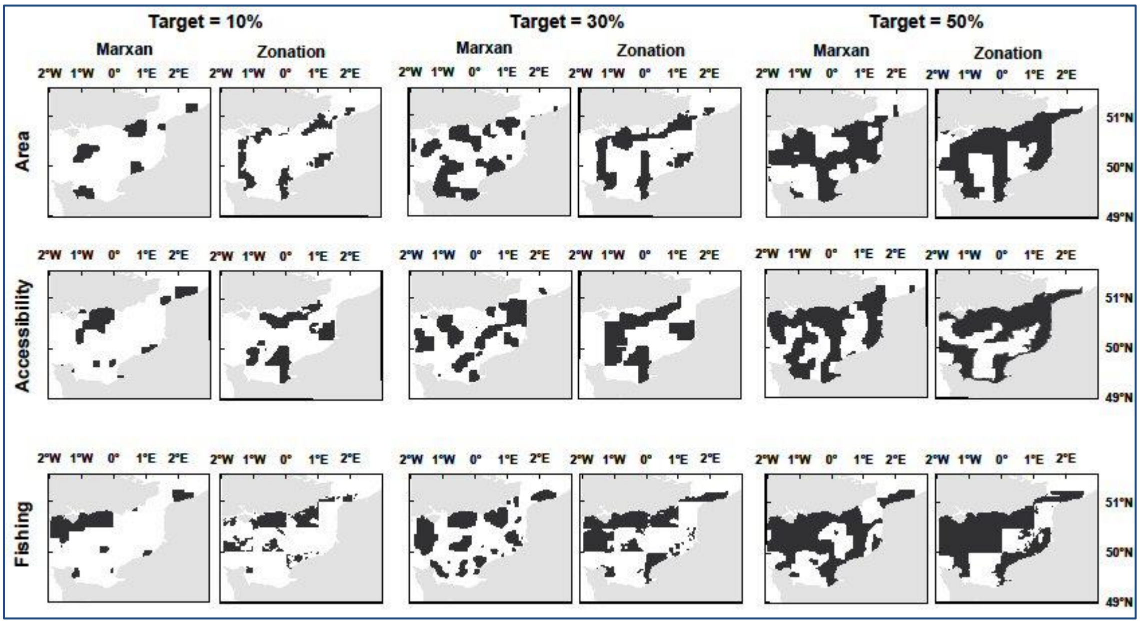


Figure 27: Priority-area maps identified using Marxan and Zonation based on the three different targets and cost metrics.

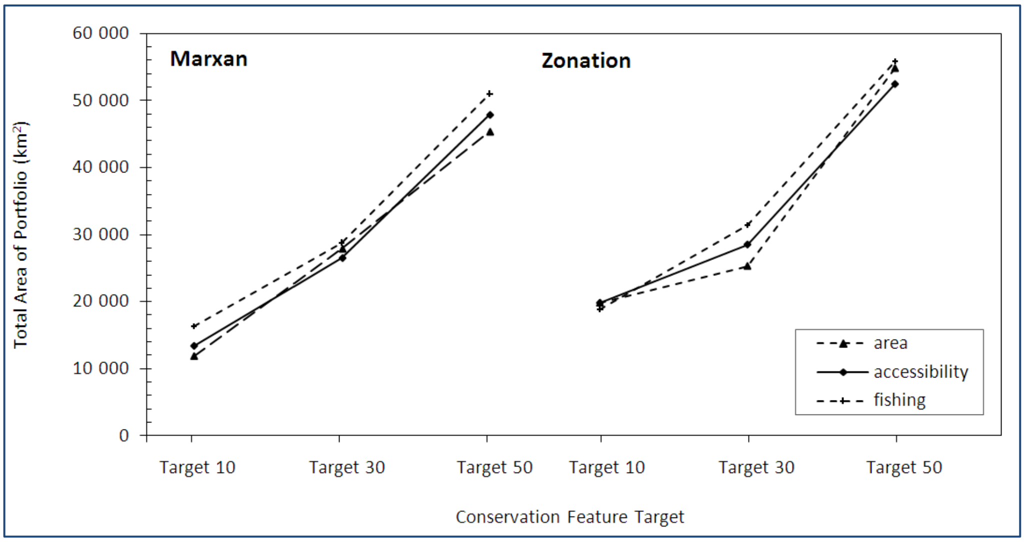


Figure 28: Area of priority areas identified by Marxan and Zonation based on the three different cost metrics and increasing targets.

5.4.1. Comparing software packages

Marxan uses the minimum-set approach to identify priority areas for meeting specific targets, whereas Zonation uses the maximum-coverage approach to identify priority areas given a fixed budget. Despite this, the software outputs can be compared because Zonation can adapt its ranked hierarchy output to identify the best areas for meeting targets, which it does by sequentially removing the least important planning units until further removal impacts target attainment. However, this ranked hierarchy output is based on the maximum-cover approach, so it will always be impossible to make exact comparisons between the two packages. Moreover, this comparison is further complicated by the different spatial constraints used by the software packages and the difficulty in determining equivalent BLM and BLP values. We used a standard approach for determining both sets of values, based on balancing the relative planning unit and boundary-length costs, but it is likely that their influence on the results differed.

It should also be noted that the conservation features and targets that we used in the analysis were designed to emphasize any differences in the results from the two software packages. This is because most of the conservation features were widely distributed, and the targets were never more than 50% of these distributions. Thus, there was a large amount of flexibility in the planning region, with no planning units always being needed to meet certain targets, and many planning units having similar conservation value. In such scenarios, it is likely that the spatial constraints would have a relatively large influence on which planning units were selected; therefore, differences in the way that the spatial constraints are used may have produced these effects. This is in contrast to previous work comparing results from Marxan and C-Plan, another conservation planning package, which used higher relative targets and included no spatial constraints and found that conservation-value outputs were very similar (Carwardine et al., 2007).

Despite these differences, it should be noted that the two software packages still produced similar results. Whilst the priority areas identified were not identical, which was expected given the flexibility in the system, there was definite overlap, and the Zonation priority areas had significantly higher Marxan selection-frequency scores in almost all the scenarios that investigated the influence of cost metrics and targets (Table 23). Moreover, the strength of this similarity increased when using real-world cost data, such as the accessibility- and fishing-cost metric. This was because using these cost metrics reduced flexibility so that planning units with similar biodiversity value differed in terms of cost, making low-cost units more important (Smith *et al.*, 2008) and more likely to be selected by both software packages (Table 22). Priority-area extent increased with increasing targets, but this relationship was more linear with Marxan than with Zonation. This may be because Zonation tended to select larger and more connected patches, although some of the Zonation outputs also included a number of small fragments, which masked any difference in patch size and number when comparing Marxan and Zonation. Thus, we found that Marxan tended to produce more efficient priority-area networks and Zonation produced networks that had higher levels of connectivity.

5.4.2. Implications for designing MPA networks

Our analysis identified three broad aspects that can help inform marine conservation planners when deciding what types of data should be included in their conservation assessments and what type of software they should use. First, we found that, although making direct comparisons between Marxan and Zonation was not straightforward, the results were not highly affected by which software package was used and that the differences were reduced when using real-world cost data. Thus, conservation planners should select the software they consider most appropriate, based on the aims of the project and the additional functionality of the different packages. Second, we found that the conservation-value scores of most of the planning units used in our analysis were generally low, which was probably the main reason for the differences in the results from Marxan and Zonation. This arose because most of our conservation values were widely distributed and the targets were relatively low, which meant that there were many similar planning units and, hence, a great deal of flexibility in which ones were selected.

The third main finding echoes that from previous studies, which shows that the type of planning unit cost-metric plays a large role in determining the location of the priority areas (Ban and Klein, 2009;

Klein et al., 2008a). We found that using real-world data not only produced more robust results, as described above, but it also significantly shifted the location of the areas selected by Marxan and Zonation. Thus, using the accessibility cost meant that most priorities were found around the coast and in major shipping lanes, whereas using the fishing cost meant that most priorities were found on the English side of the planning region. This highlights the importance of choosing an appropriate cost metric when developing a conservation-planning system to inform decision makers in the region. However, this is likely to be challenging given that not only do multiple nations share access to the same resources, but the English Channel is commercially important for fishing, transport, aggregate extraction, and energy production sectors (Martin et al., 2009).

Our fishing-cost metric also highlights the problems of using direct financial value in conservation assessments, as this can overly impact marginalized groups (Adams et al., 2010). In this case, the English fishing fleet consists of fewer and smaller boats, so establishing MPAs in English waters would have a smaller impact on the financial value of the catch. However, establishing more MPAs in English waters would have large impacts on the local economies and societies, and any plans advocating such changes would be politically untenable. Thus, our results confirm evidence from a number of studies which show that the value and success of conservation assessments generally depends much more on understanding and reflecting the social conditions found in a planning region (Smith et al., 2009) rather than on the type of selection algorithm or conservation-planning software used. The chapter six of this thesis will more deeply explore the cost-metric influence on the MPA design investigating some French and English fishing efforts data and landings.

MPAs are now expected to be possible management tools in the context of ecosystem-based management of fisheries (Pauly et al., 2002). Although some findings relating to coral reefs led to recommendations that 20–30% of each marine habitat should be closed to exploitation (Hughes et al., 2003; Roberts et al., 2003a), there are many types of MPAs, with management arrangements ranging from multiple-use to strict protection within “no-take zones”. In complex systems such as the English Channel or the North sea, MPA networks should be designed with different levels of conservation management (Watts et al., 2009) to enable a full MSP exercise. Finally, and more importantly, developing a coherent MPA network for areas shared amongst many countries will need to move away from current national approaches, which are limited to the Exclusive Economic Zone, and to work on a scale that is relevant to the eco-region. This requires international collaboration and shared access to both biological and socio-economic data, which was the approach adopted in this thesis.

6. Chapter Six.

Cost representation in conservation planning

Abstract

A common objective in designing Marine Protected Areas networks is to achieve a range of conservation targets whilst minimizing impacts on the sea users. The eastern English Channel is a congested sea where a variety of sectors of activity already compete for space: fisheries, aggregate extraction, wind-farms, shipping, leisure, and could in addition be constrained by conservation areas. Moreover, the EEC is in a trans-boundary context which complicates any marine planning action. Here, we integrate human activities in a MPA network design process. Integration of fishing as an opportunity cost considerably modified those Marxan outputs used to inform the MPA network design process. Moreover, Marxan outputs were sensitive to the variable used to parameterize the cost opportunity layer, fishing hours or revenues. While it appeared important to integrate fishing activities in the conservation planning process, the nature of the data used to reflect fisheries opportunities may have a profound effect on results. Consider other human activities in the EEC. Wind-farm sites were considered to favor ecosystem conservation, because of the probable fishing restrictions which could occur in those areas. By contrast, aggregate extraction sites were considered as not suitable for conservation purposes. The addition of other human activities in the overall opportunity cost had a lesser influence on the MPA network design process, compared to the integration of fishing activities. I propose two cost equations combining all the main human uses which could be used as opportunity cost for conservation planning in the EEC.

Keywords: English Channel, conservation cost, conservation planning, fishing effort, fishing revenues

6.1. Introduction

To be able to conserve marine biodiversity, MPAs locations should be based on solid ecological considerations (Roberts et al., 2003a). However, they should also take into account the needs and interests of all stakeholders exploiting the maritime domain, to reach a satisfactory acceptability (and compliance) level (Naidoo et al., 2006). Systematic reserve-selection tools such as Marxan allow simultaneous optimization for ecological objectives while minimizing costs. If much attention has been devoted to the biological aspects of this process, the cost effectiveness should not just represent ecological considerations. Indeed, while biodiversity is not evenly distributed over the study area, the spatial variability of costs may be as important and socioeconomic considerations should be explicitly considered (Carwardine et al., 2008; Naidoo et al., 2006).

When cost is intended to reflect the socio-economic impacts of conservation areas, the use of inappropriate cost measures may lead to serious conservation mistakes: Carwardine et al (2008) found that biodiversity targets can be up to twice as much expensive to achieve if incorrect cost data are used. The issue of socioeconomic data resolution and their influence on spatial planning has to be investigated. This is particularly critical in an area such as the EEC, the exploitation of which provides an important source of revenue for different sectors of activity, and especially fisheries. Richardson et al (2006) first explored the impact of using fine resolution fishing activity data (fishing grounds identified by fishermen during interviews) compared to the coarser official statistics (landings by ICES sub rectangle) on reserve-design. They found that reserves based on coarse data could be most costly to the fishery in term of losses in commercial fishing revenues. Other studies also included fisheries in the opportunity cost (Box 3) in conservation planning scenarios (Ban and Klein, 2009). The opportunity cost was then reflected by the Catch Per Unit of Effort (Klein et al., 2008a; Klein et al., 2008b; Lombard et al., 2007; Richardson et al., 2006) or other spatially-explicit estimations of the fishing profitability (Game et al., 2008; Klein et al., 2009a; Stewart and Possingham, 2005). Few studies considered recreational or even small-scale fisheries mostly because data are rarely available (Ban and Klein, 2009; Klein et al., 2008a). Fishing is the most prevalent human activity in the marine environment (Pauly et al., 2002) and is of first importance in the EEC. In 2005, 696 French and English fishing vessels were active in the area and French landings represented 218 M€ (Carpentier et al., 2009). However, a lot of small fishing boats (<15m) are active in the EEC and are not here considered because of the lack of accurate data. Therefore, only those vessels exceeding 15 m were considered to parameterize the opportunity cost (Box 3).

Since the aim of adding a cost layer is to minimize the impact of the proposed MPA network on human activities, the influence of the data information and resolution on the resulting MPA network should be investigated in depth. Here, we will first consider the fishing activity and develop MPA networks with a minimal impact on fishermen. This was allowed using georeferenced fishing effort data in a first time, which were then associated to fishing landings and gross revenue for a more economic representation of the activity. These data were extracted and/or derived from mandatory English and French logbooks, sale slips, and available VMS records. Second, other human uses of the EEC such as aggregate extractions and the wind farm implementations were explored as possible terms of the cost calculation. Then, all the different cost combinations have been tested as costs in Marxan scenarios.

Box 3: Different kind of conservation costs (adapted from Naidoo et al, 2006 and Ban & Klein; 2009)

Acquisition costs

acquisition costs are costs of acquiring property right to a parcel of land. These costs are atypical in the marine environment because waters are not usually privately owned.

Management costs

Management costs are those associated with management of a conservation program; such are those associated with establishing and maintaining a network of Marine Protected Areas.

Transaction costs

Transaction costs are those associated with negotiating the conservation, such as the time and staff involved in stakeholder negotiations.

Damage costs

Damage costs are those associated with damages to human activities arising from conservation activities. The review by Ban & Klein (2009) did not find any mention of damage costs due to marine conservation.

Opportunity costs

Opportunity costs are costs of foregone opportunities *e.g*. the value to fisheries and other marine uses in the marine environment. These kinds of costs are the most usually used in MPAs implementations studies.

6.2. Materiel and Methods

6.2.1. Parameterization of the cost function

Fishing activities

Fishing effort was here represented at a fine resolution scale by satellite-based Vessel Monitoring System data (VMS). From 1998, the European commission introduced legislation to monitor fishing vessels for security control and enforcement purposes using the VMS. From January 2005, all vessels over 15m in length are required to transmit their position at interval of 2h or less (European commission, 2003). Nonetheless, if VMS data are used to describe fishing activity, the transmitted data do not indicate whether vessels are fishing or not (Lee et al., 2010; Vermard et al., 2010). Several methods have been developed and applied to VMS data to obtain estimates of fishing effort (see Lee et al 2010 for a review). Here, only the 2007 English data and an aggregation of the 2008 French data were available to our study. English data were available on a 0.05° grid for all UK fishing zones for 8 fishing gears categories (Lee et al., 2010). French data were available monthly on a 0.17° (10') grid. For consistency purposes, UK data were summed over the closest French grid node with a maximal distance of 0.12° and French data were aggregated in UK gear type categories. The joined dataset extent was limited to the area where UK and French datasets overlap. We assumed here that 2007 and 2008 fishing efforts were comparable, regulations remained the same during the two years and no main changes in fish stocks were found. In a second step, logbook and sale slips data were used to derive the landings value. These data are available at the ICES area scale (30' x 60' rectangles, Cf. part 1.4.3, Figure 12), by fishing vessels and fishing trip, and as a part of the CHARM project were available for both UK and France in 2007 and 2008 respectively. To facilitate the combination of landings and fishing effort data the different fishing gears were grouped in three categories based on their gear. These gears do not have the same impact on the environment and especially on the seaground (Kaiser et al., 2000). The first group corresponds to towed gears and encompasses beam trawls, demersal otter trawls and dredges. Mid-water trawls were considered as the second group. The third group consisted of passive gears, such as nets, pots and hooklines.

Aggregate extractions

Aggregate extractions areas were also considered to parameterize the cost layer (Figure 29). We used both actively extracted areas and zones not exploited yet but where dredging concessions were granted in French and British waters. The location of current and potential aggregate extraction areas were supplied by the Crown Estate on the English side and from IFREMER on the French side. Aggregate extractions areas were considered as not suitable for conservation purposes.

Wind farms

Potential wind farm sectors were also added to the cost layer (Figure 29). We made the assumption that fishing would be restricted in areas occupied by wind farms. Therefore, in contrast with aggregate extraction sites, wind farm sectors were considered as an area to promote in the MPA design process.

Other factors

Finally the 3 miles area was also treated as a beneficial factor for the MPA implementation because of its proximity to the coast, which is assumed to facilitate management and where trawling activities are supposed to be banned. Shipping lanes and ferry routes were not considered as in chapter Five because here their inclusion would have been redundant with the use of fishing effort which should already be less important in these specific areas.

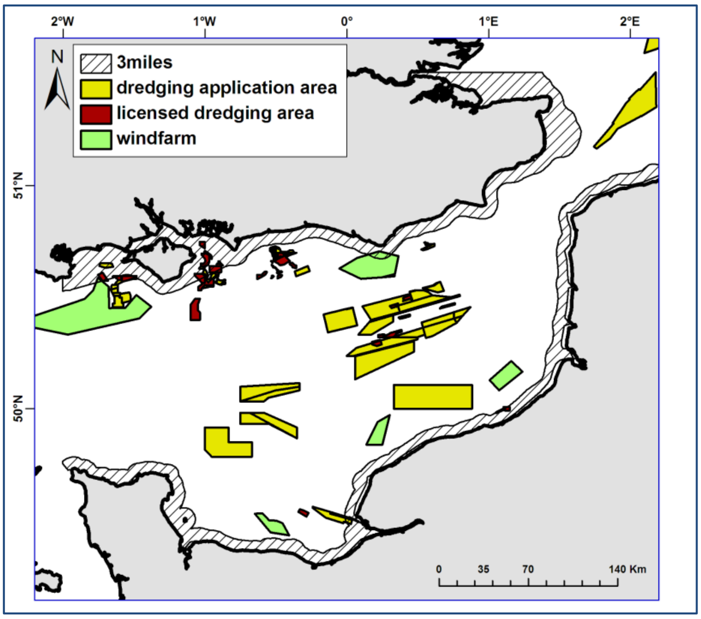


Figure 29: The other uses considered in the analysis: the dredging areas, the wind farms sector and the 3 miles coastal area

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6.2.2. Calculation of the cost metric equation

Marxan aims to meet a series of targets whilst minimizing its cost which is expressed as a “cost metric” (Cf. Part 1.3.4). This cost metric needs to be represented by one value per planning unit. To calculate this cost metric I developed here different equations considering the various economic data previously described. I started by exploring the influence, on the Marxan output, of each economic parameter taken individually as a cost value. The analysis of these results resulted in the proposition of two cost equations.

The first step in the cost metric analysis was to explore the impact of using three different proxies to mimic the relative importance, for fishers, of the different areas they exploit. A first analysis used the spatial distribution of fishing effort (in hours fished) available on the 10’ x 10’ resolution grid. Second, the Value (of all species landed) Per Unit of Effort (VPUE) was considered as a proxy for fishing profitability, and it has been estimated for each square i of the 10’ x 10’ resolution grid (Figure 30). If fishing effort is in hours, VPUE are expressed in €. h^{-1}. Two calculations were tested to express two hypotheses on fishing profitability. The first method assumed that landing value is evenly distributed over each ICES rectangle (equation (5). The landing values were divided by the number n_k of sub-divisions i_k existing in each considered ICES rectangle k and the VPUE at the 10' x 10' scale were inversely proportional to the number of hours spent at that scale. This VPUE calculation hypothesises that VPUE decreases when effort increases and reflects increasing exploitation cost and decreasing benefits. In the equation (6) the VPUE was first computed at the scale of the ICES rectangle k (this corresponds to the blue part of the equation) and then was re-allocated to the smaller squares i_k (corresponding to the 10'x10' fishing effort resolution) proportionally to effort distribution. With this calculation, the VPUE of each square i_k is proportional to fishing effort in i_k, thus, unlike in the equation (5), VPUE increases with effort. This reflects increasing exploitation benefits resulting from fishing on high fish aggregation or targeting fish with more commercial value.

(5) $VPUE_{i,k} = (\text{Landing value}_k / n_k) / \text{effort}_{i,k}$

(6) $VPUE_{i,k} = (\text{Landings value}_k) / \Sigma \text{effort}_{i,k}) * (\text{effort}_{i,k} / \Sigma \text{effort}_{i,k})$

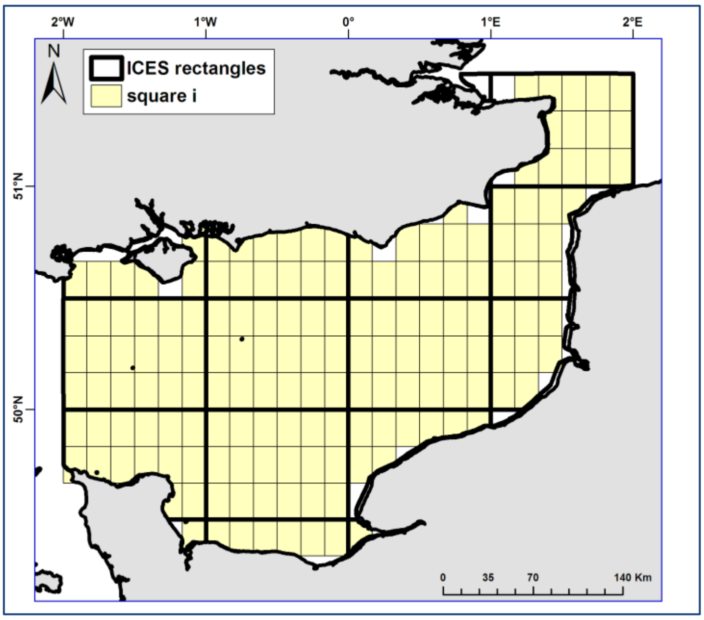

Figure 30: ICES rectangles in the eastern English Channel and the squares i which correspond to the 10' resolution grid for which VPUE values were calculated

VPUE were calculated for each gear type and each country, in 2007 for UK and in 2008 in France. The calculated VPUE and fishing effort were then recalibrated between 0 and 1 to ensure consistency across the different components of the cost layer (Legendre and Legendre, 1998; Sneath and Sokal, 1973). The cost layer was represented at the 10' latitude resolution, consistent with fishing effort data.

Different cost equations were then defined to study the relative influence of each gear type on the cost metric and then on the resulting Marxan output. The standardized VPUE were compiled in a cost equation where each gear type was weighted. The same weights were given for the two countries for the same gear type. Four possibilities were tested: Three giving a maximum weight to one of the gear types and a fourth one giving a similar weight to the three gear categories. To facilitate the interpretation of the results these different equations were named from cost one to cost twelve as described in Table 24, where VPUE 1 and VPUE 2 refer to VPUE calculated with the equations (5) and (6) respectively.

Table 24: The different cost equations explored in the sensitivity analysis. 0.1, 0.8 and 0.33 are the weighting factors applied to the three groups: BOTTOM refers to the bottom gears group, PELAGIC to the midwater trawls and PASSIVE to the passive gears. Fishing effort data are in hours and VPUE are expressed in € h^{-1}

Fishing proxy/ calculation	Fishing effort	VPUE 1	VPUE 2
0.8*BOTTOM + 0.1*PELAGIC + 0.1* PASSIVE	Cost 1	Cost 5	Cost 9
0.1*BOTTOM + 0.1*PELAGIC + 0.8* PASSIVE	Cost 2	Cost 6	Cost 10
0.1*BOTTOM + 0.8*PELAGIC + 0.1* PASSIVE	Cost 3	Cost 7	Cost 11
0.33*BOTTOM + 0.33*PELAGIC + 0.33* PASSIVE	Cost 4	Cost 8	Cost 12

The second step consisted in exploring the influence of the other human uses and of the 3 miles coastal area on the Marxan runs (Figure 29). The spatial overlap between each Planning Unit and the aggregate extractions sites and wind farms areas was calculated. To implement these parameters as a cost metric, the planning unit area was first computed. Then, for the dredging sites, the area was multiplied by the percentage of overlap*100 for the concerned Planning units to increase their related cost. The area was then divided by the overlap percentage with the wind farms areas and the 3 miles zone to decrease the cost. Finally the three obtained costs were re-scaled between 0 and 1.

6.2.3. Marxan analyses

A planning unit theme of 0.05° mesh size was used to run Marxan with the different cost equations. The benthic and pelagic habitat types (Figure 17, Chapter Three) were used as conservation features with a 20% target for each habitat. No habitat specific targets were used in this chapter to keep the focus on the cost sensitivity analyses. In each case, Marxan was run 100 times with a Boundary Length Modifier of 100 (a sensitivity analysis to choose the BLM value was run). A first analysis was run using the Planning Unit area as the cost. Second, the 12 costs reflecting the fishing activity were added. Finally, aggregate extraction sites, wind farm areas and the 3-miles zone were also used. The selection frequency outputs (conservation value maps) are presented for each case. A Spearman rank correlation (Best and Roberts, 1975) was calculated to allow an interpretation of the differences between the outputs (Tables 25, 26, 27).

Having explored the influence of the fisheries and other human's uses one by one, I considered the obtained results to select those two cost equations which would the best represent stakeholders' interests in the EEC. These two cost equations were then used in Marxan scenarios.

6.3. Results

6.3.1. Producing the cost layers

The twelve cost equations described in Table 24 were calculated and applied to the 10x10' resolution grid and all results were recalibrated between 0 and 1. Within each fishing cost proxy (fishing effort or VPUE1 and VPUE2) the cost spatial pattern was influenced by the weighting of the gear types (Figures 31-33). For example, in Figure 31, the "cost 2" scenario with a high weight on passive gears illustrated an effort distribution (more hours spent) in very different areas than those obtained with other gear dominance. This highlighted that passive gears fishing boats had very different fishing distribution than fishing vessels rigged with other gear types. Similar variations in fishing activity patterns were also found with the "VPUE 1" and "VPUE 2" scenarios (Figures 32 and 33). However, the maps showed that the spatial patterns were not comparable between the three fishing proxies. Furthermore, for similar gear weighting, the spatial distribution of cost differed strongly depending on the fishing proxy used.

The patterns obtained with the costs 1, 5 and 9, where the bottom gears had more weight, were highly variable. Considering the fishing effort as a proxy for fishing activity (Figure 31), the area between 1°W and 1°E on longitude and 50°N and 51°N in latitude provided the higher cost. Now when VPUE were considered as the proxy for fishing activity, this area did not arise as the most valuable one for bottom gear fisheries. A different pattern was obtained with the second VPUE calculation procedure: both the French coastal area and the Dover Strait prevailed over other EEC areas. Dissimilarities also arose when the passive gears were favoured (cost 2, 6 and 10). In particular, when using the VPUE 2 proxy (Figure 33, cost 10) all cells had really low values (inferior to 0.1) but few coastal locations.

Therefore, spatial variations in the fishing cost layer depended both on the fishing gears type and the proxy used to derive the metric.

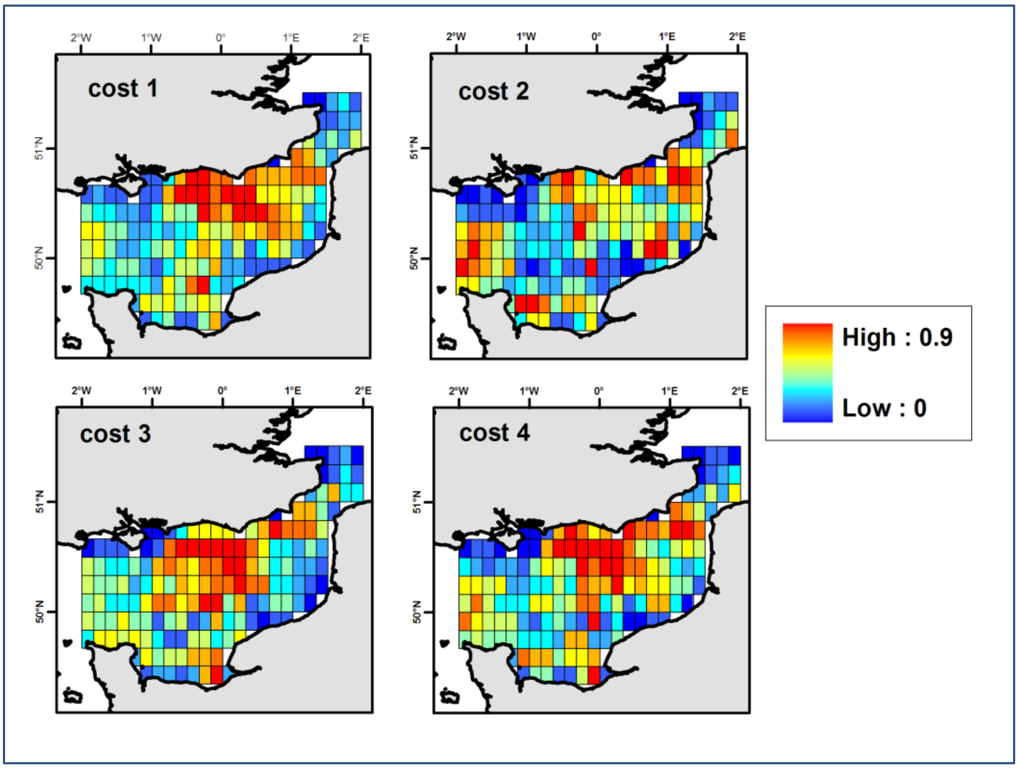

Figure 31: Cost layer calculated with the fishing effort as a fishing proxy. Costs 1 to 4 refer to the cost equations defined in Table 23

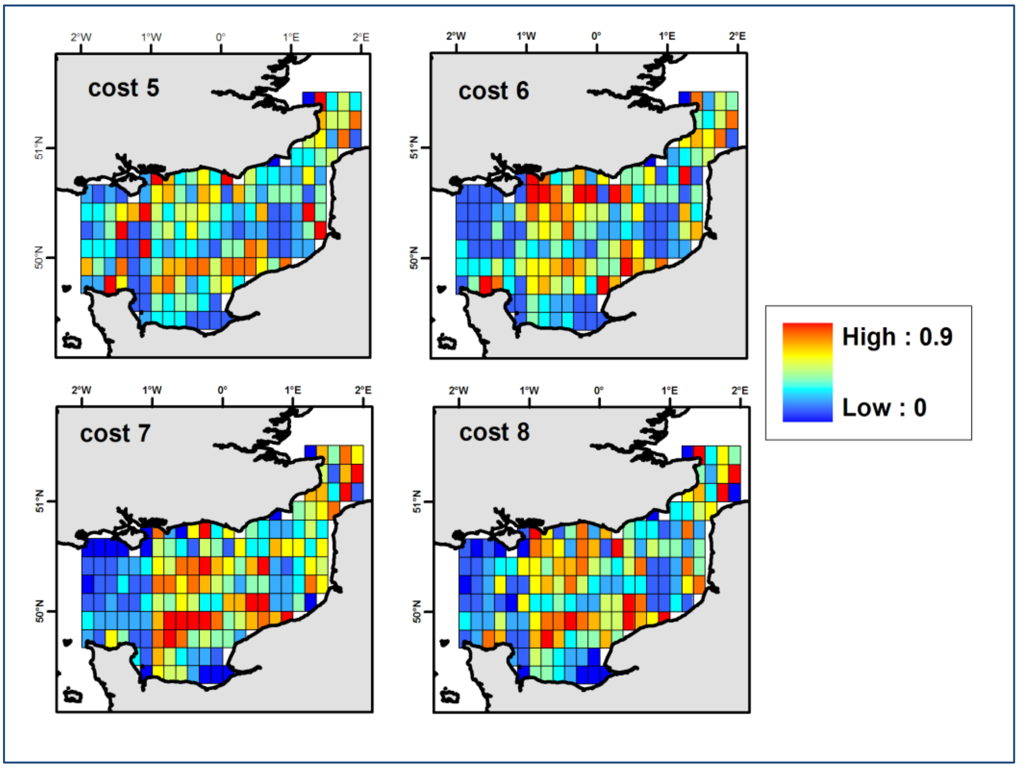

Figure 32: Cost layer calculated with the VPUE 1 calculations as a fishing proxy. Costs 5 to 8 refer to the cost equations defined in Table 23

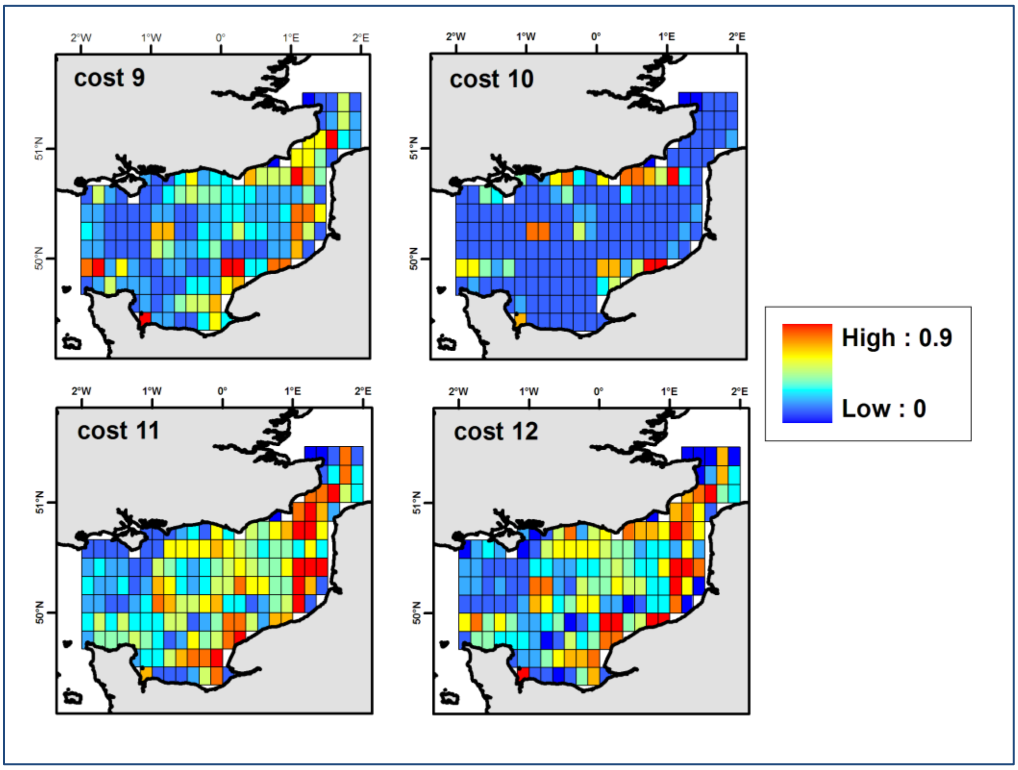

Figure 33: Cost layer calculated with the VPUE 2 calculations as a fishing proxy. Costs 9 to 12 refer to the cost equations defined in Table 23

6.3.2. Marxan results using fishing activities as cost layers

The second part of these results investigated the influence of using one or the other cost equation in a Marxan run. The conservation value map produced without integration of economic data was first considered (Figure 34). The cost value used in this case was that of the surface of each planning unit. The selection frequency pattern produced to reach the 20% target for each habitat showed a preference for the coastal areas. The benthic habitats were more precisely described on the coast with a lot of small level 4 habitats requiring a higher selection frequency to adequately represent the diversity of these zones. However, this pattern could also be an artefact resulting from some truncated PUs being located at the edge of the study area and thereby having reduced surface and cost. The Dover strait, the Northern and Southern Bay of Seine, and the coast around Beachy Head, were the zones where the selection frequency was the highest. The Western limit of the study area also came out with a high selection frequency even when all PUs had fixed equal costs. Hence, this effect was induced by the calculation of the boundary cost (Cf equation 1) where the external edges of these units are not considered in the total boundary length of the proposed Marxan solutions. This calculation method is relevant for the units representing a natural boundary such as coastal units but may induce higher selection frequency at the PU theme limits. Still this setting was preferred as most of the PU theme area was mainly boarded with coasts. This Marxan output served as reference layer to be compared to the ones produced using information on human activities and exploitation (Figures 35 and 36).The boundary cost was calculated with the same parameters for all others Marxan solutions so the comparison is not affected by this increased selection on the western limit of the study area.

The different conservation value maps depended on both the cost equation and the fishing proxy considered (Figure 35). However, the fishing proxies appeared to be the most influent parameter. Using the same weights for the three gear types, the selection frequency outputs were highly different with Spearman correlation coefficient ranging from 0 to 0.43 (Table 25). When comparing these outputs to the initial reference output (Figure 34), the introduction of fishery activities in the cost layer was found to strongly modified the conservation values maps, with Spearman rank correlation coefficient around 0.2 (Table 26). Interestingly, the outputs obtained with cost 4, where the three gear types were equally weighted, was well correlated to the three other outputs. This resulted in conservation values maps where the cost 4 outputs (Figure 35 d, h and l) showed high selection frequency in all the important spots arising from outputs using cost 1 to 3.

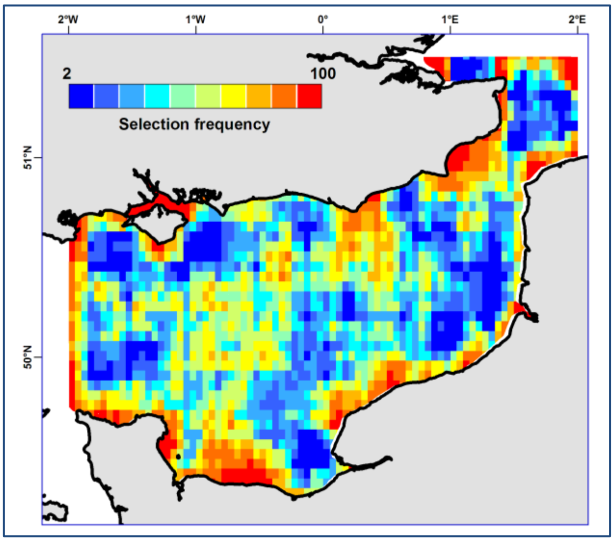

Figure 34: Conservation value maps produced with Planning Unit cost equal to its area. Selection frequencies range from 2 to 100

Table 25: Spearman rank correlation coefficients between the Marxan conservation values maps calculated with cost 4, 8 and 12 (the three equivalent cost equations calculated with three different fishing proxy). All p-values are <2.2 e^{-16}

Equation cost	Cost 4	Cost 8	Cost 12
Cost 4	-	- 0.07	0.43
Cost 8	-	-	0.14

Table 26: Spearman rank correlation values between the Marxan conservation values maps calculated with cost 1 to 4 (using VMS as a fishing proxy) and the output using no cost equation (cost = area). All p-values are <2.2 e^{-16}

Equation cost	Cost1	Cost 2	Cost 3	Cost 4	Cost = area
Cost 1	-	0.65	0.48	0.78	0.20
Cost 2	-	-	0.32	0.72	0.22
Cost 3	-	-	-	0.72	0.16
Cost 4	-	-	-	-	0.21

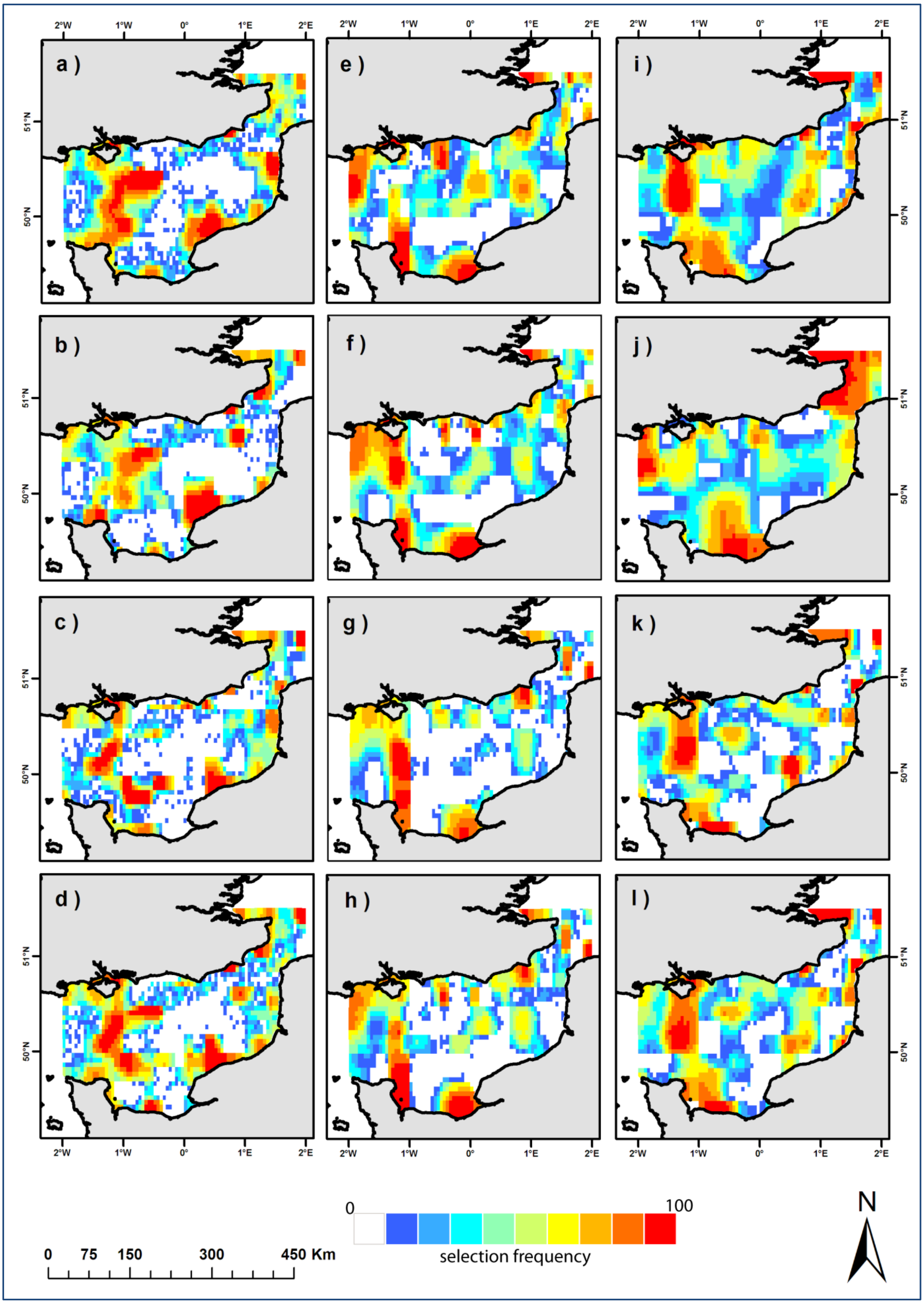

Figure 35: Conservation value maps produced with different costs, from cost 1 for a) to cost 12 for l). Selection frequencies range from 0 to 100

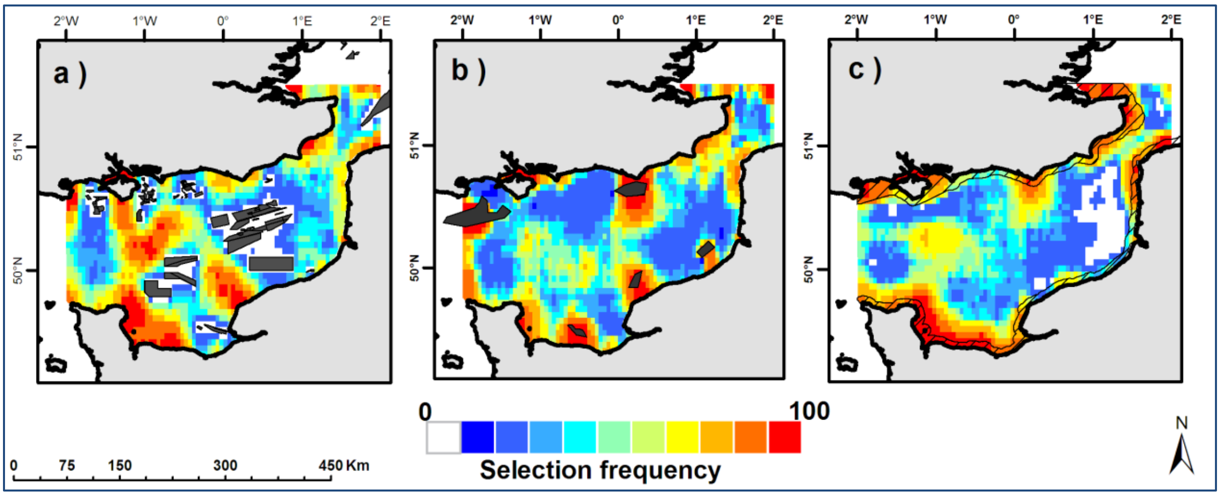

Figure 36: Conservation value maps produced with other human uses. Selection frequencies rank from 0 to 100. a) for the aggregate extraction areas, b) for wind farm areas and c) for the 3 miles coastal area.

6.3.3. Marxan results using other human activities as cost layers.

The third part of the results focused on the integration of other human uses in the cost equation (Figure 29). The conservation value maps (Figure 36) should be compared to the one without any cost equation (Figure 34). If the inclusion of aggregate extraction areas as inappropriate conservation zones induced an obvious change in the outputs, with a null selection frequency in the concerned areas, the addition of windfarm areas and the 3 miles zone led to similar patterns than that obtained in the reference case (Figure 34) but with higher selection frequency (Figure 36). These observations are supported by the Spearman correlation coefficient values (Table 27). The patterns were found to be similar to that without economic data.

Table 27: Spearman rank correlation values between the Marxan conservation values maps calculated with aggregate extraction areas, windfarm potential areas, the 3 miles coastal areas, and the output using no cost equation (cost = area). All p-values are <2.2 e^{-16}

Cost equation	Dredging areas	Wind farm areas	3 miles coastal areas
Cost = area	0.44	0.58	0.53

6.2.4. The two final cost equations and their integration in Marxan

The main goal of this sensitivity analysis on the cost layer in conservation planning was to select the cost equation that best reflect human activity and exploitation in the EEC. We had the opportunity to explore different fishing activity proxies grouped by gear types. The results of this investigation showed a strong influence of the type of fishing proxy used. The fishing effort (in hours) being the most representative of the vessel distribution (although not of their profitability) was chosen to represent the fishing activities. As a compromise between the different metiers present in the EEC the fishing cost equation with equal weights given to each gear category was chosen. The importance of the effect of the exclusion of aggregate extraction sites and the suitability of wind farm areas in Marxan outputs led to the inclusion of these costs in a second step.

Therefore, in addition to setting the cost equal to the area (basic choice), we selected 2 cost equations.

Case 1) PU cost= 0.33*bottom + 0.33 * pelagic + 0.33* passive. (In hours)

Case 2) PU cost= (0.33*bottom + 0.33 * pelagic + 0.33* passive) * (PU% of aggregate extraction areas) / (PU% of windfarms areas)

These two cost equations were integrated in Marxan (Figure 37) to compare the selection frequency with the solutions obtained without any cost (Figure 34). A Spearman rank correlation test was run between the two Marxan selection frequency outputs produced with the two selected cost equations and the one without any cost. The Spearman correlation was equal to 0.17 for the case 1 and 0.16 for the case 2. Between the case 1 and the case 2, the Spearman rank correlation was equal to 0.72.

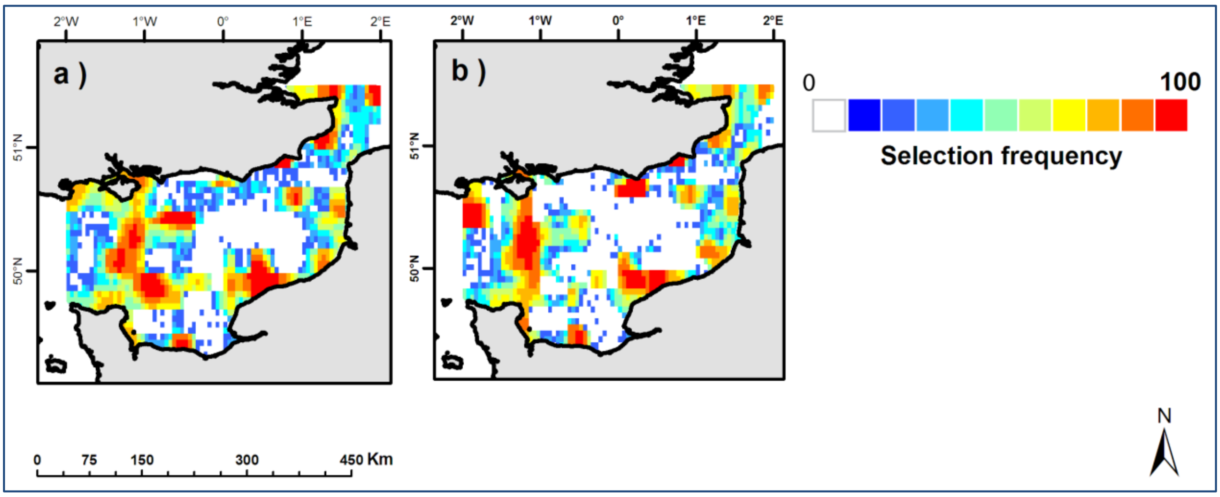

Figure 37: Conservation values maps produced with the two final cost equations. Selection frequencies rank from 1 to 100. a) for case 1 and b) for case 2

Discussion

The first noticeable result was that in all scenarios ran with Marxan in this sensitivity analysis; all the conservation targets were met. These results, confirming the ones from Chapter Three and Five (i.e.

habitat conservation targets were nor really restrictive), indicated that when a large panel of options of MPA network exists to reach conservation targets, the focus had to be set on informing the conservation planning process with relevant human activity data (Carwardine et al., 2009; Naidoo et al., 2006). It is known that economic data may have a large impact on the location of selected areas for conservation (Stewart and Possingham, 2005). The preliminary results obtained in Chapter Five already illustrated the influence of fishing data on conservation planning in the EEC. However, the spatial resolution of these data was that of an ICES rectangle, which is really coarse, compared to the PU size. This part of Chapter Six explored data with an improved spatial scale (10’ x 10’ grid). Fishing opportunities were estimated by either fishing effort in hours or Value per Unit Effort (VPUE) in Euros per hours. Two ways of calculating VPUE based on two hypotheses about fishing profitability were explored and it appeared that its calculation influenced considerably the resulting distribution of VPUE over the EEC.

Fishing data were available for different gear types. These gear types were weighted differently in the cost equations. Thus, a cost equation where the bottom gears were dominant meant that the proposed MPA network would minimize the impact on the trawlers. However, bottom trawling causes physical disturbances of benthic habitats and leads to reductions biomass and species-richness of benthic macro invertebrates (Hiddink et al., 2011; Hinz et al., 2009; Jennings and Kaiser, 1998; Kaiser et al., 2000). It is also known to be much less selective than passive gears and to generate more discards (Jennings et al., 2000). Therefore, a cost equation giving higher weight to passive or pelagic gears may be ecologically more justified as they have lesser detrimental impact on the seabed and may be more selective: The resulting MPA network would be set to limit overlapping with these activities, but would have less consideration for the impacts on the bottom gears fisheries.

The different gear types did not target the same species and so, had different fishing patterns. It was obvious when looking at the fishing effort maps, where for example vessels using passive gears such as nets seem to spend much more time in coastal areas and in the centre part of the Channel, off the Cotentin peninsula. However, these differences were less noticeable when considering the VPUE 1 maps. In this case commercial benefits are locally decreasing with the number of hours spent on the fishing grounds. However, this also could reflect the fact that the species targeted in these areas probably did not have a high commercial value. Oppositely, some coastal areas, where species with high commercial value can be found, had higher cost values when using VPUE, in particular when VPUE is made to be proportional to local effort (VPUE2). For example the 28F0 rectangle, north of the bay of Seine arose when using VPUE 2 (Figure 33, cost 10) and this is certainly due to the whelk fisheries occurring in this sector. The two calculations of VPUE rely on two opposite hypotheses about relationships between fishing effort and fishing profitability. In fact, this relationship may vary depending on the targeted species, the considered type of fishing vessels and the fishing behaviours.

When implementing these cost equations in Marxan, the conservation value outputs showed that the fishing proxy used, more than the weight attached to the gear type, had a strong effect on the selection frequency. The inclusion of fishing activity in conservation planning plans was shown to lead to poor quality or expensive conservation networks when unreliable or coarse data are used (Ban and Klein, 2009; Carwardine et al., 2008). The mismatch in the spatial patterns obtained with the different fishing activity proxies needs further investigation. VPUE is the more prevalent type of exploitation data used in marine conservation planning (Adams *et al.*, 2011; Ban and Klein, 2009). Here, VPUE were calculated at a 10' mesh size but combining fishing efforts with this resolution and landing data at the ICES area scale. Moreover the two different ways to compute the VPUE proxies also resulted in contrasting results highlighting the underlying instability of the use of these data. The two hypotheses made for the VPUE calculations can be considered as two extreme cases and a relevant VPUE representation should certainly consider more parameters depending of the considered fishing vessels and metiers as well as the targeted fish. Moreover, these investigations should be done at the finest possible spatial scale and maybe with more detailed definition of fishing gears and metiers. Therefore, since important – and yet unverifiable – assumptions about the profit distribution in each ICES rectangle had to be made to derive those VPUEs, the straightforward use of fishing effort per hour data was preferred as fishing activity proxy in the two final cost equations. The spatial distribution pattern of fishing effort being closer to the vessel distribution, this fishing proxy was believed to be closer to the usual representation of fishing activity and easier to interpret.

When focusing on the conservation value maps produced with fishing effort data, some spatial pattern differences were noticeable and illustrated how MPAs location may affect either one or the other gear types. The coastal zone between Cap Gris-Nez and the Bay of Somme was almost never selected when passive gears were given the higher weight. By contrast, these areas had a much higher selection frequency when weighting preferentially bottom or pelagic gears. However, only vessels with a length superior to 15 m were considered in this fishing effort data. These areas were certainly also important for smaller boats practicing coastal fisheries and the fact that small fisheries were unaccounted for in the present study needs to be taken into account when interpreting conservation scenarios results produced with such fishing effort data. This limitation often arose when using VMS data in MPA assessment in coastal areas (Bloomfield et al., 2012).

The other part of the cost equation was composed of other human uses currently or potentially operating in the EEC: aggregate extractions and windfarms. We also considered the 3 miles coastal area because its coastal position should in principle facilitate enforcement, and also because it corresponded to a supposedly no-trawling area. The implementation of these different elements in a cost equation was done individually to have a precise interpretation of their influence on the result. The results showed that doubling the cost of aggregate extraction areas had a noticeable influence on the selection frequency compared to the one without any cost. However, this result did not really conflict with the conservation value maps produced when considering the fishing cost: offshore areas, where most extraction sites were located, had a lower selection frequency anyway. The inclusion of windfarm areas and of the 3-miles zone as favourable areas for conservation had even less impact. Conservation value outputs showed the same pattern as when no cost was considered. However, an increase of the selection frequency was found in the concerned areas, increasing their importance for conservation purposes. The economical and spatial importance of aggregate extraction areas and windfarm potential areas pose real management issues in the EEC, and it was therefore thought important to include them in a cost equation, in addition to fishing activities. This resulted in the second proposed cost equation and both cost equation should be used in future conservation planning in the EEC. To propose different MPA networks scenarios depending of the used cost equation is an interesting output to determine MPA locations and their potential impacts on different activities. This approach was for example used in the South Africa MPA network design process (Sink et al., 2010).

At first, the existing MPA network (Chapter One, Figure 3) was meant to be included as an additional constraint in the Marxan process. However, the gap analysis (Chapter Four) showed that the existing MPAs succeeded to reach a 20% target for each habitat and so there was no point to try to add some new areas to complete it. However, in Figure 38, the existing MPA network was overlaid onto the selection frequency obtained using the second cost equation (case 2). This map showed that in most cases the actual MPA network was adequate to reach the 20% target without overly conflicting with the important human activities of the EEC. This map may be a way to predict in which areas users conflicts with MPA implementation might occur. IAs a matter of fact, competition for space was already known to occur in UK waters, between the MCZ in the south of Beachy Head and some aggregate extractions areas (BalancedSeas, 2011).

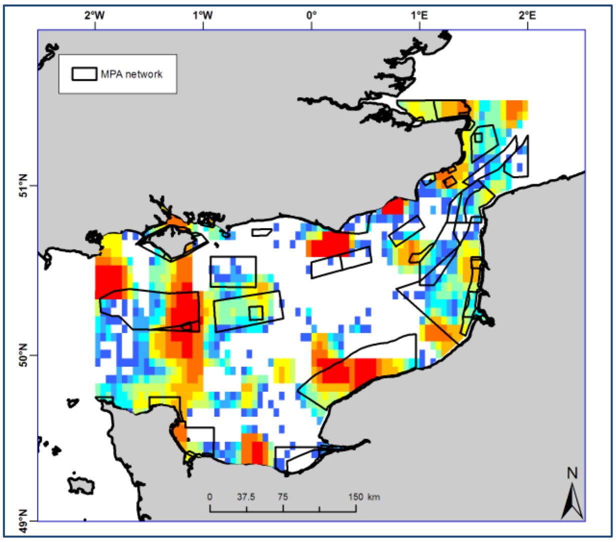

Figure 38: Selection frequency produced with the case 2 cost equation. The MPA network of the EEC is overlaid.

If MPAs are to be designed to address both conservation and management issues, they should account for a larger set of criteria than those used in the present study. In addition to precise descriptors of human activities, future analyses should account for the population dynamics of exploited species, as well as the essential habitats for the completion of their life cycles (e.g., location of spawning and nursery grounds). Moreover, it may be necessary to dynamically link conservation-planning outputs to bioeconomic models (Mahévas and Pelletier, 2004), to be able to evaluate the medium- to long-term effect of the proposed MPA network on both the exploited population and fishery viability.

7. Chapter Seven. General Discussion

7. General discussion

As part of the process of MCZ implementation in South-East England, the Balanced Seas project ran a Regional Stakeholders Group (RSG) representing the numerous stakeholders interests in the area (BalancedSeas, 2011). In their final recommendations delivered in September 2011, Balanced Seas reported stakeholders' comments on the MCZ process and resulting network. The contradictions highlighted in the comments from fishermen to marine ecologist are a good illustration of how complicated it is to implement a MPA network in a congested sea such as the EEC. French fishermen argued that "two Reference Areas (so possibly marine reserves) is not acceptable for trawling and dredging", and they were afraid they would be "the big loser in this process". By contrast, marine ecologists from Seasearch were concerned that "many compromises have already been made to accommodate socioeconomic interests at the expense of ecological integrity". The UK Chamber of Shipping notified that the United Nations Convention on the Law of the Sea (UNCLOS) must be taken into account and that the "innocent passage" right must be maintained, even in reference areas, and that the MPAs should not "restrict the development of port/terminal/anchorage infrastructure", whilst The Wildlife Trust (a conservation organization) was concerned about socioeconomics having a higher influence than ecological conservation.

All these antagonistic reactions show how important it is that recommendations on MPA locations have a solid, scientifically base and that effective implementation depends on accounting for all socio-economic factors (Naidoo et al., 2006). The main results of this thesis could be used to inform managers and decision-makers in the eastern English Channel about EEC ecology and methodological aspects of the implementation of a Marine Protected Area network in the area. Each chapter answered a specific question aiming to improve a possible conservation planning process in the EEC. Because it is essential to have a good understanding of the ecology and biology of the studied environment to run a relevant conservation planning process, marine ecology and conservation sciences were complementary throughout the thesis. These results also represent advances in both fields which led to publications in peer-reviewed journals.

7.1 Contributions to marine policy and management

7.1.1 An Ecosystem-based management perspective

Ecosystem-based management (EBM) is claimed to be a promising approach for managing oceans (Ruckelshaus et al., 2008) and can be adopted as part of a range of approaches, such as area-based management, ecosystem-based fisheries management (EBFM), marine spatial planning (MSP) and ocean zoning among others (Halpern et al., 2010). The importance of the EBM approach is now widely accepted but its implementation is still challenging, mainly due to perceptions that it is too complicated and that data requirements are overwhelmingly demanding (Fraschetti et al., 2011). Moreover, ecoregions such as the English Channel (EC) or the North Sea are shared between several countries thus compounding the difficulties.

The best known example of successful EBM implementation is the Great Barrier Reef Marine Park in Australia (GBRMPA). The GBRMPA is managed at the ecosystem-level, both conservation and sustainable use are promoted, and public and community participation is facilitated. Moreover, the zone is monitored and benefits from regular re-evaluations. However, the GBRMPA depends on one unique authority (GRBMPA authority) established in 1975 (Ruckelshaus et al., 2008), which makes management easier than trans-boundary areas, where international coordination is required. Another good example of EBM implementation is found in Antarctica where the marine resources are managed by the Commission for the Conservation of Antarctica Marine Living Resources (CCAMLR) since 1982 (Ruckelshaus et al., 2008). The CCAMLR has pioneered an EBFM to avoid significant impacts on both targeted and non-targeted species but scientific recommendations emerging from the collaborative process lack a rigorous link with explicit policies.

The EEC is shared by France and the UK, and both countries are responsible for their national waters. At a larger scale, the greater North Sea region (including the EC), one of the most exploited marine areas in the world, depends on several different countries. EBM implementation in the region is complicated by this trans-boundary context and a lack of common framework. Integrated studies such as the CHARM project, within which the present study was carried out, could provide an appropriate scientific support to inform Ecosystem-Based Management in this ecoregion. Similarly, The Netherlands, Denmark and Germany developed a common basis to protect and manage the “trilateral Wadden Cooperation Sea Area” where zoning and MSP played an important role (Ruckelshaus et al., 2008). The trans-boundary issue was partially resolved In the “trilateral Wadden Cooperation Sea Area” example, as national legislation regulated the spatial differentiation of functions and activities for each country but a common zoning system facilitated cooperation (Ruckelshaus et al., 2008).

Within the EBM context, MPAs can be used as an area-based management tool, for conservation, fisheries management or cross-sectoral regulations (Claudet et al., 2011; Halpern et al., 2010; Stelzenmüller and Pinnegar, 2011). However, MPA network implementation must not be considered as an EBM on its own (Halpern et al., 2010) and its benefits will depend on the spatial extent and type of stressors that need to be addressed by management (Fraschetti et al., 2011; Halpern et al., 2010). The dominant stressors have to be known. Depending on their nature, different EBM tools will be more or less efficient. Local-scale and spatially-explicit stressors, such as fishing, aggregate extraction, energy extraction and shorelines modifications can be managed effectively with MPAs. However, other large scale non-spatial stressors such as climate change, acidification or land-based stressors such as nutrients inputs need other regulations and management tools (Halpern et al., 2010).

The greater North Sea is one of the world's ocean areas where the global human impact is the highest (Figure 10) (Halpern et al., 2008). It suffers from various human-induced effects such as acidification (Blackford and Gilbert, 2007), eutrophication (McQuatters-Gollop et al., 2007) and rising sea temperature (Beaugrand, 2009; Philippart et al., 2011), which are all non-spatial stressors. However, the greater North Sea is also a zone of intense spatial competition for many sectorial activities such as shipping channels, aggregate extraction, fishing, aquaculture sites, harbours and other industries (Buléon and Shurmer-Smith, 2008; Carpentier et al., 2009; Desprez, 2000; Douvere, 2008). In some countries bordering the North Sea, such as The Netherlands, the overall demand for space is three times the amount available (Douvere, 2008). In this context of dominant influence of spatial stressors, MPAs could be an efficient management tool and be implemented as part of a Marine Spatial Plan taking into account the environment and conservation, but also human wellbeing and in particular human economic activities (Douvere, 2008; Halpern et al., 2010; Ruckelshaus et al., 2008). The different results found in the previous chapters of this dissertation fit in this scheme and can inform managers and practitioners about the implementation of the MPA network in the EEC, in a trans-boundary context.

7.1.2 Advances in marine ecology and conservation sciences. How can they inform an EBM implementation?

An EBM should be based on the best scientific available data. The different CHARM project outputs and products contributed to develop knowledge of the English Channel. This thesis forms part of this contribution and all the trans-boundary analyses fall within an ecosystem-based approach for the coherent management of this region.

The pelagic typology (**Chapter Two**) was produced because a description of pelagic habitats was not available and could not be used as surrogates in conservation planning assessments (Game et al., 2009; Grantham et al., 2011). In addition to its use in conservation planning, the pelagic typology supports the MSFD directives (EC, 2008) and this regional work complements the national pelagic typology produced by Gaillard-Rocher (2012) in parallel to this PhD. Benthic typologies are continually refined with new data or new methods (Coggan and Diesing, 2011; Howell, 2010; Lozach and Dauvin, 2012; MESH, 2008). In the EUNIS system, the finer levels are described with their associated biological communities. The pelagic typology of the EEC could follow the same path and be improved with additional data or refined with other methods considering both environmental factors and species composition (Ferrier et al., 2007; Koubbi et al., 2011). Moreover, other habitats descriptions exist which are focused more on species than abiotic parameters (Vaz et al., 2007). Combining all these data is of primary importance to inform an EBM.

The pelagic typology was validated using the available survey data from CGFS and IBTS, the two bottom trawl surveys from the EEC. Data from the same surveys were also used to develop sensitivity analyses focusing on the choice of biological data in the habitat target setting process (**Chapter Three**). It was shown that the ecological representativeness of the biological data was an influential parameter with respect to the considered habitat, as well as the use of interpolative post-treatment of the data to insure a regular representation in space. This comparative study showed that even if habitat targets calculated with Species-Area Relationships were more defensible than any policy-driven target, the underlying biology of the observed organisms may be of prime importance in the choice of data. In the present case, where two types of data were available, macrobenthos, sampled with grabs, were found to be more relevant in soft sediment habitats whilst megabenthos, sampled with bottom trawls, should be used in coarser pebbly or gravely habitats. Moreover, megabenthos represented opportunistic data, which were relatively inexpensive to obtain and required less taxonomic expertise than macrobenthos, reducing the risk of the Linnean shortfall which increases as the organisms decrease in size (Whittaker et al., 2005). The Linnean shortfall refers to the lack of knowledge of how many and what types of species there are (Wilson et al., 2009).

The work on typologies and habitat targets highlighted that both data accuracy and ecological relevance is of prime importance to inform conservation planning appropriately. This is true for any management or conservation action: it has to be based on solid biological data. The large benthic datasets gathered by the CHARM project allowed comparisons of different approaches for setting habitat targets. This particular step is conservation planning needs more research to ensure the robustness of the resultant targets and ensure practitioners have confidence in this approach (Metcalfe et al., 2012). The approach taken in **Chapter Three** may not have been conventional but the results obtained showed that, even without many samples or many sampled species, the calculated habitat targets were within the same range as those based on large, species-rich datasets. This comparison study highlighted that opportunistic data can be used to produce habitat targets that are similar to those produced with more exhaustive, but more expensive, datasets. The use of opportunistic data may suffer the same issues as all SAR-based targets, such as the influence of sample size or solution stability (Desmet and Cowling, 2004; Metcalfe et al., 2012; Rondinini, 2011a), but they also have important advantages, such as acquisition and processing price. In any cases, these results emphasise the importance of scientific sea surveys to produce data for an effective EBM implementation and, in the case of already implemented spatial management actions, to monitor their effect.

As was highlighted above in the discussion of MCZ stakeholder comments, integrating human-use data in conservation planning processes is essential for successful implementation. However, planners are still seeking the best way to incorporate relevant data (Ban and Klein, 2009; Naidoo et al., 2006). An initial integration of human activity and exploitation data was done in **Chapter Five**, which showed that the addition of real-world data was more influential than the chosen software or decision algorithm in the MPA network design. This result led to further thinking about how to account for fishing activities in MPA network design and this matter was explored in **Chapter Six**. Two ways of calculating Value Per Unit of Effort (VPUE) and different fishing gears weighting combinations were investigated. It appeared than even more than the weight given to a specific fishing gear, the VPUE calculation mode was really influential on the fishing cost spatial pattern and therefore on resulting Marxan outputs when integrated as cost layers. In the EEC, the spatial variability of costs is really important to consider because of the numerous human uses and important competition for space (Buléon and Shurmer-Smith, 2008).

The main conclusion about the use of socio-economic data in MPA network design is that its spatial variability makes it a really important parameter to consider, most of all in systems like the EEC where conservation targets are reached in many MPA networks scenarios. This is completed by the gap analyses (Chapter Five) results, showing that the different MPA natures and origins (French, English and European initiatives) are complementary to reach conservation targets. In a trans-boundary context and a multiple-use sea such as the EEC, conservation cannot be thought of independently from policy and socio-economic considerations. Marine spatial planning may represent a good way to deal with all these components and produce an integrated solution at the regional scale (Stelzenmüller and Pinnegar, 2011).

7.2 Limitations and further research

The different chapters of this dissertation aimed to inform a systematic conservation planning approach in the EEC and did so with the best available data. However, in any conservation planning exercise, data selection and used methods can always be questioned and the results always have to be considered with caution because of possible uncertainty linked to data or associated treatment methods (Regan, 2009; Rondinini et al., 2006). Conservation planning is meant to be a dynamic process, as well as EBM or MSP. Thus, all results produced here could be improved with any new data and method. Regan et al. (2009) distinguished between two broad types of uncertainties: parametric and structural uncertainties. Parametric uncertainty relates to uncertainty in parametric values used to produce the conservation planning outputs. Structural uncertainty relates to uncertainty in the problem specification that might affect the objectives of the conservation plan or uncertainty in future events that could impact the conservation plan's ability to reach its objectives (climate change or modification in socio-economic related to the study for example).

Section 7.3.1 deals with the use of environmental data in conservation applications. In Section 7.3.2 I will identify some limitations to the studies linked to parametric uncertainty. Structural uncertainty is less of a concern because this PhD project did not intend to produce a finalised conservation plan to be implemented in the EEC. Addressing structural uncertainty would require stepping into the management realm. However, this could be partly circumvented by further research, as noted in Section 7.3.3.

7.2.1 Uncertainty in environmental data and models

Uncertainties can be categorized into four groups: incertitude, variability, model uncertainty and linguistic uncertainty (Regan, 2009). Incertitude includes measurement errors, information gaps and the uncertainty associated with subjective judgements; variability can be due to changes in the values of parameters across time or space. Model uncertainty refers to uncertainty arising from the choice of modelling framework and linguistic uncertainty can be due to inadequate language or terminology which could lead to misinterpretations.

First, incertitude is present in the thesis as it relates to information gaps such as inaccuracies in habitat maps, errors in presence/absence data due to species catchability or partial distribution extent of the considered species. These problems are inherent to any ecological study based on field sampling and their associated sampling biases. For example, the pelagic typologies were produced on a 10 km grid corresponding to the coarsest resolution of the environmental data. Then, the results were smoothed on a 1 km grid using indicator kriging and combined into a single typology map with apparently clear cut boundaries between each class. This apparent precision is deceptive and that is why when the biological validation was done, species samples situated in a buffer zone of 5 km on each side of habitat limits were removed.

All the interpolated data used in the thesis like megabenthos probability of presence or fish abundances (**Chapter Three**), were checked to avoid extrapolation. As a result values estimated further away than the maximum distance between two observations were removed. Geostatistical interpolation also produced an estimation of the kriging error for each species. These maps were not presented in this dissertation because the data were further transformed into presence/absence layers.

7.2.2 Uncertainty in conservation planning

Incertitude is also present when evaluating habitat biodiversity (total biodiversity estimator, Chapter Three) and when applying habitat targets across different spatial scales.

Sensitivity analyses can be carried out to overcome parametric uncertainties (Regan, 2009). **Chapters Three, Five and Six** relied on sensitivity and comparative analyses with the aim of discussing variability related to the chosen data or uncertainty related to the method. Uncertainties linked to data sampling biases have been considered in the habitat target setting process (Chapter Three) with the interpolation of data to create probability of presence on a regular grid. Moreover, although the transformation of these probabilities to strict presence/absence was done with the best known method to avoid false absence (Jiménez-Valverde and Lobo, 2007; Rondinini et al., 2006), this additional treatment may generate another layer of uncertainty. Among others comparisons, Chapter Three compared habitat targets using point data or predicted distribution data (Rondinini et al., 2006), which is typically the kind of sensitivity analyses expected in a conservation planning exercise. The results were used to discuss uncertainties related to these data-driven approaches to set habitat targets (Cf. Chapter Three). However, other studies showed that concerning the different types of input data in a conservation planning exercise, predicted data are the ones which could produce the least biased solutions (Rondinini et al., 2006).

Producing various “scenarios”, as was done in the chapters using conservation planning software, (**Chapters Three, Five and Six**) is also a way to circumvent uncertainty and in particular linguistic uncertainty (Regan, 2009). Presentation of different scenarios using different sets of parameters allows clear communication about the study objectives and is a way to avoid misunderstandings of the results by the public. Marxan and other conservation planning software are sometimes criticized for being a “black box” delivering magical solutions. The comparison between various scenarios allows the comprehension of how these tools operate and shows how data variability induces outputs variability.

Presenting different scenarios was really important for all Marxan analyses run in this thesis because the biodiversity surrogates used were always large scale habitats and distribution patterns, so there was a great deal of flexibility leading to low selection frequency. To develop more Marxan analyses in the EEC, it would be interesting to integrate more point data or particular observations such as Habitats of Special Interest (JNCC and Natural England, 2010) which would better constrain the process of MPA selection even if these kind of data could also be biased with a lot of “false negative” or errors of omission (Rondinini et al., 2006). The absence of strong ecological constraints to setting the MPA network was illustrated in Chapters Three and Six, where the addition of economic data was shown to have the greatest influence on the MPA design. The improvement and precision of human exploitation surrogates is therefore important to decrease their associated uncertainty.

If variability was taking into account in the pelagic typologies, it remains a caveat of these conservation planning outputs where all biological data were sampled at one period in the year. To overcome this problem, Marxan could be coupled with dynamic models like Ecopath with Ecosim to predict the impact of MPAs on ecosystems functions. Moreover, some models such as ISIS fish (Mahévas and Pelletier, 2004) could help to predict long term effects of the proposed MPA networks design on the exploited stocks and on fishery viability.

Finally, another source of uncertainty which must be kept in mind when regarding **Chapter Four** is the uncertainty related to the management status of the existing MPAs. Moreover, when Marxan was used to produce an MPA network proposition, it was implicit that all proposed conserved areas would be strict reserves with absolutely no human uses. This thesis proposed various results, which could inform future conservation plans in the EEC. However, the future management status of the existing MPAs in the EEC was largely unknown at the time of writing, and it is probably the largest source of uncertainty. It is very unlikely that all MPAs will become no-take areas and the legislation in the majority of the network might not be restrictive enough to balance all the human uses that exist in the EEC. Balanced Seas identified Reference Areas which were announced as the more constraining areas for users but no such initiatives existed in France. Though, the “Grenelle 2” law planned to set 20% of the national waters in MPAs with half of it being no-take areas. Moreover, fisheries regulations within the national MPAs may be really difficult to set up as fishing stocks are a “common” resource which access is regulated at the European level through the Common Fishery

Policy. National restriction initiatives unsupported at the European level will only apply to local fisheries and may lead to tensions between stakeholders.

7.2.3 Further research

Conservation planning needs to be dynamic and the results presented in this dissertation could be complemented by further research which would improve our understanding of the EEC functioning. Moreover, the MPA implementation process is an on-going process in the area and once MPA locations and their management plans will be developed, further investigations about the coherence of the trans-boundary network will be needed.

Connectivity

Connectivity refers to the exchange of individuals between populations and is increasingly recognized as a key conservation objective because of its importance for population persistence and recovery from disturbance (Almany et al., 2009). If there is little or no connectivity between critical life stages and the habitats or processes supporting them, then the ability to sustain populations within MPAs and the degree to which a MPA can serve as a source of recruitment to unprotected areas or other MPAs will be limited (Roberts et al., 2003b). Considering larval retention and connectivity, MPA sizes and spacing should be optimized to achieve both biodiversity maximisation within MPAs and beyond their boundaries (Halpern and Warner, 2003; Jones et al., 2007). Shanks et al (2003) recommended an optimal spacing of 10-20 km between each MPA for species with typical larval stage durations. However, Palumbi (2004) argues for distances ranging from 10 to 200 km to better reflect larval dispersion abilities from large fish to invertebrates. Concerning the EEC, JNCC and Natural England recommended a maximal distance of 40-80 km between two MPAs (NaturalEngland and JNCC, 2010), but they also specified that this recommendation should be revised with species movements and habitat linkages knowledge. Some empirical estimates of connectivity would be a good addition for a proper MPA network design. If empirical data are hard to obtain, some ecological models could help to understand larval dispersion in the EEC, these biological investigations should be part of new research projects in the area. Moreover, some methodological advances now allow to deal with MPA sizes and spacing in the MPA network design (Smith et al., 2010).

Functional diversity

One of the caveats of the data-driven approaches to set habitat targets is that it considers species richness as representative of biodiversity. This consideration is more and more debated as maintaining functional diversity appears to be essential for healthy ecosystems. Functional diversity can be described as the relative abundances of functional traits in a given ecosystem (Díaz et al., 2007). If functional diversity matters more than taxonomic diversity, then species that play relatively unique roles in communities should be of primary importance (Claudet et al., 2011). Moreover, conservation of functional diversity is essential for the conservation of goods and services provided by the considered ecosystems (Claudet et al., 2011; Mouillot et al., 2008). Thus, functional diversity should be considered when designing MPA networks and it must be ensured that MPAs are able to protect and restore species performing particular functions. Then, when monitoring MPAs, some metrics considering species traits and functions should be investigated in addition to the more classical ones such as species richness or abundance (Claudet et al., 2011).

At the present time, in the EEC, many studies on species traits have been competed or are on-going, from benthic species to fish and focusing on various functional aspects especially trophic ones. All these results should provide valuable information for MPAs design and monitoring in the future.

Management and monitoring

In a trans-boundary sea such as the EEC, management harmonization is not an easy thing to set up. However, France and UK can benefit from international frameworks such as OPSAR to elaborate collaborations and common management (Vong, 2010). Moreover, structures as INTERREG, inter-regional programs financed by the European Regional Development Fund could provide funding to support the scientific and managerial process which could lead to trans-boundary management and so to an EBM at the EEC scale.

However, the existing MPA network in the EEC was not created as one cohesive network. It has resulted from an assemblage of different types of national MPAs and a serious monitoring program is necessary to assess if it acts as a network at all (Grorud-Colvert et al., 2011). For example, in the Mediterranean Sea, Claudet et al (2008) showed that the ecological effectiveness of one well-enforced reserve is not increased with the proximity of other MPAs. According to the IUCN Marine MPA network definition, an *ad hoc* network like that of the EEC cannot be considered as an ecologically effective network if the assemblage of MPAs does not exceed the ecological benefits of multiple unconnected MPAs (Grorud-Colvert et al., 2011). Monitoring programmes in the EEC context may not be easy to set up, though, several criteria should be adopted to provide a correct assessment of the efficiency of the MPAs as a network: (1) The "success" of the MPA network should be clearly defined with regards to specific objectives, (2) well-coupled sampling and analytical designs have to be decided to detect and interpret real change and (3) adequate funding and infrastructure to support the long time period are needed (Field et al., 2007; Grorud-Colvert et al., 2011). Moreover, a network evaluation would require reference sites in unprotected areas for each designed reserves to be able to judge the network efficiency with these criteria: replenishment within MPAs and spillover to open areas and connected MPAs (Grorud-Colvert et al., 2011). Finally, in addition to the possible ecological benefits of the MPA network, the human dimension should be monitored too. In that context, information about how concerned communities have responded to the MPAs implementations is also a way to predict its success.

8. Conclusion

This thesis aimed to inform a systematic conservation planning approach in the EEC exploring different aspects of the process. Through a trans-boundary gap analysis that considered the French and English MPAs as one global network, sensitivity analyses demonstrating the importance of biological data quality, human activity data availability, and comprehension of the conservation objectives, this thesis contributed to develop the science in support of marine conservation planning.

However, a MPA network must be assessed on more parameters than the ones explored here. Further studies could consider the connectivity and the viability of the MPA networks, through modeling and empirical processes. In the CHARM project, two PhD research projects aimed at informing conservation planning in the EEC. The second one explored multi zoning approach, links between conservation planning and trophic networks and these results are complementary to the ones produced here. Collaborative projects like CHARM represent scientific advances for an EBM approach. However, such advances should be accompanied by steps towards developing effective management approaches and operational tools. Though, in a trans-boundary area like the EEC, common policy or management rules are really difficult to put into practice because national sovereignty is at stake. Moreover, the addition of policy layers from international to local scale complicates also the task. In particular, the coastal locations of many of the existing MPAs imply that link between terrestrial and marine management plans are needed.

Conservation planning and at a larger scale Ecosystem Based Management has to be based on the best available evidence and this implies the continued need to develop knowledge in marine biology, ecology, economics and all the scientific fields which could inform these processes. Different projects in the EEC intend to produce new data on ecological coherence, connectivity, management rules and monitoring (PANACHE project), human uses interactions (VECTORS project), fisheries socio-cultural aspects (GIFS project) or integrated maritime strategy (CAMIS project). These new studies will allow us to better inform managers and practitioners. Conservation planning must continually be reassessed with regards to new information and not be a static process.

8. Conclusion

Cette thèse a étudié plusieurs aspects de la démarche de planification de la conservation en Manche Est : une analyse des lacunes transfrontalières (Gap analysis) a été réalisée sur le réseau d'AMPs français et anglais et des analyses de sensibilité ont démontré l'importance de la qualité des données biologiques, la disponibilité des données d'exploitation et l'importance de la justesse des objectifs de conservation. Ces différents résultats peuvent contribuer à la mise en place d'une démarche de planification de la conservation dans la zone.

Toutefois, la mise en place d'un réseau d'AMP doit être évaluée sur plus de critères que ceux qui ont été traité ici. Les notions de connectivité et de viabilité du réseau d'AMP devraient par exemple être explorées à travers des études empiriques ou de la modélisation. Dans le projet CHARM, deux thèses avaient pour but de participer aux études exploratoires de planification de la conservation en Manche Est. La seconde thèse a exploré les aspects de zonage multiple et a développé les liens entre planification de la conservation et impacts des réseaux proposée sur les réseaux trophiques proposant des résultats complémentaires de ceux produits ici. Les projets collaboratifs comme CHARM représentent des avancées scientifiques qui sont nécessaires pour la mise d'une gestion écosystémique et raisonnée. Ces avancées scientifiques devraient être accompagnées d'avancées vers une gestion intégrée et la mise en place d'outils opérationnels. Cependant, dans un contexte transfrontalier comme en Manche Est, la souveraineté nationale est en jeu et cela complique la mise en place de mesures de gestion et de règles communes. De plus, la tâche est encore compliquée par l'accumulation de textes réglementaires, du niveau international au niveau local. Cela est particulièrement vrai en Manche où le caractère côtier de la plupart des AMPs implique également de créer un lien fort entre gestion terrestre et gestion marine pour que cela soit efficace.

La planification de la conservation et plus largement la gestion écosystémique et raisonnée doivent être supportées par les meilleures données scientifiques disponibles, ce qui implique de continuer à développer les connaissances en biologie et écologie marine, en sciences économiques et dans tous les domaines à même d'informer le processus. En Manche, différents projets de recherche ont pour but de de produire de nouvelles données dans ces domaines : la cohérence écologique des réseaux d'AMP, la connectivité, les mesures de gestion et d'évaluation (projet PANACHE); les interactions entre les différents usages humains (projet VECTORS); les aspects socio-culturels de la pêche (projet GIFS) ou la mise en place de stratégies maritimes intégrées (projet CAMIS). Toutes ces nouvelles études vont permettre d'améliorer les connaissances afin de mieux conseiller les gestionnaires et les décideurs. Le processus de planification de la conservation se doit d'être dynamique et de toujours intégrer les nouvelles données disponibles pour améliorer les résultats.

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