Rijkswaterstaat
Ministry of Infrastructure and Water Management

RWS INFORMATION

Marine Protected Areas in Europe

An assessment of the current status and the representation of benefits in socio-economic analyses in order to support decision-making

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Prologue

This internship took place within Rijkswaterstaat and has been made possible by several people that I would like to thank. First of all, I would like to thank my daily supervisor at Rijkswaterstaat, Rob van der Veeren, for his enthusiasm and his supervision during the last couple of months. I could ask him anything and he was always willing to help me. Rob learned me a lot about policy and economy and it has been really nice to work with him. I would also like to thank Marion van Rijssel for her overall help during this internship. She provided me with very useful comments and taught me to always keep in mind the bigger picture and the purpose of the report. Furthermore, I would like to thank Tjisse van der Heide for providing me with relevant scientific literature and explanations. Finally, I would like to thank my colleagues at Rijkswaterstaat for their help and comments. In particular Xander Keijser for all his useful suggestions. Overall, this internship has been very interesting and a lot of fun. It's an experience that I wouldn't have want to miss. It has brought me a lot and I would recommend it to everyone. I hope you enjoy reading this report.

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Executive summary

Marine ecosystems are threatened and have been declining due to various human activities. Marine Protected Areas (MPAs) are a widely recognized tool to protect and restore marine ecosystems. MPAs are clearly defined marine areas in which human activities are restricted. MPAs are recognized as a potential measure in marine policies, such as the Convention on Biological Diversity and the Habitats and the Birds Directives. MPAs are also integrated in the Marine Strategy Framework Directive (MSFD) as one of the measures that may contribute to the achievement of the "Good Environmental Status" in the European marine waters. The MSFD requires that Cost-Benefit Analyses (CBAs) are carried out prior to the introduction of new measures. Determining the costs of MPAs is hard, but possible. However, determining the benefits is very challenging because it is often not clear what the exact benefits could be, and how they can be quantified or monetised in order to include them in CBAs. This has been very challenging when CBAs were carried out prior to the designation of the Frisian Front and the Central Oystergrounds, two MPAs within the Netherlands. The difficulties faced at that time were the initial reason behind this report.

The aim of this report is to get an understanding of the benefits of MPAs and how they can be quantified or monetised in socio-economic analyses. This report therefore focused on MPAs throughout Europe and to some extent outside Europe to see what lessons can be learnt. The first part of this report assesses the current status of MPAs in order to see whether or not measures are sufficient to protect the marine environment and to understand the factors that influence MPA performance. The second part of this report looked into several benefits of MPAs and discusses the methods that have been used to quantify or monetize benefits prior to MPA implementation. This resulted in a recommended approach which was tested in a case study: The Borkum Reef Ground, a possible future MPA within the Netherlands.

From the analysis of the current status of MPAs in Europe, it appears that is very likely that MPAs are currently underperforming. The fact that overall conservation targets are often not achieved may be caused by the fact that MPAs are often multiple use MPAs in which many human activities are allowed to continue. MPAs are also considered small and not connected based on ad hoc analyses. Finally, management of MPAs is often insufficient and monitoring often doesn't take place. Many benefits of MPAs may therefore go unnoticed.

The second part of this report looked into several benefits of MPAs and discussed the methods that have been used to quantify or monetize these benefits. From this analysis it is clear that MPAs are capable of producing a variety of benefits. However, large uncertainties remain on the extent of benefits that can be expected. Ecological benefits are difficult to predict due to high marine ecosystem complexity. These benefits are also highly dependent on MPA characteristics which is why these benefits can vary substantially between MPAs. Other benefits such as carbon storage and sequestration are associated with marine coastal systems. However it is questionable whether or not this is a permanent benefit, since disturbance may cause the release of the carbon, turning the carbon sink into a carbon source. Fishers may benefit from MPAs through so-called spill-over effects, but this depends on the recovery within MPAs before effects can be expected outside MPAs. It therefore takes time before these potential spill-over effects may accrue; often more than 10 years. Furthermore, fishers often need to fish further away or experience increased competition outside the MPA which increases their costs. Spill-over effects can therefore not be directly translated into economic benefits. MPAs can also contribute to recreation and tourism, as acknowledged in stakeholder consultations. However, studies that assess these

benefits prior to and after MPA implementation are missing. This makes it difficult to distinguish between the effect of the site and the effect of MPA designation.

Multiple methods have been used to quantify or monetize these MPA effects prior to designation. These methods include: bio-economic modelling, benefit transfers, stated preference methods, Multi Criteria Analysis (MCAs) and the eco point valuation method. Bio-economic modelling focuses mostly on the relations between fisheries and fish stocks. Benefit transfers use data obtained from one area to predict the value of another area, these can be used for many benefits as long as primary valuation data is available and reliable. Stated preference methods use surveys in order to obtain the value that people are willing to pay for a certain good. MCAs can be used to assess different criteria that cannot be expressed in a single unit. The eco point valuation method can be used to quantify the effects on biodiversity.

The analyses in this study show that the exact links between biodiversity and other benefits are poorly understood. However, it is widely recognized that biodiversity contributes to the generation of many benefits. The eco point valuation method can be used to quantify the intrinsic value of nature. Therefore, the eco point valuation method is recommended to quantify the potential benefits of MPAs and include this information in CBAs.

The applicability of the eco point valuation method was tested in a case study on a possible future MPA in the Dutch part of the North Sea; the Borkum Reef ground. This case study pointed out several significant knowledge gaps that need to be addressed before this method can be applied. First of all, it is not clear what a pristine marine area looks like. Therefore, it is almost impossible to make a judgement about the current quality of the marine ecosystem. Secondly, there are large uncertainties about the extent of MPA effects on the marine ecosystem. The eco point valuation method requires this kind of information in order to quantify the gain in eco points. Application of this method based on current knowledge would require many assumptions and make the results highly unreliable.

Based on all these findings, the following steps are recommended:

- For now, it is recommended to use a qualitative approach in order to illustrate the benefits of MPAs, as there are large uncertainties about the extent of the effects on the ecosystem and the potential benefits that can be expected.
- It is highly recommended that a fraction of existing MPAs become no-take zones. This would serve 2 purposes: First, no-take MPAs perform much better than multiple use MPAs, thus contributing more significantly to the achievement of conservation goals. And the second purpose is that these areas may provide an understanding of what a more pristine marine area might like. This information is needed in order to assess the current ecological quality relative to an undisturbed state.
- Connectivity is very important to keep in mind when effects of MPAs need to predicted. This information is needed in order to assess the importance of a particular area for particular species and the effects that can be expected. The developments in genetic research offers the opportunity to increase knowledge in this field.
- Furthermore, it is of utmost importance to establish a robust monitoring framework for MPAs in order to assess their effects. This is currently missing and needed in order to understand the effects that can be expected.

This type of data is needed in order to quantify MPA effects in the future and be able to predict what benefits they may provide.

The role of Rijkswaterstaat

Rijkswaterstaat is the executive agency of the Ministry of Infrastructure and Water Management. Rijkswaterstaat is responsible for the management and development of the main roads and waterways (Rijkswaterstaat n.d.a). Rijkswaterstaat is also responsible for managing the largest marine area within the Netherlands, the North Sea (Rijkswaterstaat n.d.b). The North Sea is among the busiest marine areas in the world in which many activities take place that put huge pressure on the marine environment.

The department of Water, Transport and the Environment

Rijkswaterstaat is divided into 6 nationally operating departments and 7 regional departments (Rijkswaterstaat n.d.c). The department of Water, Transport and Environment is a nationally operating department. This department creates strategies for the main roads and waterways and the environment. This departments other task is to advise and share information. This information is shared within Rijkswaterstaat but also with external parties by promoting cooperation and creating alliances and can be seen as the link between policy and implementation (Rijkswaterstat n.d.d.).
Advisory board on economic analyses

Within the department of Water, Transport and the Environment there is an advisory board on economic analyses called "Steunpunt Economische Expertise", specialized in performing economic analyses concerning infrastructural projects. These economic analyses can be used to assess whether or not a project contributes to overall welfare in the Netherlands. The main focus of this advisory board is informing about the application of CBAs prior to the implementation of infrastructural projects (Rijkswaterstaat n.d.e.). This advisory board has therefore also been involved in CBAs concerning marine spatial planning such as the designation of MPAs.

The reason behind this report

The MSFD requires that CBAs are performed prior to the designation of new measures such as MPAs (EC 2008). However, determining the benefits of MPAs is very challenging. This was particularly difficult when CBAs needed to be performed prior to the designation of the Frisian Front and the Central Oystergrounds, two MPAs within the Netherlands (van Oostenbrugge et al. 2015). This has been the initial reason behind this report. This report focuses on MPAs throughout Europe because possibly other Member States faced similar issues. The overarching goal is to assess what the benefits of MPAs could be and how they can be included in CBAs and decision-making.

Abbreviations

CBA	Cost-Benefit Analysis
CBD	Convention on Biological Diversity
CFP	Common Fisheries Policy
GES	Good Environmental Status
MCA	Multi Criteria Analysis
MPA	Marine Protected Area
MSFD	Marine Strategy Framework Directive
PoMs	Programme of Measures
RSC	Regional Sea Convention

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1 Background

1.1 Marine ecosystems are in need of protection

Marine ecosystems are extremely varied in both type and geographical extent. Marine ecosystems include oceans, reefs, mangroves, the deep sea and the sea floor. They cover about 70% of the earth's surface and play an essential role in human well-being as they provide environmental, social and economic benefits to the earth's growing population (OECD 2015). Due to human activities these ecosystems are under increasing pressure. Many of the world's marine ecosystems are degrading or are being used unsustainably. For example, 33,1% of the worlds fish stocks are over-exploited and some of them are on the verge of collapse (FAO 2018). Marine ecosystems are also under increasing pressure, for example coral reefs are suffering from ocean acidification and increasing temperatures. Not only exploitation itself imposes threats, harmful fishing practices such as bottom trawling and dynamite fishing contribute to the destruction of marine habitats (OECD 2015). Furthermore, pollution from land, such as litter and nutrients, are threatening marine habitats and species, and thereby threatening the marine ecosystems. The resulting loss in biodiversity and degradation of the ecosystem reduces ecosystem resilience. This affects both the health of the seas as well as human well-being by endangering the ecosystem services essential for meeting our needs. Moreover, these pressures are likely to increase in the future due to our consumption and production patterns and the growing population (EEA 2015a).

1.2 Protection offered by Marine Protected Areas

In Europe, marine ecosystems are threatened as well and patterns are similar to those observed in the rest of the world. A long history of human uses in the European marine waters have resulted in depleted fish stocks and habitat degradation (Fenberg et al. 2012). Nowadays, there is a growing awareness of these threats on ecosystem functioning and the consequences that it imposes. In Europe, this led to implementation of several policies which aim to protect marine ecosystems. Fisheries are for example regulated through the European Common Fisheries Policy (CFP) and the European Water Framework Directive aims to regulate the input of nutrients and chemicals into the water (EC 2000)(EC 2013). In addition to these measures, policy makers have recognized the designation of Marine Protected Areas (MPAs) as a crucial policy tool to protect the marine environment. MPAs are defined as:

"A clearly defined geographical space, recognised, dedicated and managed, through legal or other effective means, to achieve the long-term conservation of nature with associated ecosystem services and cultural values" (IUCN-WCPA, 2008).

MPAs cannot protect marine ecosystems against all anthropogenic pressures but they can help build resilience of the marine environment. MPAs protect marine ecosystems in multiple ways: (1) they contribute to the conservation of biodiversity and associated ecosystems, (2) protect spawning and nursing grounds, (3) provide sites for recovery from stresses, and (4) can protect settlement and growth areas (IUCN-WCPA, 2008). MPAs thus act as a vault for biodiversity from which ecosystems structures and functions can be recovered. In that sense, MPAs are an essential tool for the protection and recovery of marine ecosystems that are crucial for current and future human well-being (EEA 2015b).

1.3 Marine Protected Areas are integrated in EU policy

The EU protects marine areas through the designation of MPAs via the Habitats and the Birds Directives, together with protected areas on land these areas make up the Natura 2000 network (EC 2014). These are crucial tools that aim to protect specific habitats or species. In order to protect the whole range of biodiversity the Marine Strategy Strategy Framework Directive (MSFD) requires to designate MPAs in addition to existing ones designated under the Habitats and the Birds Directives. These should contribute to the establishment of a so-called ecologically coherent network of MPAs. Criteria for ecological coherence include representation, adequacy, replication, viability and connectivity (Foster et al. 2017). In order to contribute to the establishment of an ecologically coherent network of MPAs, the Marine Strategy Framework Directive (MSFD) requires to designate new MPAs in addition to existing ones designated under other policies such as the Habitats and the Birds Directives. The overarching goal of the MSFD is to achieve of "Good Environmental Status" (GES) in the European marine waters by 2020. Member States are required to comply with this directive by developing a Programme of Measures (PoM) for their marine waters. This is the operational tool in which each Member State identifies threats and puts concrete measures into place in order to protect their marine waters. These PoMs have to be put in place in order to achieve GES by 2020 (EC 2008).

1.4 The Regional Sea Conventions

The European marine waters are divided into 4 regions: the North-east Atlantic Ocean, the Mediterranean Sea, the Baltic Sea and the Black Sea. For each of those regional seas there is a Regional Sea Conventions (RSC) that is responsible for the international coordination of the management and supervision of the marine environment. The RSC of the Baltic Sea is the HELCOM, the RSC of the North-east Atlantic Ocean is the OSPAR, the RSC of the Mediterranean Sea is Barcelona Convention and the RSC of the Black Sea is the Bucharest Convention. The MSFD requires that Member States coordinate their marine strategies using the existing RSCs (EC 2008).

1.5 Socio-economic analyses

Article 13.3 of the MSFD requires that measures, such as the designation of a MPA, must be cost-effective and technically feasible. Therefore, Member States must perform impact-assessment as well as Cost-Benefit Analysis (CBA) prior to the introduction of measures (EC 2008). The objective of those analyses is to describe the positive and negative impacts of potential measures, compared to a business as usual scenario, describing what is likely to happen without the measure under consideration. This is done in order to get a clear view of the potential effects and benefits measures could have, and how much they may cost. It also provides information on the distribution of those costs and benefits over various actors. In CBA, these costs and benefits are preferably presented in monetary terms in order to have one common denominator. However, in cases where it is hard to find monetary values, often also quantitative and/or qualitative information is presented. Determining the costs of designation of a MPA is usually difficult, but not problematic. However, determining the benefits of an MPA is often very challenging, which can lead to evidence gaps (Bertram et al. 2014).

1.6 Estimating the benefits of Marine Protected Areas

Estimating the benefits of MPAs is usually very difficult (Bertram et al. 2014). What are the benefits of the designation of MPAs, how can those benefits be quantified and maybe also monetised? These were some of the questions the Netherlands faced when they performed the CBA for the designation of the Frisian Front and the Central Oyster grounds as one of the measures they wanted to include in their program of measures for the MSFD (van Oostenbrugge et al. 2015). Part of the difficulty is that benefits arising from MPAs, such as the protection of marine biodiversity, provision of nurseries for fish species, coastal protection and carbon sequestration are hard to determine and mostly not represented in the prices of goods and services on the market. This makes it usually very difficult to include these benefits in CBAs (OECD 2015).

1.7 The aim and structure of this report

This report has been commissioned by Rijkswaterstaat, the executive agency of the Ministry of Infrastructure and Water Management. The aim of this report is twofold: The first aim is to provide an overview of the current status of MPAs in Europe in order to assess whether measures are sufficient to protect marine ecosystems and their functioning. The second aim of this report is to assess how benefits can be better represented in socio-economic analyses in order to support decision making and promote the implementation of MPAs.

This report has the following structure, the second chapter provides an overview of how MPAs are part of or linked to marine policy. The role and place of MPAs in global policy (including the Convention on Biological Diversity) and EU policy (including the Habitats and the Birds Directives, the MSFD, and the CFP. The third chapter provides an insight in the current status of the European MPA network. This includes an overview of the number of MPAs under the jurisdiction of the RSCs, an assessment of whether the MPAs in the European marine regions can be considered ecologically coherent, and a description on how this may affect MPA success. The fourth chapter starts with a description of some of the possible benefits of MPAs. After that, the representation of MPA benefits in socio-economic analyses will be described as this could support decision-making processes in the future. Here, valuation methods are described and explained by using examples from the MPA literature. In the fifth chapter, these recommendations are used to estimate the benefits of a possible future MPA in the Netherlands, the Borkum Reef Ground. The final chapter presents the overall conclusions and recommendations.

2 Marine Protected Areas in marine policy

This chapter describes how MPAs are part of marine policy globally and regionally. The Convention on Biology Diversity, as well as the Habitats and the Birds Directives, the Marine Strategy Framework Directive and the Common Fisheries Policy will be briefly explained. This chapter also discusses how these policies are connected. Moreover, the role of the Reginal Sea Conventions in the establishment of MPAs will be discussed.

2.1 Convention on Biological Diversity

The United Nations prepared a legal instrument for the conservation and sustainable use of biological diversity: the Convention on Biological Diversity (CBD). In 1992 the CBD was opened for signature. The CBD was signed by 168 countries and international parties, among them the European Union (CBD 2020). Marine measures were specified at the seventh convention of the CBD. It was stated that effectively managed, representative MPA networks should be implemented by 2012, and a global target was set at MPA coverage of 10%. However, in 2012 the target was not met as global MPA coverage at that moment was 1,08%. In 2010 in Japan 193 contracting parties again committed themselves to conserve at least 10% of the coastal and marine areas by establishing effectively managed, representative and connected systems of MPAs and other area based conservation measures. This is known as “Aichi target 11” (EEA 2015b).

2.2 The Birds and Habitats Directives

In 1979, the Birds Directive was adopted which aims to protect all wild birds and their habitats across the EU. The Habitats Directive was adopted 13 years later in 1992 and extended the coverage to approximately 1000 rare, threatened or endemic plants and wild animals. Moreover, it protects 230 rare habitats and forms the cornerstone of the EU’s legislation on protection and conservation of nature. The goal of the two directives is to achieve “Favourable Conservation Status” of the species and habitat types it protects (EC 2014). A fundamental part of the objectives is the use of special conservation areas, both on land and at sea, which should create a coherent ecological network. Together, these areas make up the Natura 2000 network. Thus, the Natura 2000 network includes “Special Areas of Conservation” designated under the Habitats Directive, and “Special Protection Areas” designated under the Birds Directive (EEA 2015b). In 2000, it was recognized that the loss of biodiversity was continuing, and was threatening sustainable development. In response, the EU Biodiversity Action plan was launched in 2006 which was followed by the European Biodiversity Strategy in 2011. Target 1 of this strategy describes the complete implementation of the Habitats and Birds Directives, including the completion of the Natura 2000 network in the marine environment. It also aims at good management by 2012 (EC 2006). The latest update on the European Biodiversity Strategy was launched this year in which more ambitious targets were set at protecting at least 30% of the marine area within MPAs of which one-third (10%) should offer strict protection before 2030. Member States are responsible for the additional designation of these MPAs, in which the need to complete the Natura 2000 network of protected areas is again emphasized (EC 2020a). In that sense, the Natura 2000 network plays an essential role in the establishment of MPAs in the European marine environment. However, the Natura 2000 network protects only specific vulnerable marine species and habitats. As knowledge increased, it became apparent that the directives’ approach to the protection of marine fish, invertebrates and habitats was not coherent, and that better protection of the marine environment was needed (EEA 2015b).

2.3 The Marine Strategy Framework Directive

The Marine Strategy Framework Directive (MSFD) was adopted in 2008 and aims to achieve "Good Environmental Status" (GES) of the European marine waters by 2020 (EC 2008). The MSFD describes GES based on 11 descriptors (box 1). Detailed Criteria and methodological standards were also produced in order to help Member States with the implementation of the MSFD. These descriptors were updated in 2017 (EC 2017).

Descriptors for Good Environmental Status (GES)

(1) Biological diversity is maintained. The quality and occurrence of habitats and the distribution and abundance of species are in line with prevailing physiographic, geographic and climatic conditions.

(2) Non-indigenous species introduced by human activities are at levels that do not adversely alter the ecosystems.

(3) Populations of all commercially exploited fish and shellfish are within safe biological limits, exhibiting a population age and size distribution that is indicative of a healthy stock.

(4) All elements of the marine food webs, to the extent that they are known, occur at normal abundance and diversity and levels capable of ensuring the long-term abundance of the species and the retention of their full reproductive capacity.

(5) Human-induced eutrophication is minimised, especially adverse effects thereof, such as losses in biodiversity, ecosystem degradation, harmful algae blooms and oxygen deficiency in bottom waters.

(6) Sea-floor integrity is at a level that ensures that the structure and functions of the ecosystems are safeguarded and benthic ecosystems, in particular, are not adversely affected.

(7) Permanent alteration of hydrographical conditions does not adversely affect marine ecosystems.

(8) Concentrations of contaminants are at levels not giving rise to pollution effects.

(9) Contaminants in fish and other seafood for human consumption do not exceed levels established by Community legislation or other relevant standards.

(10) Properties and quantities of marine litter do not cause harm to the coastal and marine environment.

(11) Introduction of energy, including underwater noise, is at levels that do not adversely affect the marine environment.

Box 1. The eleven descriptors of the MSFD used to describe Good Environmental Status (EC 2008).

The MSFD uses an ecosystem approach in order to achieve its goals, thereby integrating the concepts of both environmental protection and sustainable use (EC 2020b).

The MSFD has a cyclic nature and includes several steps to achieve GES:
- The first step is the initial assessment of the environmental status of the national marine waters and the environmental impacts of human activities as well as socio-economic analysis of these activities in 2012 (art 8). Also, GES has to be determined for national marine waters and targets and indicators for GES have to be established (art 9, 10).
- The second step is the establishment of a monitoring programme for assessment and updates of targets in 2014 (art 11).

- The third step is the development of a "Programme of Measures" (PoMs) in 2015 in order to achieve GES (art 13).
- The fourth step is the implementation of the Marine Strategy in 2016 (art 5).
- The fifth and final step is the assessment of the different elements of the Marine Strategy and the start of a second cycle in 2018 (art 18). (EC 2020b)

Article 13.4 of the MSFD specifically requires the use of spatial protection measures contributing to the creation of coherent and representative MPA networks that protect the diversity of the marine ecosystems together with MPAs designated under the Habitats and Birds Directives. The designation of MPAs is an important contribution to the achievement of GES (EC 2008). Thus, together with the Habitats and Birds Directives, the MSFD plays an important part in the establishment of an ecologically coherent MPA network in the European marine environment.

The MSFD also requires Member States to coordinate their Marine Strategies in the European marine regions using the existing Regional Sea Conventions (RSC) (EC 2008). The European marine environment is divided into 4 regions all regulated under a different RSC. Via the RSCs, Member States cooperate and coordinate their Marine Strategies in order to achieve GES at a regional level across the EU.

According to the MSFD Member States must also keep in mind sustainable development and the social and economic impacts of the measures. Therefore, impact assessment and CBAs have to be carried out by Member States prior to the introduction of new measures (EC 2008).

"Member States shall ensure that measures are cost-effective and technically feasible, and shall carry out impact assessments, including cost-benefit analyses, prior to the introduction of any new measure." (EC 2008)

Moreover, the MSFD (article 14.4) mentions that exceptions can be made if there are disproportionate costs of a measure and achieving GES will not be permanently compromised if the measure is not implemented. Thus, for Member States performing CBAs it is of the utmost importance to be able to demonstrate the benefits of measures.

2.4 The Common Fisheries Policy

The Common Fisheries Policy (CFP) is the legal framework for managing the European fishing fleet and the conservation of fish stocks. The CFP should make sure that fisheries and aquaculture keep in mind the long-term ecological, economic and social sustainability. Fisheries management includes: rules on access to waters, fishing effort controls, and technical measures, such as gear usage, and the management of the timing and location of fisheries activities.

The CFP is an exclusive competence of the EU, therefore, the EU decides on any fisheries related measure. Legal measures such as fisheries restrictions within an area because of MPA designation must therefore be regulated through the CFP (EC 2013).

2.5 The Regional Sea Conventions

The RSCs are cooperation structures that make agreements in order to protect the European marine environment. Contracting Parties include both EU Member States as well as third countries that surround the marine regions (figure 1) and the European Union (except in case of the Bucharest Convention, see 1.5.4) (EC 2019). The MSFD requires that Member States coordinate marine strategies among themselves and among third countries by using the RSCs (EC 2008).

By improving cooperation and coherence between states, the RSCs can support the implementation of the MSFD. The task of the RSCs is to protect the marine environment under their jurisdiction while making sure that marine resources are used in a sustainable way. In order to do that, the RSCs work on measures that for example address eutrophication, marine litter, underwater noise and various other impacts (EC 2019).

The RSCs also make significant efforts in order to protect the marine environment via the establishment and assessments of MPAs. The RSCs and their role in the establishment of MPAs will be briefly discussed.

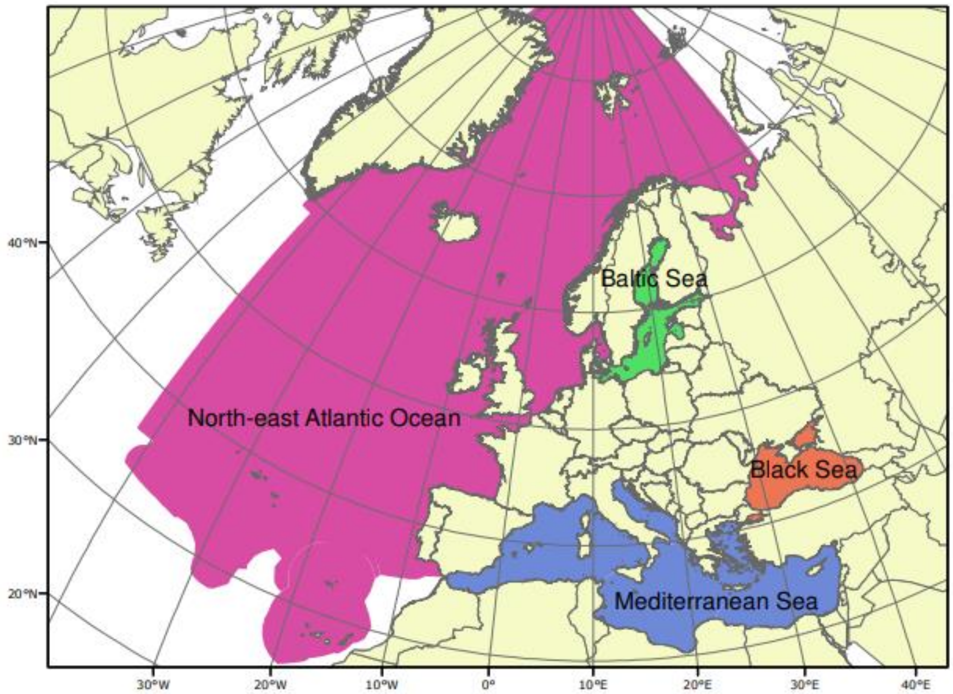


Figure 1. The marine regions under the Marine Strategy Framework Directive (EEA 2017).

2.5.1 *The OSPAR Convention*

OSPAR is the regional sea convention that aims at protecting the North-East Atlantic, which represents the biggest part of the European marine waters. The contracting parties are: Belgium, Denmark, Finland, France, Germany, Iceland, Ireland, Luxembourg, the Netherlands, Norway, Portugal, Spain, Sweden, Switzerland, the United Kingdom and the EU (OSPAR n.d.). OSPAR's goals are described in their strategy plan, the North-East Atlantic Environment Strategy. This strategy is divided into 5 sub strategies that address different threats to the marine environment. One of these strategies addresses biodiversity and ecosystems. Within this sub strategy, OSPAR declares that it aims to strengthen the OSPAR MPA network because of its contribution to maintaining ecosystem integrity and to the resilience against human impacts and climate change (OSPAR 2010).

2.5.2 *The HELCOM Convention*

HELCOM is the RSC of the Baltic Sea. The contracting parties are: Denmark, Estonia, Finland, Germany, Latvia, Lithuania, Poland, Russia, Sweden and the EU (HELCOM n.d.). HELCOM describes their plans for the Baltic Sea in the Baltic Sea Action Plan. This plan contains 4 themes. One of these themes addresses the conservation of biodiversity and nature. In this part HELCOM describes to designate additional MPAs and to collaborate closely with Contracting Parties in order to implement measures for fisheries inside MPAs and fulfill conservation targets (HELCOM 2007).

2.5.3 *The Barcelona Convention*

The Barcelona Convention is the RSC of the Mediterranean Sea (EC 2019). The Contracting Parties of the Barcelona Convention are: Albania, Algeria, Bosnia-Herzegovina, Croatia, Cyprus, Egypt, France, Greece, Israel, Italy, Lebanon, Libya, Malta, Monaco, Morocco, Montenegro, Slovenia, Spain, Syria, Tunisia, Turkey and the EU (RAC/SPA n.d.). Together, the Contracting Parties agreed on the Mediterranean Action Plan (MAP). This includes 7 protocols, which form the legal framework of the Barcelona Convention. One of these protocols is the SPA/BD protocol. Its goal is to protect biological diversity within the Mediterranean Sea by means of the creation of Specially Protected Areas, establishing a list of Specially Protected Areas of Mediterranean importance, and the protection and conservation of species (UNEP 1995)

2.5.4 *The Bucharest Convention*

The Bucharest Convention is the RSC of the Black Sea. The Contracting Parties of the Bucharest Convention are: Russia, Turkey, Ukraine, Georgia, Bulgaria and Romania. The EU expressed its wish to become a Contracting Party in order to better coordinate the implementation of the MSFD and support further cooperation in the region (EC 2019). MPAs designated in the Black Sea will not be discussed in this report since there is no overview of MPAs in the Black Sea available (EEA 2015b).

3 Assessing Marine Protected Areas in Europe

This chapter will examine what the current status of MPAs in the EU is, to see whether conservation goals set for the EU are being achieved. This chapter will also discuss criteria that could influence MPA effectiveness in order to provide answers on how MPAs and the MPA networks could be improved. Over the last couple of years, the RSCs assessed their MPA networks and some of their assessment methods and findings will be discussed as well. That way, this assessment may provide insights for future MPAs which in turn is essential for efficacy and delivery of ecological, social and economic benefits.

3.1 Establishment of Marine Protected Areas

The RSCs have made considerable efforts to establish MPAs in the main marine regions in Europe. The RSCs also expressed their willingness to protect at least 10% of the marine regions and sub-regions. However, these goals were set prior to the latest update of the EU Biodiversity Strategy which aims to protect 30% of the marine environment within MPAs of which 10% should offer strict protection (EC 2020a).

The situation in the various regional seas with respect to the absolute and size of MPAs are as follows:

- According to OSPAR’s report in 2018, the OSPAR MPA network is comprised of 496 MPAs including 10 MPAs designated in areas beyond the limits of the national exclusive economic zones. Together, these MPAs cover 864,337 km2 which is 6.4% of the OSPAR marine area (Annex 1) (OSPAR 2018).
- The latest HELCOM assessment report dates from 2016. In 2016, 174 HELCOM MPAs covered 49107 km2 of the Baltic Sea. This is 11.8% of the surface area, thus HELCOM already reached the 10% target set by the CBD in terms of overall coverage (Annex 2) (HELCOM 2016).
- In 2016, there were 1215 MPAs designated in the Mediterranean Sea. Together, these MPAs covered 171362 km2 of the marine surface area which is 6.81% (MedPAN 2019).

These percentages make it clear that conservation goals set by the CBD were not achieved in the North East Atlantic and in the Mediterranean Sea. In order to achieve the latest conservation goals even more areas will need to be designated as MPAs, that is why much more effort is required in the coming years in order to achieve conservation goals.

3.2 Protection level of Marine Protected Areas

MPAs vary greatly in the level of protection. In multiple use MPAs not all activities are forbidden, while in marine reserves or no-take MPAs all extraction is forbidden (Fenberg et al. 2012). These two MPA types can actually be further divided into a variety of different categories which allow different kind of human activities that have more or less severe impacts on the marine ecosystem (annex 3) (Dudley 2013). However, these variety of categories and the differences between human impacts in these areas will not be discussed as research on MPA benefits tend to refer to no-take MPAs and multiple use MPAs without further categorization.

Multiple use MPAs often allow activities such as recreational fishing, subsistence fishing, or less destructive types of fishing (Lester and Halpern 2008). Although multiple use MPAs could also generate ecological effects, marine reserves generally generate greater ecological effects than multiple use MPAs (Edgar et al. 2014; Shears et al. 2006). The ecological effects that have been observed most often are: an increase in biomass, higher densities, larger size of species within MPAs and higher species richness (i.e. the number of species) within MPAs (Fenberg et al. 2012).

Even though marine reserves generate greater ecological effects, prohibiting all extractive activities in specific areas can generate significant socio-economic costs. For example, local fishers may be affected negatively by the loss of fishing areas, at least in the short-term. Implementation of no-take MPAs may sometimes be preferred, because of less complicated regulations and enforcement. However, in general, marine reserves can face heavy resistance by extractive users. This makes the implementation of no-take MPAs polarizing and politically difficult (Lester et al. 2009). Stakeholder consultation could facilitate this process as MPAs supported by local communities are far more likely to be successful (Dudley 2013). However, in areas where stakes are high, less restrictive types of MPAs are often seen as a more feasible strategy and a way to compromise (Shears et al. 2006).

In Europe, only a very small fraction (+/- 0,5%) of MPAs provide full protection against all extracting activities. These fully protected or no-take MPAs are of great importance for marine biodiversity and are also more likely to be successful (Fenberg et al. 2012). Many MPAs in Europe are multiple use MPAs in which many activities are permitted. A recent study assessed 727 MPAs designated in the EU, and found that commercial trawling was happening in the majority of these MPAs (59%). The data also demonstrated that trawling intensity was actually higher inside MPAs, in comparison to non-protected areas (Dureuil et al. 2018). Major shipping lines may also intersect MPAs. For example, in the Baltic Sea, some major shipping lines go through MPAs or are very close to MPAs and fishing activities take place within MPAs (HELCOM 2016). In the Mediterranean Sea, the coverage of areas with strong protection levels (no-go, no-take, and no-fishing) cover only 945,67 km2 of the 171362 km2 which is 0.54% of the MPAs (MedPAN 2019). Several studies therefore suggest that targets should be set and no-take MPAs should be established (Fenberg et al. 2012; The Benyon Review Panel 2020).

Prohibiting all activities in all MPAs is unlikely to happen as the marine environment is essential for many human activities. That is why currently, there are very few no-take zones established in Europe. However, just recently the need to establish no-take zones has been recognized in marine policy in the updated European Biodiversity Strategy (EC 2020a). This view is supported by research and an important step forward in the protection of marine ecosystems.

3.3 Marine Protected Area size

A frequently discussed factor determining the success of an MPA is its size. Various theoretical studies suggest that size is a very important factor, since small MPAs are for example not capable of supporting population persistence, especially populations of more mobile species (Hasting and Botsford 1999; Roberts et al. 2003).

However, empirical studies provide mixed findings. On the one hand, it has been suggested that effectiveness of MPAs is independent of size, suggesting that multiple small MPAs would have the same effects as one single large reserve (Halpern 2003). Moreover, multiple small MPAs would be better at exporting larvae and adults, because of the larger edge to area ratios. This may provide local benefits to fisheries through spill-over effects (Vandeperre et al. 2011).

On the other hand, the results show that effectiveness increases more per increase of size. This means that a single large reserve would have more effects than multiple small ones that cover the same amount of surface area (Claudet et al. 2008). These contradictive results may, however, also be the result of other factors that determine effectiveness. For example, it has been suggested that small MPAs can be successful if they are supplied with recruits from nearby populations. MPAs in heavily impacted areas are less likely to be supplied with recruits from nearby populations as habitats are more degraded in the surrounding areas. Therefore, larger MPAs are probably necessary in areas that are more heavily impacted (Roberts and Hawkins 2000; Roberts et al. 2010).

MPA governance and public acceptability may also affect MPA size. For example, a single large MPA can be preferred because it is easier to enforce than multiple small MPAs. However, large MPAs may face more resistance, especially if the proposed area is used intensively (Roberts et al 2003).

MPA size thus can affect its success, however, determining what MPA size is sufficient to support ecosystem functioning remains a challenge and depends on conservation targets. Nevertheless, efforts have been made to determine size criteria. The European Environmental Agency for example, sets the target at a minimum of 100 km2 because it was demonstrated that 100 km2 was sufficient for supporting shallow-reef fish communities (Edgar et al. 2014; EEA 2015b). At the same time, HELCOM sets the target at a minimum of 30 km2 for at least 80% of MPAs (HELCOM 2016). In the Mediterranean, no specific targets were set as it was considered that judgement based on size alone would not determine whether an MPA is adequate. However, in order to compare the Mediterranean MPA network with the HELCOM MPA network, an assessment was carried out based on the same criteria. EEA concluded that a vast majority (>85%) of European MPAs do not meet the 100 km2 target (EEA 2015b). HELCOM concluded that their target of at least 80% of MPAs larger than 30 km2 was not met as 68% of their MPAs reached this target. However, the majority of HELCOM MPAs range between 100km2 and 1000 km2. Thus, even though the target was not reached, in general, HELCOM MPAs are of considerable size (HELCOM 2016). Mediterranean MPAs are in general much smaller than HELCOM MPAs, only 31.54% of Mediterranean MPAs are larger than 30 km2. Moreover, most MPAs in the Mediterranean are smaller than 10 km2 (MedPAN 2019). For OSPAR MPAs, no specific targets were set or analysed. The latest OSPAR report mentioned that size should be determined by the purpose of the site and it should be large enough to maintain the features for which it was selected (OSPAR 2018).

Based on the work performed by the EEA it can be concluded that in general the vast majority of MPAs are smaller than 100 km2 (EEA 2015b). However, the optimal MPA size always depends on conservation targets. The protection of highly mobile species for example, requires a larger MPA (Fenberg et al. 2012). Moreover, connectivity to other populations, together with the human activities that take place in the MPA, can determine the size that can be considered to be appropriate (Roberts and Hawkins 2000). Therefore, it is almost impossible to set a single target for MPA size as this should be determined based on conservation goals, connectivity and human expected impacts in the selected area.

3.4 Networks of Marine Protected Areas

Throughout their life, many marine species use multiple habitats and move actively or passively (via currents) from one habitat to another (Med PAN 2019). Therefore, networks of MPAs are advocated because often a single reserve cannot encompass all the habitats necessary for vulnerable life stages (e.g. spawning grounds, nursery areas) (Roberts et al. 2003). Although some populations have little exchange with other populations and depend greatly on self-recruitment, others are of much importance to surrounding populations. For example, by exporting a large number of individuals to surrounding populations and vice versa (Rossi et al. 2014). Also, multiple MPAs can spread risks in case of extreme catastrophic events. In that way, affected populations in one MPA can be replenished by other populations in undisturbed MPAs which contributes to the resilience of the MPA network. These exchanges between populations can greatly affect population growth and mortality rates. Moreover, evolutionary processes can be influenced because connectivity increases genetic diversity and thereby promotes adaptation under harsh conditions (Hastings and Botsford 2006).

Connectivity of MPA networks is therefore widely recognized as an important factor for MPA success and mentioned as an important factor contributing to the ecological coherence of MPA networks (Rees et al. 2018). OSPAR assessed connectivity by looking at the distribution of their MPAs in order to see whether there are major gaps in the network. For this analysis maximum distances were determined for the coastline (250 km), the EEZ (500 km) and offshore areas or the high seas (1000 km). This scale is thus smaller as one moves inshore, because the scale of ecological processes become smaller as one moves inshore. For example, because species of commercial interest offshore are more mobile than species nearshore and can move distances up to hundreds of km seasonally (Roberts et al. 2010). Based on their assessment, OSPAR concluded that major gaps still exist, especially in area I, the Arctic (OSPAR 2018).

HELCOM performed a similar theoretical analysis but took this one step further by also taking species specific dispersal distances into account. Theoretical connectivity was assessed by looking at the distance between similar benthic marine landscapes within 25 km and 50 km as a second level. These targets are assumed to be met if 50% or more of the benthic marine landscapes had at least 20 connections within these theoretical dispersal ranges. For species-specific connectivity the assessment was based on the distance between suitable habitats that support 5 different species with different dispersal distances. Here, as well as in the theoretical assessment, connectivity is assumed to be achieved when 50% of the suitable habitats had at least 20 connections within the species-specific dispersal distance (annex 4). Theoretical connectivity was not achieved for any of the marine benthic landscapes, not within 25 km and not within 50 km. Species-specific connectivity was not achieved for 1 of the 5 species which could be explained by its very short dispersal distance of 1 km while the other species had a dispersal distance of at least 25km (HELCOM 2016).

In the Mediterranean, connectivity was also assessed by looking at distances between benthic marine landscapes. No specific target was set for a minimum number of connections but in order to compare to HELCOM they tested whether 50% of the benthic marine landscapes had at least 20 connections within 25 and 50km. For none of the benthic marine landscapes the target was reached (MedPAN 2019).

The above description shows that connectivity is assessed differently by the various RSCs. However, all concluded that their MPA network cannot be considered connected based on their own criteria, and they all acknowledge that more information is needed in order to better assess connectivity. HELCOM took the first step by taking into account species specific dispersal distances. In the Mediterranean it was acknowledged that a species approach could be the next step and that a regional analysis which is performed by OSPAR could improve this analysis (MedPAN 2019).

OSPAR also recognized that species specific connectivity based on life history traits could improve their analyses as it would make the analysis more scientific and ecologically robust (OSPAR 2018). Another way in which assessing connectivity can be improved is via genetic research as genetic similarity/dissimilarity between populations can be used to assess gene flow (i.e. the transfer of genetic information from one population to another). Information about dispersal distances and connectivity between populations or habitat patches can be analysed using intraspecific genetic data. This may also help identifying source populations from which recruits are exported to nearby populations (Jenkins and Stevens 2018).

What is clear that connectivity in the marine environment is difficult to assess because it is characterized by openness and complex dynamics. Nevertheless, multiple methods have been used to assess MPA network connectivity. These methods are mutually supportive and provide different information which can be used to assess and improve MPA networks.

3.5 Protecting biodiversity

It has been argued that MPAs should cover all the different biogeographic regions because species assemblage will be unique in each one of them (WWF 2000). Moreover, biogeographic regions represent a wide distribution of habitats. Habitats can be defined based on their physical characteristics or by their biological attributes (e.g. seagrass beds are defined based on the dominant role of seagrasses).

The value of an MPA increases as the variety of features it protects increases. As habitats are often used as a proxy for species richness, including more habitats can make sure that all features are protected. This is not a linear relationship, because not all habitats are equal indicators of species richness. For example, tropical regions are almost always more species rich than temperate regions. This means that habitats in different biogeographic regions cannot be compared directly. Therefore, habitat heterogeneity should be compared between sites within regions (Roberts et al. 2003). Determining where to place and what to protect with MPAs is thus crucial for meeting conservation goals as each habitat supports unique communities. Moreover, most marine organisms use multiple habitats during their live. Therefore, MPAs should contain many different habitats to maintain biodiversity (IUCN-WCPA 2008). However, recent findings in the Baltic Sea find that habitats are not always the best representation of species. Based on data assessed before (>140.000 samples) it was concluded that including both species specific data and environmental data together should be used to make sure that all features are protected (Virtanen et al. 2018). However, in the marine context data is scarce, protecting habitats is therefore often used as a proxy instead (Foster et al. 2017).

The need for protecting the full range of biodiversity is highlighted in the MSFD as the cornerstone for achieving GES and is also acknowledged by the RSCs in their assessments (EC 2020b). OSPAR, HELCOM and the Barcelona Convention all use a more or less similar approaches to assess whether their MPA networks are located correctly and protect all important features. All RSCs assessed their MPA network at a regional scale and concluded that regions further offshore are not protected as well as nearshore regions (figure 2).

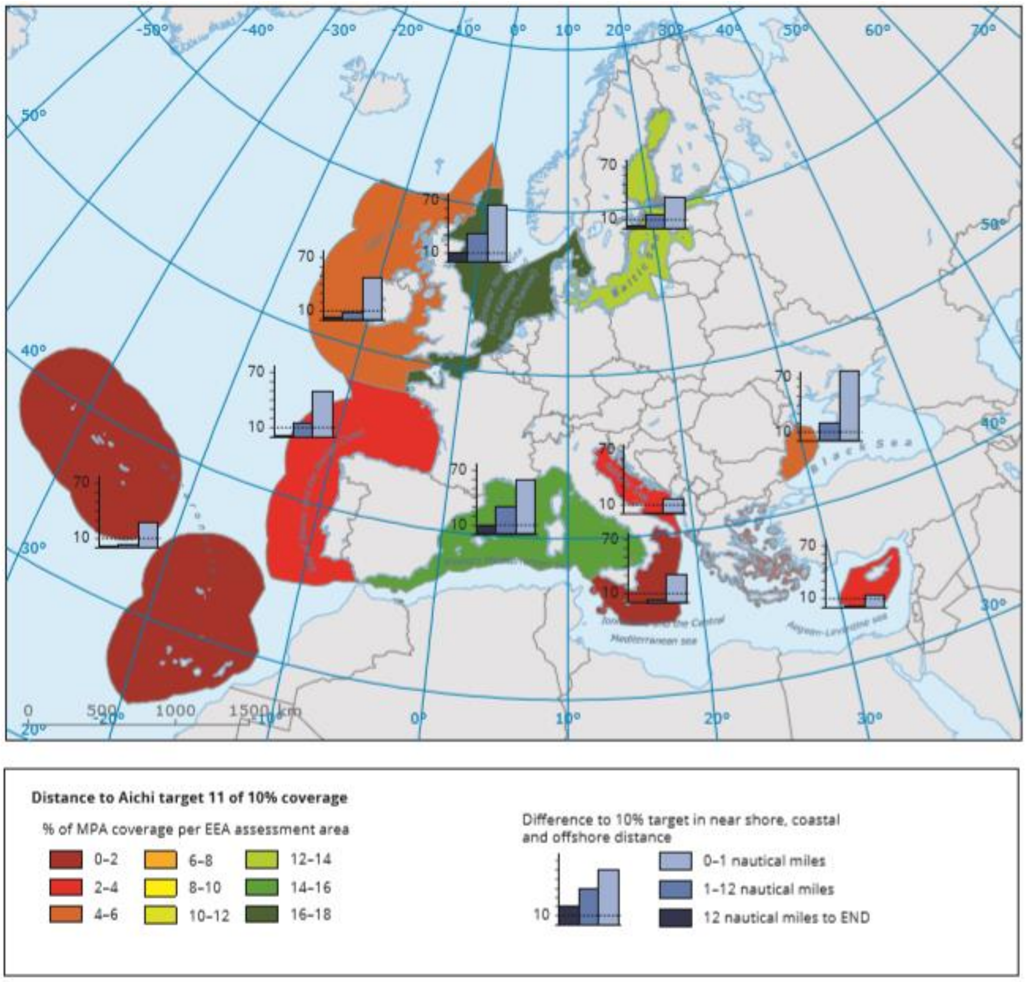


Figure 2. Overview of MPA coverage in marine waters of Member States. Graph shows difference between MPA coverage in nearshore marine waters to further offshore (0-1, 1-12 and >12 nautical miles) (EEA 2015b).

The RSCs also concluded that not all features are protected within their MPA networks. The same was concluded for replication which is the protection of certain features in multiple MPAs (OSPAR 2018; HELCOM 2016; MedPAN 2019). Since replication can be used as a proxy for resilience of the MPA network, it can be concluded from the fact replication is insufficient, that also resilience can be considered to be insufficient. Therefore, further efforts are required to protect specific habitats and/or species in multiple MPAs.

3.6 Ecosystem recovery takes time

MPA age or time since establishment can have great influence on the effects of MPAs as it takes time for ecosystems to recover from degradation (Claudet et al. 2008). The expected time before effects of assigning MPAs can be observed can affect the willingness to accept/support them. Support will be much greater if benefits are expected quickly after establishment (Roberts and Hawkins 2000). A rapid increase in biomass is expected when there is protection from fishing. Recruitment is not essential for biomass increase because growth of present individuals will assure this. However, both growth of reproductive animals and recruitment are required for fisheries to benefit from establishing MPAs next to their fishing grounds as this is a prerequisite for spill-over effects to surrounding areas. As a consequence, the benefits for fisheries lag behind biomass increase (Roberts and Hawkins 2000). Moreover, these effects depend on life history characteristics of the target species as it may take years or even decades for long-living slow-growing species to recover (Fenberg et al. 2012). It also depends on other factors, for example how depleted population were to begin with (Roberts and Hawkins 2000).

Empirical evidence indeed demonstrated that age significantly contributed to the ecological impact observed in MPAs. A global analysis of 87 MPAs revealed that MPA effects such as increases in size and biomass depend heavily on time since establishment and that especially old MPAs (>10 years) generate large conservation effects (Edgar et al. 2014). Thus, assessments on the effectiveness of MPAs will depend heavily on the time since establishment in combination with other factors such as life history characteristics and the state of the inhabiting populations in the MPA.

In 1995 the first MPAs were established and added to the Natura 2000 network. Therefore, some effects can be expected by now (EEA 2015b). The MSFD was established in 2008, but Member States had up to 2016 to implement their PoMs (EC 2020b). Therefore, it is likely that it may still take a few years before effects can be noticed in many MPAs that were established under the MSFD. This should also be taken into account when future MPAs are being implemented as it will influence the acceptance of MPAs. It may take years before benefits of MPAs are being noticed in surrounding fishing areas. When fishermen are highly dependent on fishing for their livelihoods it is likely that they will not support MPAs no matter how much they would like to benefit from MPAs in the future. Therefore, even though MPAs could possibly create benefits, they may entail costs in the short-term which should be taken into account in socio-economic analyses.

3.7 Management of Marine Protected Areas

Management of MPAs is key for achieving conservation goals. In fact, it has been suggested as one the major factors determining MPA effectiveness. Based on MPAs worldwide it was concluded that many MPAs are not effectively managed which has major effects on their effectiveness. Especially staff and budget capacity were inadequate and a strong predictor for conservation impacts. Ecological effects were 2.9 times greater in MPAs with adequate staff capacity compared to those lacking staff capacity. Improper MPA management may thus undermine MPA effectiveness and the benefits that they provide (Gill et al. 2017). Inadequate resources may also result in inadequate monitoring. Monitoring is needed in order to establish baseline data and assess performance over time. Streamlined monitoring protocols can be used to compare MPA performance. Results must also be reported as this increases transparency and enables information sharing on management approaches and their effectiveness (OECD 2016).

The importance of management is also recognized by the RSCs. Based on results from questionnaires sent to MPA managers OSPAR concluded that management is not optimal. The questionnaire made it clear that for a part of the MPAs no management was documented, nor were measures implemented or monitoring taking place in many MPAs. Also, for many MPAs it is not clear whether they are moving to or have already achieved their conservation objectives (OSPAR 2018). A similar approach was used in the Mediterranean Sea, where results also suggested that ecological data were missing, there are no management plans or they are not fully implemented, and resources such as staff and budget are insufficient. Also, monitoring is not taking place or only taking place sporadically (MedPAN 2019). Based on their MPA database, HELCOM also concluded that many MPAs don't have management plans and that in many MPAs with management plans, monitoring is not taking place, which makes it very difficult to assess their effectiveness (HELCOM 2016). From the above it appears that MPAs in Europe are apparently not managed properly. This undermines the conservation goals set by the RSCs and the EU. Increased investment in MPAs is therefore necessary. This could potentially result in higher returns on investment for both nature and people (Gill et al. 2017).

3.8 Concluding remarks

It is generally understood that many factors influence MPA performance. This has been studied widely and is recognized by the RSCs in their MPA network analyses. Factors that affect MPA and MPA network performance include: overall coverage, connectivity, size, representativity, protection level and management. However, marine ecosystems are characterized by complexity and data is often scarce. This makes it very difficult to determine specific criteria which are necessary to assess and establish an ecologically coherent network of MPAs. Nevertheless, efforts have been made to assess MPA networks.

What can be concluded from these analyses is that currently the European MPA network cannot be considered an ecologically coherent network of MPAs. Currently, the target of protecting 10% of the marine environment within MPAs has not been achieved except in the Baltic Sea. The updated target of the European Biodiversity Strategy that aims at protecting at least 30% of the marine environment requires even more effort and emphasizes the importance of MPAs. The representation of different regions, habitats and species that are protected within MPAs also varies. Management is also insufficient in many cases.

Furthermore, it is recommended that parts of existing MPAs become no-take MPAs as these generally perform better than multiple use MPAs. These sites can also serve as reference sites in order to assess the impacts of human activities and other environmental changes. The importance of such areas is recently acknowledged in the European Biodiversity Strategy by setting a target of protecting 10% of the marine environment within highly protected MPAs.

Finally, establishing criteria for both size and connectivity is very difficult. Single large MPAs may be needed in areas where there is no replenishment of nearby populations while smaller MPAs may be large enough when they are replenished by nearby populations. This emphasizes the need to assess connectivity in marine ecosystems which has been done differently by the RSCs. Each of these assessments provides different information and they can be mutually supportive. The developments in genetic research could also be used to get a better understanding of connectivity in marine populations.

Based on these findings it is very likely that the European MPA network is currently underperforming. An increased understanding of MPAs is needed in order to improve MPA networks which in turn contributes to overall performance and the benefits that can be expected from them. Some of these benefits will be discussed in the next chapter.

4 The benefits of Marine Protected Areas in socio-economic analyses

The conservation, protection and valuation of nature is becoming increasingly important (OECD 2015). More and more efforts are made to examine the connections between ecosystems and human wellbeing. It is understood that economic development is subject to the limits of nature (Vassallo et al. 2013). Socio-economic analyses provide a community perspective that is otherwise difficult to determine. An understanding of the distribution of both costs and benefits across society can support decision-making processes that ensure equitable outcomes (Davis et al. 2019). As for MPAs, the designation of an MPA or an MPA network can have consequences for society, in particular for fisheries, industry and other practices such as recreation, especially if measures are strict. Socio-economic analyses can particularly support implementation if it is demonstrated that positive impacts or benefits that arise through the designation of an MPA outweigh costs. However, MPAs are also a highly debated management strategy as it is often unclear what the benefits and the magnitude of the benefits are (Davis et al. 2019).

At the same time, socio-economic analyses are an integral part of the MSFD. For example, Member States are required to perform an economic analysis of the activities in their marine area, and assess the costs of degradation. Also, impact assessments and CBA must be carried out prior to the introduction of a new measure. Moreover, exceptions may be justified if it turns out that the costs of measures are disproportionately high, while making sure that there is no further degradation (EC 2008). The MSFD thus highlights socio-economic analyses as an important part in decision making processes.

Estimating the benefits of MPAs is considered to be very challenging as it is often unclear what those benefits are and how they can be quantified or monetised (Bertram et al. 2014). The first part of this chapter will therefore start with a description of some of the benefits of MPAs, including both ecological and economic benefits. The second part of this chapter will describe methods that have been used to quantify and/or monetise benefits of MPAs in socio-economic analyses and are supported by examples. Finally, the chapter will end with a discussion on the various approaches and suggest one that will be applied on the Borkum Reef Ground, a possible future MPA in the Netherlands. This will be described in the next chapter.

4.1 Benefits of Marine Protected Areas

4.1.1 Ecological benefits

MPAs can generate great ecological effects. Many of the studies examining MPA effects find that MPAs affect size and abundance of species within MPAs. For example, a before and after study on an MPA that completely protected shellfish and partially protected fish showed that after 6 years the catch per unit effort of European lobster had increased by 245% and Atlantic cod were on average 5cm longer (Moland et al. 2013). Another study in New Zealand revealed that the biomass of legal size spiny lobsters was 25 times higher within the no-take MPA compared to areas outside the MPA (Shears et al. 2006). Data from a study in a no-take MPA in France suggested that abundance of rocky reef fish increased after the establishment of the MPA. Data was collected before and after the MPA establishment, in areas within and outside the MPA. Before the establishment of the MPA there were no differences in fish abundancy between the MPA and surrounding areas. Six years after the establishment of the MPA fish became significantly more abundant within the MPA compared to outside the MPA. Especially large fish and medium sized fish became more abundant. Fished species seem to benefit most from the MPA (Claudet et al. 2006). In the United Kingdom positive effects after MPA establishment have also been observed. In Lyme Bay additional MPAs were assigned that offer protection against bottom-towed fishing gear, creating a single large MPA of 206 km2. Significant positive changes in biodiversity, abundance and size have been observed 3 years after establishment in benthos and mobile species. These changes were more pronounced in the closed areas compared to the open fishing areas. Moreover, the recently closed areas were becoming more dissimilar to the open areas and more similar to the areas that have been closed for 5 to 10 years. The areas that were closed for a longer period also showed significant positive changes, indicating that these areas were still recovering (Atrill et al. 2012).

Increases in abundancy and size can in turn also contribute to the resilience of marine ecosystems. In Mexico, the coastal area faces catastrophic events such as periods of hypoxia (low oxygen levels) that in turn, can cause mortality of marine species. After an hypoxia outbreak abalone populations within the MPA remained stable while populations outside the MPA suffered immensely from this event. The MPA contributed to this because density of abalones was higher in the MPA and abalones were generally larger in the MPA which in turn affects the reproductive output of this species (Micheli et al. 2012). Indeed, size contributes to the reproductive output of species. In red snappers for example, a single female of 12,5 kg produces the same amount of eggs as 212 females of 1,1 kg (Pauly et al. 2002). Similar results were found in tropical groupers, where egg production increases exponentially with length. By allowing fish to increase in size, MPAs can contribute to reproductive output and possibly future population persistence (Roberts et al. 2017). Egg quality is also influenced by age, older individuals usually produce eggs of higher quality than younger ones. Fishing is known for affecting age structure of populations which in turn affects temporal variability. Exploited species therefore show higher temporal variability than unexploited species (Hsieh et al. 2006). MPAs also allow populations to increase which makes sure that genetic diversity is maintained which also has a positive effect on adaptability and resilience (Roberts et al. 2017). MPAs may thus promote larger populations with larger and older individuals which in turn makes sure that decline can be buffered and enhanced reproductive output facilitates recovery. These factors together make sure that populations are more resilient to human pressures and sudden events.

There are many examples of MPA effects through increases in size and abundance of fished species (Halpern and Warner 2002). Generally, species targeted by exploitation respond more than non-targeted species. These are often larger species higher up in the food chain (Côté et al. 2001).

Changes in higher trophic levels can in turn affect lower levels. Changes in food web structure are called trophic cascades, which can have far reaching effects. In New Zealand, a so called trophic cascade has been observed after the establishment of MPAs. It was expected that fish and lobsters would become larger and more abundant after MPA establishment. However, because fish and lobsters became larger, they started to predate more on small urchins. Larger urchins were too big to predate on, but once they died out the grazing pressures on the rocks decreased and several seaweed species appeared. These effects have been observed in other areas in New Zealand as well (Ballantine 2014). MPAs located in the Great Barrier Reef had a similar effect. There, it was demonstrated that there were significant less outbreaks of coral eating crown-of-thorns starfish because of increased predation pressure within MPAs.

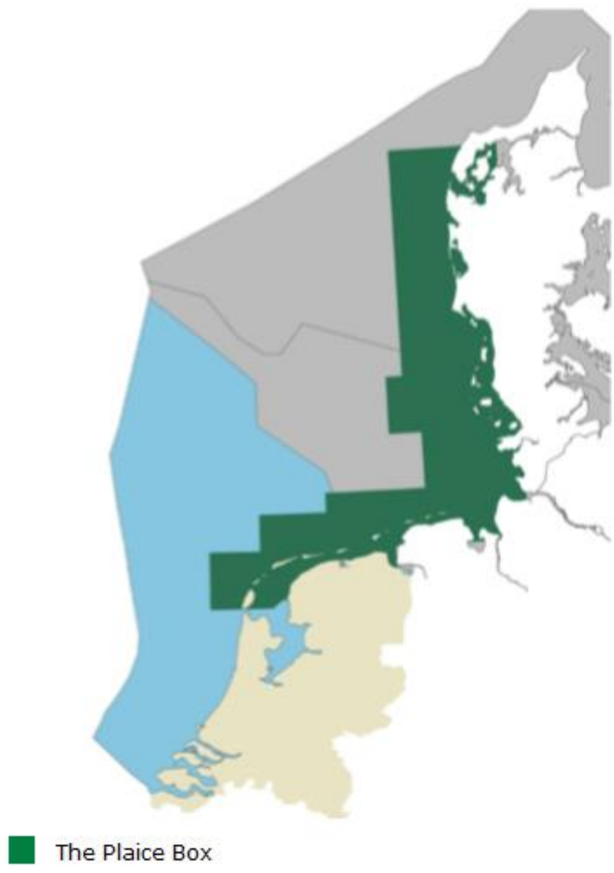
These outbreaks have direct effects on coral reefs. After outbreak periods the coverage of coral was markedly higher in MPAs compared to fished areas (McCook et al. 2010). These phenomena have been observed in Mediterranean MPAs as well. Here, it was demonstrated that MPAs contributed to the recovery of fish that in turn affect sea urchin densities. However, this was a non-linear response which suggested that a certain predator density is needed before effects on lower trophic levels can be expected (Guidetti and Sala 2007). Similar phenomena have been observed in the Caribbean and in North America as well (Leleu et al. 2012). These experiences made it clear that changes in trophic levels in turn can affect whole ecosystems and its functioning many years after establishment. Also, most responses include major surprises, even in cases where conditions and activities were well-known (Ballantine 2014).

Although there are many examples of MPA success and their contribution to ecosystem recovery, MPAs may also fail. In the Netherlands the protection of the plaice via the so-called plaice box is still used to question the efficacy of spatial protection measures (box 2). A global analysis confirmed that MPAs do not always meet their objectives. In fact, more than half (59%) of the MPAs studied were ecologically similar to fished areas, indicating no conservation effects. However, the success of the MPAs turned out to be strongly affected by the degree of fishing allowed, management, age, size, and the presence of continuous habitat allowing movements across boundaries. This emphasizes that MPA design and management are key factors for success (Edgar et al. 2014).

The Plaice Box

The Plaice Box is an area of 40000 km2 located in the North Sea where fishing with large beam trawls (>221 kW) is prohibited in order to reduce discarding of undersized plaice. After the establishment in 1989 and contrary to expectations, the landings and biomass of plaice decreased. This resulted in a loss of credibility in MPAs in the Netherlands (Beare et al. 2013).

Nowadays, the Plaice Box is still a point of discussion. Fishermen use this example to show that MPAs do not work. They claim that bottom trawling makes sure that food is available for plaice. Conservationist on the other hand, claim that the Plaice Box was never protected to begin with because smaller boats were still allowed to fish in the area. Also, other environmental factors, such as eutrophication and temperature increases may have caused a general decline in plaice stocks. The exact cause of decline is still uncertain but this example clearly demonstrates how negative experiences affect the credibility of MPAs (Vrooman et al. 2018).



Box 2. Summary of the experience with the Plaice Box in the Netherlands. The plaice Box is indicated by the green area on the map (adapted from: Vrooman et al. 2018).

What can be concluded is that well designed, properly managed MPAs can have major positive effects by increases in size and abundancy of species within MPAs and by promoting biodiversity in general. This promotes population resilience and may also lead to cascading effects which can affect the whole ecosystem. However, marine ecosystems are characterized by their high degree of openness and complexity, the latter results in a certain degree of surprise and unpredictability (Franzese et al. 2017). Also, knowledge on marine ecosystems is often missing or insufficient, especially compared to terrestrial systems (Carr et al. 2003). Although many examples demonstrate MPA effectiveness, research on design as well as management in specific conditions is needed. That way, MPAs in general can be more successful and making predictions about the extent of effects will be more easy which in turn facilitates implementation.

4.1.2 Carbon storage and sequestration

In many countries carbon emissions are increasing as a result of economic development. In the meantime, natural ecosystems are degrading, which reduces their capacity to absorb CO_2 (Nelleman et al. 2009).

Marine ecosystems play a crucial role in the worlds capacity to take up carbon. Carbon uptake by marine ecosystems can be distinguished into carbon storage and carbon sequestration. Carbon storage means that carbon is stored in marine plants or organisms during its lifetime. Carbon sequestration means that carbon is sequestrated for millennia deep below the seabed where it can be decomposed (Thompson et al. 2017).

In the marine context there are multiple ways in which carbon can be stored and sequestrated. CO_2 can for example be converted into sugars by photosynthesizing plankton at the surface waters. Other sea creatures consume these organisms and transfer carbon from the surface waters to the deep sea via movements and trophic interactions (McCleod et al. 2011). Carbon can reach the sea bottom via multiple ways, for example, packaged into fecal pellets or because dead organisms sink to the bottom. This process in which carbon is transported to the sea bottom via marine organisms is also known as the biological pump (Thompson et al. 2017).

Another way in which carbon can be stored and sequestrated is via vegetated coastal systems. Especially vegetated coastal systems such as salt marshes, seagrasses and mangroves are known for being among the most efficient carbon sinks on the planet. Carbon is stored within the marine plants and sequestered for millennia in the underlying sediments (Roberts et al. 2017). In response to rising sea level these sediments accrete vertically which makes sure that they do not become saturated with carbon. For example, carbon has been found up to 10 meters below seagrass meadows (McCleod et al. 2011). Protection and restoration of these ecosystems is suggested to be crucial for the world’s capacity to mitigate the effects of emissions (Nelleman et al. 2009).

Within Europe, there are 4 species of seagrasses. These are neptune grass, seahorse grass, eelgrass and dwarf eelgrass. Of these 4 species Neptune grass is best at capturing CO2 from the atmosphere and most abundant in the Mediterranean Sea. In other parts of Europe eelgrass is more abundant (Lafolley and Grimsditch 2009).

Because of the capacity of saltmarshes and seagrasses to capture carbon an attempt has been made to estimate the value of saltmarshes and the seagrasses neptune grass and eelgrass in Europe. Also, the possible economic losses in different future scenarios were estimated for the period of 2010-2060 (Luisetti et al. 2013).

Past and current trends predict a decline in these essential ecosystems. Therefore predictions of future trends were made for future scenarios. The first scenario is based on continued application of current conservation policies and predicts relative small losses of 365 ha of seagrasses per year, while the other scenario predicts much greater losses of 45,000 ha per year. In the first scenario for saltmarshes it is predicted that 744 ha is lost per year. In the second more pessimistic scenario this is estimated at 744 ha per year for the first 20 year and 947 ha per year for the following 30 years is lost. The current economic value of this ecosystem is based on the average price of traded carbon (i.e. price of emission allowances companies receive or need to buy to cover their emissions). Monetary values on costs in future scenarios were derived from estimates on the social cost of carbon (i.e. cost caused by emitting one extra tonne of greenhouse gas) and on marginal abatement costs (i.e. the cost of reducing pollution).

Current economic value of carbon storage (i.e. economic value of carbon storage per year) in saltmarshes is estimated at €9,108,815. Based on the different cost estimates the future costs predicted in the first scenario ranged from €60,416 to €6,388,100. In the other more pessimistic scenario this ranged from €67,000 to €7,405,819.

Current economic value of seagrasses in Europe was estimated at €137,194,900. Based on the different cost estimates the future costs predicted in the first scenario ranged from €57,439 to €6,013,652. In the other more pessimistic scenario this ranged from €7,079,941 to €741,441,350. These results emphasize not only the current value of these ecosystems because of their contribution to carbon storage they also predict the possible future costs if they are not well protected (Luisetti et al. 2013).

Even though the importance of carbon storage in marine organisms is widely recognized, it is questioned whether it can be considered as long-term sequestration. Opponents claim that carbon storage in living organisms is a temporary solution (Thompson et al. 2017). Human activities such as bottom trawling and wetland drainage disturb these ecosystems. As a result, carbon will be released and the carbon sink will become a carbon source (Pendleton et al. 2012).

Disturbances related to climate change could both enhance or reduce the capacity of these ecosystems to capture and sequester carbon. Sea level rise could enhance these ecosystems because of increasing landward areal extent and increased vertical accretion. At the same time, seaward extent could be lost. Also, extreme storms could potentially increase sediment deposition which could lead to soil elevation and thereby reduce the effects of sea level rise while enhancing carbon storage and sequestration. On the other hand extreme storms could damage these ecosystems and the sediments which could lead to loss of carbon in the atmosphere. Whether or not climate change has a positive or negative effect on these ecosystems remains unclear and a point of discussion (Macreadie et al. 2019).

The capacity to sequester carbon (ability to store in underlying sediments) ranges immensely. This for example depends on salinity, nutrients and sediment supply. In salt marshes the estimates differ the most (18-1713 g C m^{-2} yr^{-2}), followed by mangroves (20-949 g C m^{-2} yr^{-2}) and seagrasses (45-190 g C m^{-2} yr^{-2}) (McCleod et al. 2011).

Even though there are many uncertainties on the capacity of marine ecosystems to store and sequester carbon their protection is very important. First of all because restoration of marine coastal systems is 10-400 times more expensive than restoration on land. Also, more than half of these marine restoration projects fail (Bayraktarov et al. 2015). So even though estimating the true potential of marine coastal systems is difficult, protecting them will probably be less expensive than restoring them.

In addition, marine coastal systems provide other ecosystems services as well. For example, marine coastal ecosystems offer coastal protection because they attenuate waves. They also prevent erosion as the roots stabilize the sediments. Water purification is another ecosystem service as these ecosystems take up excess nutrients and pollutants. These ecosystems also support marine life because they provide suitable reproductive habitats and nursery grounds (Barbier et al. 2011).

In conclusion, there remain large knowledge gaps on marine ecosystems and their capacity to mitigate emission effects. Further research should focus on carbon sequestration rates, the stability of carbon stocks and the human drivers that affect these ecosystems and their functioning (Thompson et al. 2017). What is clear is that these ecosystems play an important part in the global carbon cycle and provide many other ecosystem services as well (Barbier et al. 2011). MPAs could play a part in mitigating emission effects. They are not a substitute for greenhouse gas reductions. However, MPAs protect against many disturbing activities and could therefore ameliorate some of the effects associated with climate change and ensure continued ecosystem functioning (Roberts et al. 2017).

4.1.3 Possible future benefits for fisheries

MPAs worldwide have demonstrated that they affect size and abundancy of organisms within them. It is suggested that these increases in density and biomass can lead to spill-over effects to adjacent areas, thereby benefitting fisheries (Halpern et al. 2009). Spill-over effects are often observed as declining patterns of catches with increasing distance from MPA boundaries. Some studies suggest that these effects can take place.

For example, spill-over effects have been observed after the establishment of a no-take MPA near Sardinia. Here it was demonstrated that after 12 years biomass of the European Spiny Lobster had increased by 500% within the MPA. The most productive area was estimated to be within 6 km from the MPA border (Follesa et al. 2011). Catch per unit of effort increased the closer to the MPA border a vessel fished, suggesting that fisheries closer to the MPA boundary benefit from the spill-over effects from the MPA. In the northeast of the United States fishing effort and in particular catch per unit effort was studied in relation to distance from MPA, in which all gear for catching demersal fish is forbidden; the gear types that are allowed in the MPA are lobster traps, midwater trawls and limited dredge fishing for scallops. Catch data showed a negative trend with increasing distance from the MPA boundary. However, these effects were only found for part of the investigated demersal fish and within very short distance (<4km from MPA boundary) (Murawski et al. 2005).

Similar effects were observed in Mediterranean no-take MPAs where spill-over effects were noticed up to 800 meter from MPA boundaries on average (Goñi et al. 2008). It is likely that these results are an underestimation of the effects, as most studies use abundancy to measure spill-over effects. Catch data of fishers is probably more reliable because increased fishing effort around the MPA may result in lower abundancies, thereby masking the observed effects (Halpern et al. 2009).

Studies suggest that multiple mechanisms can contribute to the magnitude of spill-over effects and these are not guaranteed. It is for example demonstrated that spill-over effects depend on density, seasonal migrations and on suitable habitats in surrounding areas (Abesamis and Russ 2005) (McClanahan and Mangi 2000) (Forcada et al. 2009). Spill-over effects are also likely to differ between species as species with a larger home range are more likely to cross MPA borders (Goñi et al. 2008). Although spill-over effects can be beneficial, but they do not directly translate into economic compensation for fishers. For example, fishers may have to go to fishing grounds further away because of displacement. It also takes time for spill-over effects to accrue (Halpern et al. 2009). This all shows that spill-over effects are likely to accrue, however, they are a long-term benefit of MPAs, and it is unclear whether they can compensate the losses to fisheries, at least in the short-term.

This uncertainty also became clear in the UK where stakeholder perceptions were studied after the establishment of the MPA in Lyme Bay mentioned before. Questionnaires were used to examine the effects of the MPA on stakeholders and their level of support, over a third year period post-designation. The study interviewed fisherman, sea anglers, divers, dive businesses and charter boat operators. Results demonstrated that the majority of the stakeholders supported the MPA. Static gear fishermen inside the MPA reported increased catches and landings. Anglers reported improved fishing experience within the MPA. Most stakeholders also thought that the social and economic benefits outweighed the costs. However, the towed gear fishermen, which were banned from the MPA, and the static gear fisherman outside the MPA reported negative effects, such as increased costs and more conflicts due to displacement. Part of the fishermen (12%) also reported a change in fishing gear. Generally, fishers outside the MPA felt a sense of injustice and reported that it would be fairer to close the area for all kinds of fishing (Rees et al. 2013)(Mangi et al. 2012).

Perceptions of fishers of MPAs can also vary between different locations. The perceptions of stakeholders on MPAs located in the Western Mediterranean and the Atlantic Ocean were studied with a large scale field survey. In total, 9 MPAs were included in the survey for professional fishing and 354 responses were received from fishers fishing in and around these MPAs. Their answers revealed that the proximity of the MPA did not impact fishing strategies. Fish abundance was the main reason to fish in an area. If an MPA generates significant spill-over effects, fishers are likely to be attracted to the area. The majority of the fishers also declared that MPAs have a positive effect on biodiversity, except for fishers in Malta. The majority also thought that the MPAs have a positive effect on fish abundancy except for an MPA in Spain and one in Italy which had only 1 reply. Fishers were more sceptical about spill-over effects. In 2 case studies the majority of the fisherman were positive about this topic while the majority were negative (4 case studies). MPAs also do not seem to affect turnover, however, opinions vary a lot (Alban et al. 2008).

Fishers thus do not always support MPAs as it may take several years before they can benefit from them. If an MPA causes losses in income then fishermen may not support the MPA, even though they would like to benefit from it in the future (Roberts and Hawkins 2000). It also suggest that current fishers are not likely to benefit from MPAs while future generations may do so. Compensation for fishers may be necessary in such cases, providing that it does not promote overcapacity as this is one of the major threats to sustainability (Pauly et al. 2002).

4.1.4 *Benefits for recreation and tourism*

Economic benefits from tourism and recreation result from increased visitor numbers and changes in visitor behavior which in turn leads to higher revenues, increased jobs and other livelihood opportunities. Literature suggest that MPAs can have a direct and indirect effect on this and that MPAs can be more profitable because of these benefits (EC 2018).

MPAs can for example attract visitors. In New Zealand the establishment of the first MPA attracted the public and schools. It was estimated that the MPA attracted approximately 5000 visitors per year. People near the area were well aware of the regulations in the area and also paid attention to make sure that others were not breaking the rules. These effects have been demonstrated in a varying degree near other MPAs as well depending on accessibility (Ballantine 2014).

Another study on the Great Barrier Reef showed that this area has many positive effects on tourism. Annually, this MPA generates approximately 3.2 billion euro, with the majority of revenues provided by tourism. This estimate includes only use values and is likely an underestimation of the total value. The Great Barrier Reef also contributes to approximately 53,800 full time jobs, most of which are related to tourism. Overall revenue from tourism has been increasing steadily in the years it was studied (McCook et al. 2010).

In the UK the recreational benefits of an MPA around the island of Lundy have been studied using the Travel-Cost method. The Travel Cost Method is a revealed preference method and relies on actual market data and human behaviour to reveal peoples' preferences for MPAs. The method estimates the expenditures incurred by households or individuals (e.g. money and time) to reach a specific MPA, in this case, Lundy Island. Data was obtained from surveys on site. Questions were designed to obtain information on the tendency of visitors to go to coastal areas, how long they would stay etc. There were also questions related to the knowledge of visitors on the protected area, the trips visitors would make and demographic questions (e.g. year of birth, education level, income). This resulted in 86 responses that were used in the final economic analyses. Using regression models, it was estimated that visitors were willing to pay £359 to £574 to visit the island. The results also suggested that the no-take zone had significantly contributed to this amount (Chae et al. 2012).

Another study in the UK in Lyme Bay also assessed whether the MPA had effects on recreation in the area. This study used spatial analyses as well as quantitative and qualitative survey methods for a period of 3 years, post-designation. Stakeholder groups included charter boat operators, dive businesses, divers and anglers. Within the 3 years, dive businesses had increased activities within and outside the MPA. Activity of charter boat operators have decreased outside the MPA and increased within the MPA. Dive businesses have seen an overall increase in activity both outside and within the MPA. However, they perceived little or no effect on their businesses. Divers have also increased their activity outside and within the MPA and reported that the MPA has effects on the diving location. Sea angling activity decreased outside the MPA and increased within the MPA. Angling expenditures increased with 1.5 million, diving with 0.5 million. Because of the increase in activities, expenditures of charter boat operators increased with 108,427 and expenditures of dive businesses increased with 39,864. Together, the potential increase in value of this MPA because of these activities is estimated at £2.2 million within the 3 years studied (Rees et al. 2015).

MPAs in southern Europe have also shown that they can be profitable, especially because of their impact on tourism. Using surveys and empirical data, it was estimated that MPAs in Southern Europe, on average generate 13 jobs in scuba diving and 2.1 jobs in recreational fishing. The added value derived from recreational expenditures was estimated to be €88,000 per year for recreational fishing, and €551,000 for scuba diving. These benefits together generate €639,000 per year, thereby outweighing the management costs of €588,000. The benefit estimates can be considered conservative because only non-residential recreational (e.g. tourists, people living further away) were taking into account. Also, non-market benefits and indirect effects were not taking into account. However, the problem is that it is generally very difficult to distinguish between the MPA effect and the site effect. Therefore, a qualitative survey approach was used to assess the users' perceptions. The study suggested that the MPA effect was clearer for scuba-diving than for fishing (Roncin et al. 2008).

In Scotland the effects of several MPAs on tourism were also assessed via interviews with key informants. These informants were engaged in the process of MPA designation and helped developing the management plans. The informants stated that marine related tourism had not changed since the introduction of MPA measures. However, the interviews took place within less than one year after designation and the informants stated that it was too early to see changes. They pointed out that there are a number of plans and projects that would be implemented in the next year (Marine Scotland Science 2016).

A difficulty in assessing the benefits for tourism is the fact that there is often no baseline, which makes it difficult to distinguish between the site effect and the MPA effect. Collecting data before MPA establishment could support research and help quantifying the effects (Roncin et al. 2008). Research so far suggest that MPAs can have positive effects on recreation and tourism. This view is supported by the fact that stakeholders hardly ever report negative effects of MPAs on recreation and tourism (EC 2017). Inshore MPAs are more easily accessible and therefore visited more often than offshore MPAs. The benefits for recreation and tourism will therefore be greater for these sites. However, it generally takes time before they occur (The Benyon Review Panel 2020). Recreational activities and tourism can in turn have adverse effects on the environment. For example, divers may damage coral reefs with their fins or they may grasp hold on kelps when currents are strong. Fish feeding may alter behavior and watching marine animals may cause stress. Boats may also cause environmental change if they anchor on corals or seagrass beds. It is therefore important to keep the number of visitors within the carrying capacity of the area (Alban et al. 2006). When kept within boundaries, recreation and tourism are a likely benefit for coastal communities. The possibility of benefits from tourism and recreation can generate increased support towards MPAs and facilitate implementation.

4.2 Estimating benefits in socio-economic analyses

4.2.1 Bio-economic modelling

Bio-economic modelling is a tool which is often used to estimate effects of MPAs on fisheries. There are different types of bio-economic models. Spatially non-explicit bio-economic models are usually based on one species, one gear type and an unchanging environment. The purpose of these models is to assess how fishing is impacted by closing areas. Spill-over effects are modelled based on the difference of fish stocks inside versus outside the MPA and the mobility of the investigated species. Outcomes of these models usually suggest that spill-over effects depend on a variety of factors, such as fish mobility, MPA size, protection levels etc. The major disadvantage of these models is that assumptions are often oversimplified which may lead to inappropriate conclusions about MPA effects (Alban et al. 2006).

There are also spatially explicit models that take into account multiple subpopulations which are distributed in different patches. These subpopulations are connected via biological and economic relations. These models are usually based on multiple species and the focus is often on the location of the MPA. Location characteristics that can be included are oceanographic processes, ecological features of habitats, larval dispersion and socio-economic factors. These models thus require a certain level of knowledge before they can be applied (Alban et al. 2006).

Such a spatially explicit bio-economic model was used to assess the effects of MPAs on demersal fishing in the strait of Sicily. This area is among the most productive fishing areas in the Mediterranean Sea. Three commercial species (deep water rose shrimp, European hake and red mullet) were taken into account which account 70% of total revenue of demersal fish resources in this area. The model uses information from trawl surveys, commercial catch data and vessel activity data provided by Vessel Monitoring Systems for the years 2006-2010. The model is named “SMART” (spatial management of demersal resources for trawl fisheries).

The model is based on the assumption that these demersal species are affected by sea bottom characteristics and sea temperature which in turn affect benthic communities. Furthermore, resource abundance in space and time is influenced by the combined effect of demography (e.g. reproduction, mortality etc.) and fishing effort is an external source of size dependent mortality. Finally, in mixed fisheries, fishermen independently exploit different resources because economic factors such as market prices and costs may change over time.

The final model integrates the following tools:

- a spatial analyses that models the distribution of the species, fishing activities and abiotic factors. Results are used to create geographical data of this area for the years 2006-2010.
- An artificial neural network is then used to capture the relationships between distribution, fishing activity and abiotic factors to predict species abundance in the future.
- Classic fishery science equations are used in a model to predict the size structure of catches which are then converted into revenues based on market prices. Costs are modelled based on fishing effort and fuel prize. Revenues can then be used to calculate gains.
- Simulations are used in the final step in order to assess effects of fishing effort caused by different management scenarios on species abundance in the future and predict bio-economic effects.

MPA effects are estimated for four scenarios, in the first 3 scenarios, the nursery area of the three species are closed for bottom trawling separately while in the other scenario the nursery areas of all three species are closed for bottom trawling.

Results of this model suggest that closing all three areas result in the most benefits for all three species but it also includes major costs. This is caused by the fact that the nursery areas and thus the MPAs are located nearshore. Fishermen therefore have to fish further offshore which increases the costs. However, over successive years the species keep recovering while costs remain almost constant. Lower profits could therefore be partially compensated. On the other hand closing a single specific nursing areas is beneficial for that particular species but it can have detrimental effects on the other species, unless there is a large overlap in distribution between the protected species and the other species (Russo et al. 2014).

Bio-economic models can also predict possible outcomes for recreation and tourisms on top of fishery effects. Here, the economic effects of a three-zone MPA including a no-take zone, a partially protected zone and a non-protected area. The partially protected area acts as a buffer zone for the no-take zone. In the partially protected area recreational and commercial fishing is only allowed till certain extent. Spill-over effects of the no-take zone to the partially protected area and to the non-protected area is modelled using a density dependent function. Fishing mortality is dependent on fishing effort. Fishing revenues are estimated based on sale of catches of each species by different fleets at a certain price. Revenues from tourism are modelled with a function that includes protected area size, the effort in attracting tourists and fishing effort. Here it is assumed that fishing affects fish diversity and therefore makes the area less attractive for touristic activities such as scuba-diving, mammal watching and sailing. Empirical data used in this study comes from the Medes Islands (Spain) where a three-zone MPA has been established.

The results show that increased fishing intensity affects income from fisheries positively as a result of current under exploitation while tourism is affected negatively. The model estimates the revenues from touristic activities (scuba diving, glass-bottom boats and other activities) to be 5.9 million. In contrast to fishing which generates 0.2 million. When the complete area is used for fishing only, the revenues increase to 0.5 million. The potential for fisheries is thus significantly lower than current revenues generated by tourism (Merino et al. 2009). Although this study is not exactly an ex ante analysis, it provides insides in some of the benefits that can be expected as a result of MPAs in a system that allows multiple activities in different zones.

Bio-economic models can be used for analysis of policy scenarios and to understand the relations between natural resources and human welfare. The major benefits include a better understanding of the feedback mechanisms between human impacts and natural resource dynamics. Bio-economic models use mathematical functions to describe both economic and biological systems and the way they are linked to each other. This is generally very challenging because it includes variability in biological systems and human behaviour. Data collection on ecosystem dynamics, interactions and feedback mechanisms is required in bio-economic models, the actual availability of data determines what can be modelled in practice (Prellezo et al. 2012). Assumptions are necessary when insufficient data is available, which makes the results less reliable. A general point of critique is that bio-economic are too simplistic. In the case of MPAs it is argued that important ecosystem components such as predator prey interactions, or the effects of improved habitats are ignored in these analyses. On the other hand it is suggested that adding too many features will create a black box in which underlying relations are not clear. This makes the model less useful (Armstrong 2007).

In conclusion, bio-economic models in the marine context have been focusing mostly on effects on fish stocks and fisheries and in some cases other activities have been taken into account as well. Bio-economic models can be useful in estimating some of the benefits of MPAs when enough data is available. When a lot of assumptions are made the reliability of the results can be questioned. Moreover, one can question the reliability of sometimes relatively simple models in complex marine ecosystems with many interactions.

4.2.2 Benefit transfer method

Benefit transfer methods use values obtained in one context to estimate the value in another context. This approach can be especially useful in cases where there are gaps in evidence and resources are not sufficient to perform a site-specific primary valuation study (Hussain et al. 2010). Estimates are taken from a case study site, which has been studied to some extent. The estimates are then applied to a so-called policy site, a site which will be subjected to a proposed policy action. In environmental applications, this approach focuses on the change in biological and physical characteristics of an area caused by the proposed policy (Plummer 2009). Spatial data is often used to quantify the ecosystem services. This approach is known as ecosystem service mapping and works the following way. First, the area is classified in biomes, land cover or other ecological landscape types. Then, ecosystem services are linked to the landscape types. After that, original valuation data can be collected for these landscape types and its ecosystem services. The next step is to take the estimate of the value of these ecosystem services and divide this by the landscape type area, creating a value of that ecosystem service per landscape type area unit (e.g. carbon sequestration per km2 seagrass). This value per unit area can then be calculated for the landscape types in the proposed area. The total value of ecosystem services provided by the area is the sum of the values calculated for the ecosystem services provided by the landscape types (Plummer 2009).

A benefit transfer approach was used to predict the benefits of a proposed MPA network in the United Kingdom. Prior to this study, 3 possible network options were selected based on OSPAR criteria and other criteria. There were also 2 possible management regimes (more restricted and less restricted).

All in all, the benefits of 6 policy scenarios (network scenarios and management regimes) relative to the baseline scenario were estimated over a period of 20 years. In total, 11 possible ecosystem services were considered. However, data could only be found for 7 of these ecosystem services. The ecosystem services that were analyzed are: food provisioning, raw materials, nutrient cycling, gas and climate regulation, disturbance prevention and alleviation, cognitive values and leisure and recreation. For these 7 ecosystem services, only aggregate values were available instead of values for specific marine landscapes.

Once the ecosystem services were determined the marine landscapes that would possible be protected were identified and linked to corresponding ecosystem services. The classification of marine landscapes was based on geophysical attributes such as depth, seabed sediments and other data. This resulted in 26 marine landscape types and 9 habitats with corresponding ecosystem services provided by them according to literature.

After that, the impact of the possible management regimes (less and more restricted) was scored for each ecosystem service per hectare of landscape type. This was based on the expected difference between the status quo and the possible policy options for the next 20 years. These estimates were performed by a group of ecologist that were crosschecked afterwards. Expected time before changes would be observed as well as the expected response curve (linear, logarithmic or exponential) were taken into account.

Then, benefit estimates were apportioned to individual landscape types/habitats. Part of this was to estimate how important a particular landscape was for delivering a specific ecosystem service. This assessment was based on the surface area of a particular landscape as well as possible impact of that particular area for that ecosystem service where possible. This resulted in a percentage of the importance of particular landscapes for each of the ecosystem services under consideration. With the monetary values obtained in literature and the calculated percentages, the monetary value of each of the landscapes could be determined per hectare.

Combining the expected change in ecosystem services (in %) of different landscapes with the monetary value obtained for that particular landscape and ecosystem service resulted in the present benefit of one a particular ecosystem service for a particular landscape type. After that, this value could be multiplied with the total surface area of that landscape and repeated for every combination of landscape type and ecosystem service to obtain the total current value under different MPA networks and management regimes.

The total present value ranges between £10.2 and the £23.5 billion when applying a 3.5% discount rate (Hussain et al. 2010). The costs were estimated in a different study and ranged between £0.4 and £1.2 billion. When the lowest benefit estimate with the highest cost estimate is applied the benefit to cost ratio ranges from 5.5 to 12.7 depending on the policy scenarios, suggesting that the benefits outweigh the costs by far (Hussain et al. 2010).

A similar approach has been used in Scotland to assess the possible benefits of three theoretical MPA networks and two levels of management regimes. The approach is quite similar to the previous one, as it assigns ecosystem services to certain landscape types. Future impacts of protection were also estimated based on expert judgement. The main difference between the previous study is that estimates for direct use, indirect use and non-use values were taken into account contrary to direct use and indirect use values in the previous example (Álvarez García et al. 2012). These estimates were adjusted from a choice experiment that derived the value people were willing to pay for halting biodiversity loss in the United Kingdom (McVittie and Moran 2010).

Results showed that the benefits of designating a Scottish MPA network ranged between £6.3 billion and £10 billion depending on the different network designs and management scenarios. The results also suggested that the benefits were not very much affected by management regimes, contrary to scientific results that demonstrate that more strict MPAs are more effective. It was suggested that this result is likely to be caused by the fact that both management regimes restrict bottom-towed fishing gear. However, it was acknowledged that there remain large uncertainties about the possible effects of MPAs. The non-use values in this analyses account for 12-14% of the overall benefits. The study mentions a few shortcomings. First, several ecosystem services could not be taken into account due to a lack of data. Second, the offsite benefits (spill-over effects) were not taken into account. Finally the possible network effects could also not be taken into account because of this lack of data, so networks were treated as series of individual MPAs. Therefore, it was suggested that the values obtained represent the minimum values of the possible MPA networks and the actual values are likely to be higher (Álvarez García et al. 2012).

Benefit transfers use the value obtained in one site to estimate value in another site (Plummer 2009). Benefit transfers are especially useful when gathering site specific data is too costly and/or takes too much time. It also allows for including a variety of ecosystem services (Hussain et al. 2010). This relatively simple approach is defensible provided that sites, goods and context are highly similar (Bateman et al. 2011). The degree of similarity between the study site and the policy site is called "correspondence". The dissimilarity between study site and the policy site may lead to what is called "generalization". This means that sites are mistakenly treated as equivalent in all characteristics based on a few similar characteristics. Such errors imply that benefit transfers will never be successful. However, assessing the magnitude of errors could potentially allow for evaluation of the benefit transfer and the tradeoffs between its lower costs and the shortcomings (Plummer 2009). In order to do that primary valuation data needs to be compared to benefit transfers. However, the links between marine ecosystems and the services they provide are often not clear (Townsend et al. 2018). This is reflected by a small amount of primary valuation studies (Torres and Hanley 2016). In theory, this method is relatively easy and allows for including many ecosystem services. However, gathering more information on the links between ecosystems and the services they provide is needed before it can be applied more widely.

4.2.3 Stated preference methods

Stated preference methods can be used to measure non-market benefits. These methods are based on surveys. The main advantage of stated preference methods is that they be used to obtain use and non-use values associated with a particular good. Non-use values are bequest values (i.e. the value on conserving resource for future generations) and existence values (i.e. value placed on knowing that the resource will continue to exist) (Jobstvogt et al. 2014).

Stated preference methods can be categorized into contingent valuation and discrete choice experiments. Contingent valuation is based on creating a hypothetical market (Halkos and Galani 2012). The economic value is derived from information related to preferences which is obtained via surveys. A discrete choice experiment, on the other hand, uses multiple options which differ in their outcomes and their costs. Participants are then asked to select their preferred choice. From these responses, the valuations of the outcomes can be determined statistically (Davis et al. 2019).

In Norway a discrete choice experiment was used to value cold-water corals. Most of these cold-water corals are discovered recently which poses challenges for fisheries, coastal management and deep-sea resources. In order to estimate people’s willingness to pay for increased protection of the cold-water corals a discrete choice experiment was performed. Across the whole country 397 people were surveyed. Together, these participants provided 4683 choice observations.

The discrete choice experiment revealed that protecting a larger area than the current area was chosen in 75% of the choices. Annually, participants were willing to pay 166 euro more if the area included important habitats for fish and participants were willing to pay 39 euro more if the area is important for fisheries and 16 euro if the area is important for the oil industry. Finally, people were willing to pay 53 euro for an extension of the MPA from 2445 km2 to 5000 km2 and 66 euro if the area is extended to 10000 km2 (Aanesen et al. 2015).

A combination of a discrete choice experiment and contingent valuation was applied in the UK to elicit divers’ and anglers’ willingness to pay for potential MPAs. The willingness to pay for visiting diving and angling sites (use values) was determined with a travel cost based choice experiment in which participants could select 5 sites which they would like to visit based on distance and features within these sites. This is a more realistic approach than selecting 1 site because participants were able to visit a site with certain characteristics further away once and then visit sites closer by more often for example. Participants also had a stay at home option to make it even more realistic.

In order to elicit the non-use and option values a contingent valuation approach was used. Participants were asked how much they were willing to donate to protect a hypothetical dive/angling site against future degradation. Sensitivity of participants payments were explored by using different payment scales, the first one ranging from £0–£20, the second one ranging from £0–£40. Participants were also able to state that they were willing to pay more than the top of the scale.

In total, 1332 participants completed the survey. The vast majority (76%) of them were divers, the rest (24%) were anglers. Willingness to pay in travel cost instead of staying home regardless of site attributes was £7.52 for divers and £20.78 for anglers. The protection of species of conservation interest increased willingness to pay. Divers were willing to pay £0.44 more per species and anglers were willing to pay £0.30 more. MPA size did not affect willingness to pay of divers and decreased willingness to pay of anglers with £0.79 per 10 times increase in size.

Based on the contingent valuation approach it was concluded that divers were willing to pay more than anglers (8.83 vs 8.29). The range of donation possibilities also influenced willingness to pay. In the lower range (£0-£20) participants were willing to pay £7.89 compared to £9.55 in the higher range (£0-£40). In general, participants were willing to donate money irrespective of site characteristics. It was concluded that it is crucial to take the use and non-use values of cultural ecosystem services into account as this would make a stronger case for protection than conservation of biodiversity alone (Jobstvogt et al. 2014).

Although stated preference methods can be helpful in estimating the value of non-market benefits there has been well-documented criticism of the approach. There are differences in the approach used in surveys. Willingness to pay refers to the amount of money people are willing to spent for a proposed welfare gain while willingness to accept elicits the amount of money people are willing to accept for the loss of a certain good. Generally, the amount of money people are willing to accept for the loss of good is greater than the willingness to pay for improvement. This is partially explained by the fact that willingness to pay is affected by income, while willingness to accept is not. Another reason for higher willingness to accept is the so called substitution effect which simply means that the willingness to accept increases when there are few or no substitutes of a particular good (Venkatachalam 2004).

Another issue concerns the scope of the proposed measure. It has been argued that willingness to pay is insensitive for the scope of a certain good which highlights the non-existence of individual preferences for public goods (Stolwijk 2004). On the other hand, advocates of this approach state that the scope effect is mainly caused by incorrect survey approaches. Making clear what the differences are, for example by using maps, showing pictures and let people revise the bids are solutions proposed to minimize this scope effect (Venkatachalam 2004). Furthermore, stated preference methods are based on what participants say they would do, not on what they actually do, which is a source of uncertainty (Liefveld et al. 2011). Hypothetical willingness to pay is indeed often higher than actual willingness to pay. The participants unfamiliarity with the particular goods is a major cause for this. Making people more familiar with the good in question could decrease the bias between the hypothetical willingness to pay and the actual willingness to pay (Venkatachalam 2004).

This unfamiliarity makes it application in the marine context especially challenging as the subject is simply too distant for many people. Biodiversity is lost without people noticing it because it happens under water and/or at large distances. It is thus questionable whether results from these approaches can be reliable in the marine context (Liefveld et al. 2011). However, there are simply no real markets for all ecosystem services provided by MPAs while these ecosystem services should not be neglected in decision-making processes. Application of stated preference methods could potentially help in eliciting the value of these services. When applied properly, uncertainties may be diminished (Venkatachalam 2004). Whether or not any number is better than no number depends on decision-makers willingness to accept its limitations.

4.2.4 Multi criteria analyses

Multi criteria analysis (MCA) is a tool that can support decision-making by assessing the pros and cons of multiple alternatives. It allows for comparison of alternatives against criteria that represent the most relevant aspects in a given decision making process. In practice, MCAs support structuring of the decision problem, the assessment of different options across criteria, exploring trade-offs of these options, formulating a decision and test its robustness. MCAs can be very useful when a multi-objective problem cannot be reduced to a single-objective problem. For example when criteria are expressed in different units (e.g. monetary values, biophysical units, qualitative evaluations etc.) MCAs allow combining the analytical performance of different alternatives with the priorities and preferences of stakeholder in transparent and replicable way (Esmail and Geneletti 2018).

MCAs can also be integrated with geographical tools, which has been done to assess the current MPA network in Finland and explore how this network could best be expanded. The spatial program used is called Zonation which produces a hierarchical prioritization across the landscape by taking into account habitats, ecosystem services, area connectivity and costs and threats. These factors can also be weighted according to their importance in different situations.

The features that were used in the analysis are the habitats under the EU Habitats Directive, ecosystems in the IUCN Red list of Ecosystems, fish reproduction areas, key species, IUCN Red list species, invasive species and marine pressures. Invasive species and pressures were given a negative weight while features important for conservation were given positive weights. This was done so that important features would be included while areas in which there are threats would be avoided by the program. Connectivity was also taken into account which was done by letting the program identify similar habitats within a specific range.

Information on species came from an existing database with data of approximately 140,000 locations. For other areas species distribution models were used. These models use environmental data and species occurrence in studied areas to predict occurrence in other areas. Other environmental data on habitats, pressures was already available. In the case of pressures a buffer zone was applied around these areas.

Information on all these features was put into the program Zonation. Based on the spatial data it was concluded that the current MPA network is performing poorly as only 27% of the most important ecological features is protected. Increasing the current coverage of the MPA network from 10 to 11% could potentially protect 60% of the ecological features instead of the current 27%.

This tool in particular could be very useful in identifying areas that are most interesting to protect. It could also be used for impact avoidance as it also reveals the least interesting areas for conservation (Virtanen et al. 2018). A major disadvantage is that such an extensive analyses requires a lot of information on important features such as species occurrence, habitats etc. Another disadvantage is that it does not provide information about effects and possible future benefits of MPA designation.

MCAs can also include stakeholder perceptions which can elicit the preferences and the expectations towards MPAs prior to establishment. This has been done in Canada prior to the establishment of an MPA in order to assess what effects would be desired by different stakeholders groups. The stakeholders that were taken into account are: tourism operators, boaters, recreational fishers, commercial fishers, NGOs, local government and the national government. Stakeholders were asked to rank performance criteria in a pairwise comparison. These pairwise comparisons were made between and within criteria categories which could be related to environmental effects, social effects, economic effects and management effects. For example, less pollution versus more fish etc. The basic idea is that participants score how important they find a certain effect compared to another effect.

From this analysis, it was concluded that environmental effects were considered to be more important than social, economic and management effects according to all stakeholder groups. However, how much more important this was differed between stakeholders groups. Tourism operators, boaters and local and commercial fishers put lower weights on environmental effects than NGOs, local government and national government. These stakeholders, on the other hand, put lower weights on economic effects compared to the other stakeholders. Tools like this help in the process of MPA designation as it makes clear what performance criteria are most important according to different stakeholder groups. This in turn can help in the development of the MPA by setting clear goals and objectives (Heck et al. 2011).

MCA is a tool which can support decision-making by assessing and comparing the pros and cons of different options. MCAs can be particularly useful when a multi objective problem cannot be reduced to a single objective problem. It allows for combining analytical performances of alternatives with the preferences of stakeholders in a transparent way (Adem Esmail and Geneletti 2018). The combination with geographical tools allows for strategic siting of MPAs. Environmental factors and human factors are taken into account which enhances cost-effectiveness. It is suggested that such geographical tools are used more often (OECD 2015). MCAs also acknowledge that it is difficult to produce a single right answer in decision-making processes involving intangible objects such as biodiversity and economic gains. Moreover, in MCAs, stakeholders are often involved throughout the process. In that sense, developing an MCA can be described as a joint problem solving process in which stakeholders learn more about the various aspects of the problem and may therefore find new solutions (Saarikoski et al. 2016). The downside of using multi criteria analysis to estimate benefits is that the value is expressed in non-monetary terms and does not reflect human welfare (Bos and Ruijs 2019). Another problem is that aggregation of scores in multi criteria is more difficult compared to analyses that use monetary valuation only (Saarikoski et al. 2016). Finally, a common pitfall is the use of excessive and unbalanced amounts of criteria which should be avoided, keeping its initial purpose in mind is very important (Esmail and Geneletti 2018).

MCAs are thus especially useful when multiple criteria are important for decision-making and the effects cannot be expressed as a single denominator. The risk of applying MCAs is that too many criteria are taken into account which makes it more difficult to come to conclusions.

4.2.5 *Eco point valuation method*

In the Netherlands it was concluded that it is generally very difficult to assess the effects of measures on nature, which makes it very difficult to perform cost benefit analyses. Therefore, a standardized approach was developed to quantify the effects of measures on nature (Sijtsma 2009). This method, which is called the eco point valuation method, can be used to calculate ecological values before and after the implementation of measures. It also allows for comparison of different scenarios. Eco points can be calculated by using a formula that takes into account the size of the habitats, the quality of the habitats and a weighing factor (figure 3).

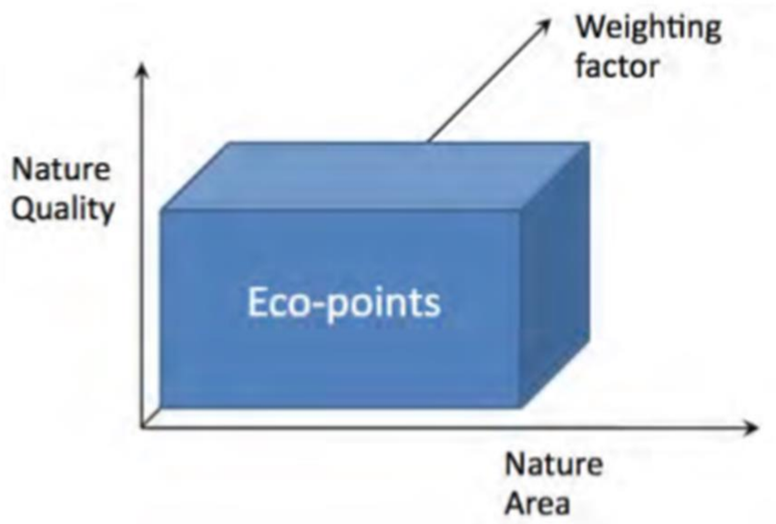


Figure 3. Schematic overview of the eco point valuation method (Liefveld et al. 2011)

The total number of eco points for a given scenario can be calculated using the following formula:

Eco point total = ∑all habitats(Area * Quality* Weighting factor)per habitat

In the equation, the area refers to the size of specific habitats. The quality refers to the presence of specific species. The idea behind this is that the presence of specific species in an area tells something about the quality of that area. When areas are of similar size and similar quality this would result in the same value for the areas even though the areas can be very different from each other. Therefore, a weighing factor is applied, which is used to determine the contribution of specific areas to a whole array of biodiversity on national, European and global level. For example, if rare or threatened species occurs in an area, then the loss of this area is more severe than when there are only common species within an area. In this way, the weighing factor offers the opportunity to include the importance of specific areas as well. A prerequisite for this approach is that the indicators are based on stakeholder consensus in order to come to a joint decision (Sijtsma et al. 2009). This approach which was initially developed for terrestrial and freshwater systems has now been used a few times for marine systems as well.

An adapted approach has been used to determine and compare effects of different MPA scenarios for the Central Oysterground and Frisian Front, two areas in the Dutch part of the North Sea. In this adapted cost-benefit analyses, 6 scenarios were compared based on their socio-economic effects and their effects on biodiversity using the eco point method. Quantity was determined based on the size of the MPAs in each scenario in km2. Quality was calculated based on benthic biodiversity maps. Multiple weighing factors were applied that represented various ecological points of view. The weighing factors referred to the potential improvement of specific habitats, protection of rare habitats, the protection of all species groups, and favoured the protection of larger areas over many small ones (van Oostenbrugge et al. 2015).

Using this approach, the quality of the variants could be compared to each other. For each variant the potential costs and benefits in terms of eco points were calculated and compared to each other to make a decision. In this analyses only current ecological value was taken into account as it was acknowledged that there are large knowledge gaps. Therefore, future benefits were not analysed. The authors recognize this and stated that there could be more ecological benefits in the future and it would be better if these could be taken into account as well (van Oostenbrugge et al. 2015).

The eco point valuation method has also been applied to assess the future gain in ecological quality as a result proposed measures. This has been applied to assess the effects of adding hard substrate to the Dutch part of the North Sea.

Current eco points of habitats in the North Sea are calculated based on surface area, habitat quality. The weighing factor is based on habitat fidelity (i.e. habitat contains species that occur in that habitat only) (Liefveld et al. 2011).

Habitat quality is based on indicators for benthos, fish, birds and marine mammals. For benthos indicators such as density, biomass, rarity, large species and species richness were used. For fish indicators such as rarity, large individuals within species, large species and species richness were used to determine quality. For birds quality of a particular habitat is based on a single metric called bird value which is based on a combination of ship-based surveys and aerial surveys all year round. For marine mammals, density was used as an indicator (Bos et al. 2011). Maximum habitat quality is based on the maximum scores for these indicators in the current situation and set to 100%. All values were scaled accordingly.

Two possible scenarios were compared to the status quo. The difference between them is that hard substrate is added to different habitat types to compare their possible effects. In the first scenario hard substrate was added to a habitat type characterized by a soft bottom, while the other habitat is characterized by harder gravel. Impact of the measure (adding hard substrate) was based on expert judgement and available literature. The main impacts are that hard substrate become habitat elements and fisheries are obstructed because of it.

Based on expert judgement and literature the effects were expected to have the largest effects on benthos and to a lesser extent on fish. Effects on marine mammals and seabirds were considered negligible. Quality scores after implementation were averaged for benthos, fish, seabirds and marine mammals and compared to the current quality score of the investigated areas.

Eco points were calculated for the pre and post measure scenarios based on surface area of the measure, habitat quality and habitat fidelity as weighing factor. The difference between the pre and post situations were then used to determine the gain in eco points. Based on the results it was concluded that adding hard substrate has similar effects in terms of quality increase for both habitat types. However, the second habitat characterized by gravel has a higher score for site fidelity which resulted in an greater increase in eco points between the pre and post measure situation (Liefveld et al. 2011). Although this example does not describes its application in the context of MPAs, the measure in question can be replaced by any other measure associated with MPAs.

The main advantage of the eco point valuation method is that the intrinsic value of nature can be included in CBAs in a clear repeatable way which allows for comparison of different options (Liefveld et al. 2011). It also presents a single objective measure of biodiversity which can be more clear and more easily understood than presenting a whole range of impacts (Bos and Ruijs 2019). Moreover, by applying weighing factors certain MPA characteristics, such as connectivity or favouring one large area over many small ones, can be favoured (van Oostenbrugge et al. 2015). The most important disadvantage is that eco points do not reflect a change in human welfare (De Blaeij and Verburg 2011). Eco points are not expressed in monetary terms. Therefore, they do not influence the net benefits in cost-benefit analyses which implies that they may be neglected in decision-making processes. However, the goal of MPAs is the protection of biodiversity and recovery of marine ecosystems. It is also generally understood that biodiversity contributes to the generation of other ecosystem services (Townsend et al. 2018). Therefore, eco points can be very useful as an additional tool in order to quantify and represent biodiversity in CBAs (Bos and Ruijs 2019).

The eco point valuation method has been developed for projects that have an impact on nature. The intrinsic value of biodiversity is quantified in a clear repeatable manner and allows for area comparison as well as the analysis of future impacts. This suits the overall goals of MPAs which aim to protect and conserve biodiversity. Eco points could therefore be very useful in the context of MPAs.

4.3 Concluding remarks

MPAs are a tool used for the protection and recovery of marine ecosystems. MPAs provide many benefits for both nature and humans. Benefits of MPAs include: ecological benefits, carbon storage and sequestration, future benefits for fisheries and benefits for tourism. However, these benefits have in common that they vary per site which makes it difficult to predict the extent of effects that can be expected.

Multiple methods have been used to quantify or monetize the benefits of MPAs, each having its own strength and weaknesses. The overarching goal of MPAs is the protection of biodiversity and the recovery of marine ecosystems. The exact links between biodiversity and the ecosystem services are not well understood. However, it is generally agreed that biodiversity contributes to the generation of ecosystem services. The eco point valuation method is a method in which the intrinsic value of biodiversity is quantified in a clear repeatable manner which is why this method is recommended as an additional tool in CBAs. Therefore, an attempt will be made to apply this method on a possible new MPA in the Netherlands: the Borkum Reef Ground. This will be discussed in the next chapter.

5 Case study the Borkum Reef Ground

The North Sea is among the most heavily used marine areas in the world. It contains vulnerable habitats and it is an important source of food for both humans and animals. Major shipping lines go through the North Sea, recreational activities take place and the area is used as for practice operations of the navy and the air force. The area is also used to harvest wind energy, sand, oil and gas (Statistics Netherlands 2016).

The future of the Dutch part of the North Sea and the activities that may take place there are currently under debate. The Netherlands is for example moving towards more sustainable energy practices. Therefore, in the next decades, many and large offshore wind farms will be developed in the Dutch part of the North Sea. Furthermore, there are plans to make fishing practices more sustainable. In addition, some areas will be closed for bottom-towed fishing gear in order to protect the sea bed and overall biodiversity (Ministry of Infrastructure and the Environment & Ministry of Economic Affairs 2012). The latter has declined in the last century, mainly due to the removal of hard substrate, such as stones and biogenic reefs, by fisheries (Didderen et al. 2018).

In order to restore biodiversity and protect the marine ecosystems a target was set at protecting 10-15% of the seabed in the North Sea against human activities (Ministry of Infrastructure and the Environment & Ministry of Economic Affairs 2012). Therefore, additional areas are under consideration for MPA designation. One of these additional areas is called the Borkum Reef Ground which may be protected against bottom towed fishing gear. This is especially important in the light of the latest conservation goals within the EU that aim to protect at least 30% of the marine environment within MPAs (EC 2020a). Additional MPAs are thus needed in order to achieve conservation goals.

The aim of this case study is to describe the possible ecological benefits of protection against bottom towed fishing gear in the Borkum Reef Ground. It starts with a description of the area and the ecological features within this area. This part also briefly discusses how the Borkum Reef Ground contributes to the establishment of an ecologically coherent network of MPA. After that the human activities in this area are discussed as well as their impact on ecological features. Finally, information on the ecological features and the human impacts are combined in order to estimate the possible future ecological benefits that may arise through protection against bottom towed fishing gear.

5.1 Area description

The Borkum Reef Ground is an area located 25km north of the island of Schiermonnikoog. Water depth ranges between 10 and 40m, temperature varies between 3 and 19 °C and maximum currents vary between 0.4 and 1.0 m/s (Coolen 2017).

The area is considered an ecologically interesting area in the Netherlands because unlike many other areas, the bottom still contains hard substrate such as rocks which were left behind after the second last ice age >126,000 years ago. While some of these rocks have been removed by fishers, part of them remained. These rocks support a variety of biodiversity (van Duren et al. 2016).

Furthermore, dense aggregations of sand mason worms are present within this area. This tube-dwelling annelid modifies its habitat by creating a heterogeneous habitat which creates attachment surfaces for other species. Enhanced benthos density may also attract demersal predators such as plaice and sole (Rabaut et al. 2008). This way, the sand mason worm has a considerable positive effect on the local biodiversity (Coolen et al. 2015).

The total surface area of the Borkum Reef Ground is 554 km2. The surface area of rocky reefs in the Borkum Reef Ground were estimated at 9.8 km2 (1,6%) and the surface area of the reefs build by sand mason worms were estimated to be around 74 km2 (12%) (Coolen 2017).

Designating the Borkum Reef Ground as MPA contributes to the ecological coherence of the Dutch and European MPA network in multiple ways. First of all this area is connected to the North Sea Coastal zone (Noordzeekustzone) protected under the Habitats Directive (figure 4). Second, this area is connected to the German Borkum Reef Ground protected under the Habitat Directive (Federal Agency for Nature Conservation n.d.). This creates a single large protected area that covers a variety of habitats. Finally, this area contributes to the protection of rocky reefs in multiple MPAs as this habitat type is protected in the Cleaver Bank (Klaverbank) as well. Designating the Borkum Reef Ground as MPA would contribute to the representation of this habitat in multiple MPAs, which is used as a proxy for resilience (see chapter 3.5).

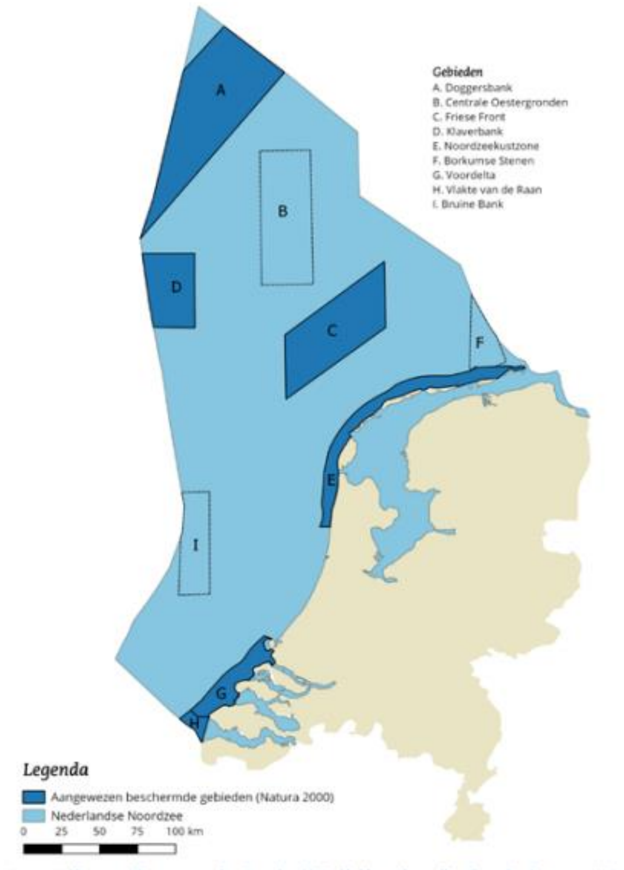


Figure 4. Overview of MPAs within the Netherlands (in blue) and the areas that may qualify for protection. Area F is the Borkum Reef Ground (Vrooman et al. 2018)

The Borkum Reef Ground plays and important role in the North Sea for species associated with hard substrate. It is one of the last areas in the Dutch part of the North Sea where reefs are still present (Didderen et al. 2018). Protecting this area is therefore of great importance for biodiversity conservation and would make a great contribution to the MPA network in the Netherlands.

5.2 Current activities in the Borkum Reef Ground and their environmental impacts

5.2.1 Fishing

In 2017 there were 103 fishing vessels active within this area. The fishing gear used in the Borkum Reef Ground are: otter trawls and otter twin trawls, Scottish seines, beam trawls and shrimp trawls. Catches per gear type and value of those catches is presented in table 1. Other gear types such as pelagic trawls are not used in the Borkum Reef Ground.

Table 1 Overview gear types used by the Dutch fleet in the Borkum Reef Ground and their catches in 2017 (WEcR).

Gear type used	**Total catches per year (KG)**	**Value of catches (€)**
Otter twin trawls/otter trawls	68	118
Scottish Seines	997	1.948
Beamtrawls/sumwings/ pulswings	7	16
Shrimp trawls	124.374	813.773

The results show that shrimp trawls represent the vast majority of gear type used within the Borkum Reef Ground. In the North Sea commercial fishers usually use beam trawls on both sides of the vessel to catch shrimp. The net is kept open with a large pole or beam. Floats make sure that the upper mouth is kept open and weighted bobbins keep the lower mouth open and sink the net (van Denderen 2015). Especially the Southern part of the Borkum Reef Ground is fished intensively while the other parts are less intensively fished (OCEANA 2020). The total amount of shrimp caught in the Borkum Reef Ground in 2017 was approximately 14,000,000 kg. Shrimp caught in the Borkum Reef Ground thus represent a small fraction (<1%) of the total catches (Wageningen University Research 2019).

However, due to the small mesh size used, shrimp fisheries catch large amounts of bycatch. In order to reduce by-catch shrimp fishnets often include a sieve netting which prevents the capture of fish larger than 10 cm. However, by-catch of small fish and undersized shrimp is not prevented (Glorius et al. 2015). A large fraction of by-catch is undersized shrimp, which can make up half of the total caught shrimp (Steenbergen et al. 2015). Fish species that are often caught are: plaice, common dab, sprat, herring, whiting and gobies. In the case of plaice, especially 0-1 year old plaice gets caught. This means that a fraction of young plaice cannot mature and reproduce which affects population stability (Glorius et al. 2015).

In addition to the effects on fish, shrimp trawlers affect benthic communities. Shrimp trawls affect the seabed by removal and damaging of bottom structures and benthos. After trawling clear marks are visible and the seabed can be affected up to 6 cm. Opportunistic short living species are less vulnerable or recover more quickly. However, long living slower growing species are very vulnerable to disturbance and often disappear resulting in an altered system with lower biodiversity (van Denderen et al. 2015). Benthic species that end up as bycatch include: anemones, crabs and cockles (Glorius et al. 2015; Steenbergen et al. 2015). Bottom-trawling, including shrimp trawling also poses major threats for sand mason worms and their associated reefs in particular (van Duren et al. 2016).

Generally, this type of fishing gear leads to a disappearance of underwater structures, lower biodiversity in benthic species and an affected age structure in fish populations that are caught as bycatch (Lindeboom et al. 2008).

5.2.2 Ship traffic

The North Sea is very important for shipping. The shipping lanes are used intensively and are among the busiest in the world. This sector has been growing steadily since 2005 (Statistics Netherlands 2016). Two of those shipping lanes go through the Borkum Reef Ground (fig 5) (Ministry of Infrastructure and the Environment & Ministry of Economic Affairs 2015).

Shipping includes multiple threats. Shipping is associated with accidental and operational discharges of oil. The latter mainly occurs inside ports because of tank washing, refueling etc. (Abdullah et al. 2008). Oil spills may cause direct mortality in seabirds but it can also affect shell thickness and affect the breeding season. The natural lipids in feathers are also affected by oil which causes the feathers to lose its water repellent capacity and isolation ability (Jägerbrand et al. 2019).

Ship traffic is also associated with the spread of invasive species. Shipping causes the transportation of species attached to the hull or via ballast water of ships which is used to stabilize the vessels at sea (Jägerbrand et al. 2019). Ship ballasting and reballasting happens in ports, but invasive species can spread easily towards other areas. Ship traffic is therefore considered as a major cause for the spread of invasive species. However, it remains difficult to predict where and when a non-indigenous species will become invasive and spreads and damages local ecosystems (WWF 2009).

Shipping may also cause accidental collisions with marine mammals which can have lethal consequences because of massive trauma. The suction of propellers, for example, draws marine mammals towards the ship (Jägerbrand et al. 2019).

Finally, ship traffic is also associated with underwater noise (Huntington et al. 2015). Sounds travels faster in water compared to air and can be heard over longer distances. Noise is caused by vibrations, machinery and sonar. The natural sounds in the area is interfered by anthropogenic noise and affects navigation, communication, habitat selection, detection of prey and mating. The harbor porpoise, for example, is less likely to be recorded around ships (Jägerbrand et al. 2019). The effects of noise on invertebrates and fish are less clear. Shipping noises are within the auditory range of fishes and may cause avoidance reactions (Huntington et al. 2015).

5.2.3 Sand extraction

In the Borkum Reef Ground licenses for sand extraction have been provided by the Department of Waterways and Public Works. In the Netherlands this sand is used to maintain shorelines for example.

In the North Sea sand is extracted using trailing suction hopper dredgers (figure 5). These vessels use suction to extract the sand while they continue to move forward (van Duin et al. 2017). Sand is extracted and is relocated after that. Sand extraction has considerable local impacts because benthos gets killed in the process, resulting in an empty seabed. Recovery may take years depending on the species in that particular area (Lindeboom et al. 2008).

Figure 5. Illustration of a trailing suction hopper dredger. (van Duin et al. 2017)

During extraction, fine particles get released and end up within the extraction area and the surrounding area. This may affect the production of algae, benthos and associated species. Sand extraction may also disturb fish, marine mammals and birds as a result of underwater noise and vessel movements (van Duin et al. 2017). Three licenses for sand extraction have been granted within the Borkum Reef Ground (OCEANA 2020). However, these areas are relatively small, major impacts are therefore not expected.

5.2.4 *Gas extraction*

Currently, two gas extraction areas are operating in the Borkum Reef Ground. One of these areas is located on the border between Germany and the Netherlands and the other one is located in the southern part of the Borkum Reef Ground. It is also likely that the possibility for gas extraction in other areas in the Borkum Reef Ground will be explored (figure 6) (ONE-Dyas B.V. n.d.)

Gas extraction includes a variety of activities that may affect the environment. The establishment of the mining platform and the construction of pipelines results in a loss of natural habitats. These construction activities may affect surrounding sediments. However, this is a temporary cause of disturbance and results suggest that surrounding areas recover quickly. Drilling may cause stress and flight behavior in marine organisms. Especially the Harbor Porpoise is sensitive for underwater disturbance. Disturbance above the surface can negatively affect seabirds. Discharge of pollutants such as wastewaters may affect surrounding habitats (Tamis et al. 2011).

Mining platforms also provide opportunities for some species because the platform itself provides hard substrate for species to establish. This may have a positive effect on mussels, algae, seaweeds anemones etc. (Lindeboom et al. 2008). Moreover, fishing activities are forbidden within 500 meter of the platform. These platforms thus offer shelter for certain species. Fish species associated with oil platforms include: Cod, Rockfish and Lings. However, under current regulations mining platforms need to be removed when they are no longer used (Fowler et al. 2018). Therefore, these platforms offer no lasting opportunities for recovery and these effects are not taken into account.

5.2.5 Military operations

Part of the Dutch North Sea is used for military exercises. The upper part of the Borkum Reef Ground overlaps with one of these areas. Military exercises includes activities such as shooting, flight exercises and mine detection. The intensity of these activities varies (Ministry of Infrastructure and the Environment & Ministry of Economic Affairs 2015). Military exercises can cause disturbance in marine fauna and ammunition remnants end up in the sea. However, research suggests that these remnants do not cause environmental damage (Noordzeeloket n.d.a).

5.2.6 Wind farms

Currently, an operational wind farm is located above the Borkum Reef Ground. Moreover, the options for extending this wind farm are explored (figure 6) (Rijkswaterstaat 2019). Offshore wind farms are associated with both positive and negative impacts. Just as mining platforms, wind farms offer hard substrate for species to attach on which in turn attract other organisms such as shellfish and fish (Bergström et al. 2014). These wind turbines eventually have to be decommissioned and offer no lasting protection opportunity for the species that attach to them (Fowler et al. 2018). These temporal effects will therefore not be taken into account.

Wind farms also have significant other effects. Pile driving during construction for example, causes significant avoidance behaviour in marine mammals. When wind farms are operating they also cause significant noise, which leads to avoidance behaviour in marine mammals (Bergstrom et al. 2014). Wind farms are also associated with bird mortality as birds collide with the blades from wind turbines. The vulnerability of particular species depends for example on flight altitude, flight agility and whether or not a species flies at night. Another threat of wind farms is disturbance and displacement of seabirds (Furness et al. 2013).

5.2.7 Flat oyster reef restoration

Recently, a flat oyster restoration project has been initiated by the WWF in the Borkum Reef Ground. Flat oysters are considered a keystone species because it has substantial influence on marine communities. Flat oyster reefs provide settlement for other species, they provide protection and nursery grounds, they stabilize sediments and they filtrate large amounts of water which has large impacts on visibility and water quality (Didderen et al. 2018).

According to historical records, flat oyster reefs were once present in 30% of the Dutch North Sea and within a part of the Borkum Reef Ground (Didderen et al. 2018).

Nowadays, flat oyster reefs are endangered and in need of protection. The protection of suitable areas where they were once present is therefore highlighted as a measure (Sas et al. 2019).

For this project 3D printed structures are used and 5500 kg oysters (approximately 80000 individual oysters) imported from Norway were attached to this structures before they were placed back in sea. Results seem promising as many oysters survived. There were also signs of growth and reproduction. Future evaluations will show whether this project and similar projects are successful in restoring the lost oyster reefs (Didderen et al. 2018).

5.2.8 *Concluding remarks*

The Borkum Reef Ground is an area that is used intensively for many human activities (figure 6).

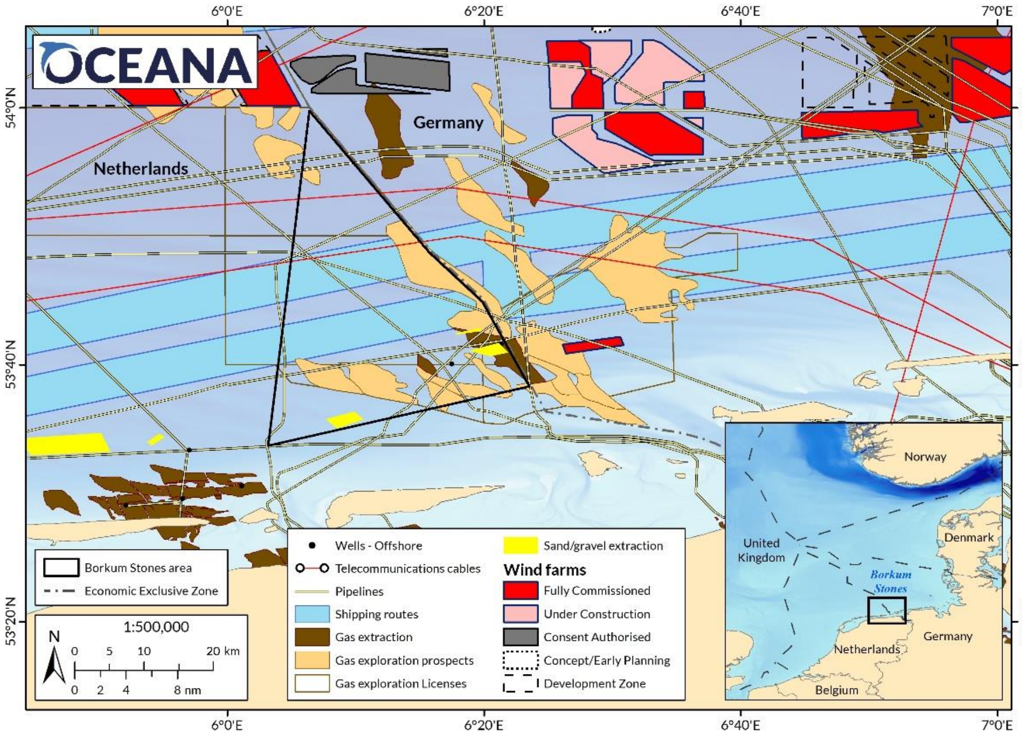


Figure 6. Overview of human activities in the Borkum Reef Ground. Borkum Reef Ground is indicated by the triangle. Activities include: shipping, sand extraction, gas extraction and wind farms © OCEANA (Oceana 2020).

Protecting this area against bottom trawling will therefore result in a multiple use MPA in which many other activities are allowed to continue. These activities have a variety of effects on different species groups. The combined impacts of protection against bottom trawling while taking into account other impacts on these species groups will be further explained in the next part.

5.3 Combined effects on biodiversity in the Borkum Reef Ground

Therefore, this chapter describes the combined effects of protection against bottom trawling while taking into account remaining activities on different marine species groups. These species groups are: marine mammals, sea birds, fish and benthos. They reflect some of the major biotic components in the marine ecosystem and their presence/absence provides information on the state of marine areas (Ministry of Infrastructure and the Environment & Ministry of Economic Affairs 2012).

5.3.1 Effects on benthos

The Borkum Reef Ground is one of the few marine areas within the Netherlands were hard substrate is present. This ranges from gravel to large boulders. The largest boulder had a length of 4 meter (Bos and Paijmans 2012). These boulders were overgrown with species characterized by hard substrate. Species that were found attached on or near these hard substrates include: plumose anemone, sponges, hydroids, dead men’s fingers, edible crab, Aesop shrimps, hermit crab and spider crabs (Bos et al. 2014).

Another relatively unique characteristic of the Borkum Reef Ground is the presence of dense aggregations of sand mason worms. Sand mason worms are a so called keystone species that can alter its habitat by creating biogenic reefs. Sand mason worms construct these reefs by gluing shell fragments with sand grains. These reefs provide shelter, food and substrate for other species and therefore enhance biodiversity. The sand mason worm itself lives only 1 or 2 years. However, the tubes remain to exist for several years and offer settlement for young worms (Bureau Waardenburg 2020). Other species that were found in these biogenic reefs included many worms such as bee spionids and bristle worms. In total, 23 different species of worms were identified, followed by 8 crustaceans and 5 molluscs (Bos et al. 2014). These areas characterized by the presence of hard substrate or biogenic reefs create a suitable habitat for many species and are characterized by higher biodiversity compared to more homogenous (Coolen et al. 2015).

The protection against bottom trawling will have most effects on this species group because bottom trawling removes both the suitable habitats for benthic organisms as well as the direct removal of benthic organisms. Even though a large fraction of the Borkum Reef Ground is not fished intensively, trawling leaves behind clear marks in the seabed (van Denderen 2015). This suggests that less intensive fishing pressures also causes considerable damage. These findings are supported by a global analysis on trawling impacts that finds that trawling frequencies of 1 time per year may already cause an average decline of 15.5% of benthic biomass. Another finding is that communities associated with hard substrate may be more sensitive to trawling. These communities often include more long-lived species which are especially sensitive to trawling (Hiddink et al. 2017). Long-lived species are present within the Borkum Reef Ground (Bos et al. 2011). These species in particular could therefore benefit from protection against bottom trawling. Recovery time also depend on the presence of less impacted areas because individuals from these areas are able to recruit and migrate (Hiddink et al. 2017). Recovery in more intensively trawled parts of the Borkum Reef Ground may therefore be expected as surrounding areas are not fished intensively. Other human activities that may affect benthos are sand extraction and gas extraction. The main impact is direct mortality because of the removal of sand and organisms and because of platform and pipeline construction. However, these activities have very local impacts (Lindeboom et al. 2005). Therefore, it is not likely that these activities have a widespread negative effect on the benthic communities within the Borkum Reef Ground.

In conclusion, positive effects are expected in the benthic communities within the Borkum Reef Ground as less intensive bottom trawling may already cause significant damage. This is because bottom trawling causes direct removal of benthic species as well as their associated habitats. Moreover, the long-lived species within the Borkum

Reef Ground are especially vulnerable to bottom trawling. The removal of this harmful fishing practice allows for recruitment of these vulnerable species within other parts of this area and may therefore have an overall positive effect on benthic biodiversity within the Borkum Reef Ground.

5.3.2 *Effects on fish*

The distribution ranges of fish species are determined by its ability to return to a certain area. However, determining where a population is present at a certain point in time depends on environmental influences including human impacts and life history characteristics. Therefore, fish species often show considerable spatio-temporal variation (Teal 2011). This makes it difficult to determine how important particular areas are for different species of fish and most information that is available focuses on commercial species.

Survey catch data showed that the abundance of plaice is highest in south-eastern North Sea. The Dutch exclusive economic zone represents a large fraction of this area. Shallower coastal waters are used as nursery areas for juvenile Plaice before they move into deeper waters. The shallow Dutch exclusive economic zone is therefore important for juvenile Plaice (Teal 2011). The Borkum Reef Ground in particular is suspected to be a spawning area for Plaice (van Kooten et al. 2015). The distribution of Sole is more or less similar to that of Plaice. Shallower waters are also used as spawning and nursery areas. The Dutch exclusive economic zone and the Borkum Reef Ground specifically could therefore be important for juvenile Sole. Another commercially interesting species is Cod. They are found in almost every area in the North Sea. Cod doesn’t show strong migration patterns and spawning and feeding grounds are not separated (Teal 2011). Cod also prefers the presence of hard substrate which is present in the Borkum Reef Ground. It is suggested that the Borkum Reef ground functions as a nursing area for this species. The Borkum Reef Ground may also function as a spawning and/or nursing area for shrimp and sprat (van Kooten et al. 2015). For many other species, distribution patterns remain largely unknown. Other species that were observed in the Borkum Reef Ground are: goby’s, gray gurnards and dragonets (Bos et al. 2014). Some of these non-commercial species could be of great importance for higher trophic levels, but more information is needed in order to assess their presence and distribution patterns (Teal 2011).

The major fishing technique in the Borkum Reef Ground is shrimp trawling. Undersized Plaice and Goby’s often ends up as bycatch (Glorius et al. 2015). The expected effects are small because large parts of the Borkum Reef Ground is not fished intensively. However, shrimp trawling also removes underwater structures and the benthic species that are associated with them (Lindeboom et al. 2008). Sole and Plaice, for example, are predatory species that feed on benthic species. Protection against bottom trawling may therefore have positive effects as prey species are not removed any more by fisheries (Rabaut et al. 2008). However, the exact effects of bottom trawling on food availability for demersal fish remains a point of discussion after the experience with the Plaice Box (box 2) and is in need of further investigation (Vrooman et al. 2018). Based on current information it is likely that protection against bottom trawling will not have significant on fish stocks. However, the exact effects of bottom trawling on food availability are difficult to predict and requires long-term monitoring. Additional information on the presence and distribution of non-commercial species is also needed in order to get a better understanding of ecosystem functioning and the interactions between species groups.

5.3.3 *Effects on marine mammals*

Marine mammals that have been observed in the Borkum Reef Ground are the common seal, the grey seal and the harbor porpoise. The common seal uses this area intensively for foraging. This area is less important for the grey seal as they are mainly located in the western part of the North Sea. However, populations seem to move more towards the east. Therefore, this area may become more important in the future for the grey seal (Bos and Paijmans 2012). Another marine mammal that has been observed in the Borkum Reef Ground is the harbor porpoise. The distribution of the harbor porpoise is not well known. Observations suggest that the harbor porpoise is present in all parts of the Dutch North Sea. However, temporal and spatial patterns need to be studied. It has been suggested that the German Borkum Reef Ground may be a key foraging area for harbor porpoise, but it is currently not clear whether or not this could be the case for the Dutch Borkum Reef Ground (Bos et al. 2011).

There are multiple activities within the Borkum Reef Ground that affect marine mammal populations. For example, the shipping lanes that cross the Borkum Reef Ground are used intensively (Ministry of Infrastructure and the Environment & Ministry of Economic Affairs 2015). Ship traffic causes collisions with marine mammals which may cause direct mortality. Shipping is also associated with disturbance and avoidance behaviour in marine mammals (Jagerbrand et al. 2019).

Other impacts that affect marine mammals include gas extraction and wind farms, which are expected to expand in the future. Current use of these mining platforms and wind farms and future construction activities cause disturbance and marine mammals tend to avoid these areas (Bergstrom et al. 2014; Tamis et al. 2011). As no significant effects are expected on fish populations marine mammals are not likely to benefit from increased food availability. Therefore, marine mammals are not likely to benefit from protection against bottom trawling. On the contrary, the current human activities and the development of these activities results in increasing pressures on this particular species group.

5.3.4 *Effects on seabirds*

Seabird assessments are based aerial surveys and ship-based surveys. These observations make it clear that there are large temporal variations in the presence of seabirds in the Dutch part of the North Sea. Therefore, a variety of bird species are observed at the Borkum Reef Ground at different times. During spring the migrating Red-throated diver is relatively abundant in the Borkum Reef Ground (Bos et al. 2011).

There are multiple activities that could potentially threaten seabirds. First of all shipment is associated with accidental oil spills that may cause direct mortality or affect the condition of seabirds (Jagerbrand et al. 2019). Another threat is posed by wind farms and the extension of current wind farms. They cause direct mortality via collisions and avoidance behaviour in seabirds (Furness et al. 2013). Another threat is the possible disturbance caused by activities on and near gas extraction platforms (Tamis et al. 2011).

The possible positive effects of protection against bottom trawling would be the availability of fish (Leopold et al. 2011). However, as mentioned before, the Borkum Reef Ground is not fished intensively. That is why no significant positive effects are to be expected on fish populations from which seabirds can benefit. If any, the development of other activities are more likely to have a negative effect on seabird populations.

5.4 Application of the eco point valuation method on the Borkum Reef Ground

The eco point valuation method allows for a quantification of biodiversity effects. The eco point valuation method is based on the quality of habitats within an area, the area size and a weighing factor can be applied when multiple areas are compared (chapter 4.2.5) (Sijtsma et al. 2009). The latter is not the case for the Borkum Reef Ground as this area will not be compared to another area.

The main difficulty of applying the eco point valuation method is assessing habitat quality and possible increase in quality. In order to assess the current quality of this area maximum quality needs to be determined. However, the North Sea has been fished intensively for many decades (Compendium voor de Leefomgeving 2017; Didderen et al. 2018). Therefore, it is very difficult to determine what maximum quality could be or determine what these habitats could look like when they are left intact.

There remain also questions on the exact effects of protection on different species groups as well as the abundancy and distribution of species. Fish and seabirds for example, shows large temporal variations (Teal 2011; Bos et al. 2011). Moreover, other human activities within this area cannot be ignored as they have significant effects on different species groups. The impacts of wind farms and gas extraction may also increase as additional wind turbines will be constructed and other opportunities for gas extraction may be explored. These future developments are an extra cause of uncertainty when possible positive effects need to be predicted.

Therefore, quantification of the effects on biodiversity using the eco point valuation method is not recommended as long as there are this many knowledge gaps. Quantification of biodiversity effects using the eco point valuation method would mean that these uncertainties are added up and the results would therefore be highly unreliable.

5.5 Concluding remarks

This case study describes the possible benefits of protecting the Borkum Reef Ground against bottom trawling by taking into account different human activities and multiple species groups.

Quantification of these effects using the eco point valuation method is currently not recommended as there are large knowledge gaps concerning habitat quality and quality improvement which is needed in order to calculate eco points. Therefore, a qualitative approach has been used in order to provide insights in different human activities and the impacts these activities have on different species groups.

What can be concluded is that protection against bottom trawling will most likely have positive effects on benthos. Effects on other species groups such as fish, marine mammals and seabirds are not expected but there remain large uncertainties about the exact extent of effects. Future developments in other human activities may also cause increased pressure on marine mammals and seabirds.

This case study points out different knowledge gaps. These knowledge gaps include uncertainties about assessing habitat quality, uncertainties about species distribution and uncertainties about the effects of protection against bottom trawling on different species groups. It is clear that monitoring is very much needed in order to get a better understanding of MPA effects on different components of the marine ecosystem and the interactions between them. The Borkum Reef Ground offers the opportunity to put in place long term monitoring schemes and gather data both before as well as after MPA establishment. This kind of information is essential for predicting future MPA effects and support their implementation.

6 Conclusions and recommendations

The first aim of this report was to perform an assessment on the current status of MPAs in Europe in order to see whether or not measures are sufficient to protect marine ecosystems and to determine what factors influence the performance of MPAs. The second aim of this report was to assess what the benefits of MPAs are and how they can be quantified or monetised in socio-economic analyses in order to support decision-making. Therefore, this report focused on MPAs both within and to some extent outside Europe in order to see what can be learnt from them. Moreover, the lessons learnt from these analyses were applied in a case study, the Borkum Reef Ground, to assess the applicability of the recommendations and highlight current knowledge gaps.

6.1 The current status of MPAs within Europe

Based on the assessment on the current status of MPAs in Europe the following conclusions can be drawn. First of all, it is clear that many factors contribute to MPA performance. By assessing these factors in the light of the European MPA network it is very likely that the MPA network is underperforming.

Overall targets such as the BDC target to protect at least 10% of the marine environment within MPAs and within sub regions are not achieved. Also, many habitats and species are currently not protected within MPAs, while being listed as in need of protection. Furthermore, many MPAs in Europe are multiple use MPAs in which many human activities are allowed to continue although it is demonstrated that no-take MPAs usually perform much better. The updated conservation targets of the European Biodiversity Strategy on highly protected areas recognizes this need, which is an important step forward concerning nature conservation within Europe. One also needs to be patient; as it may take up to 10 years before significant effects of protection can be expected.

The appropriate MPA size and the necessary level of connectivity between MPAs is difficult to determine, because this depends on the species that need to be protected within MPAs and their mobility and/or migration patterns. MPA size may also depend on its connectivity to other MPAs as a more isolated MPA needs to be more self-sustaining, thereby requiring a larger area including a variety of habitats. Genetic research may offer the opportunity to better assess connectivity in the marine context.

Finally, management of MPAs is often not sufficient. There is often a lack of resources such as budget and staff and therefore monitoring does not take place. Monitoring is of the utmost importance because otherwise the size of the effects of MPAs and benefits remain largely unknown.

6.2 Benefits of MPAs and their representation in socio-economic analyses

In order to get a better understanding of MPAs, the following potential benefits were analysed in more detail: ecological benefits, carbon storage as benefit, benefits for fishers and benefits for recreation and tourism.

Based on this analyses it is clear that MPAs are capable of generating a variety of benefits. However, there are also large uncertainties. Ecological benefits such as increases in abundancy and size of organisms within MPAs, increases in biodiversity and trophic cascades are difficult to predict beforehand because the marine environment is characterized by complexity and many interactions. Research shows that these benefits can vary substantially between MPAs.

Other ecological benefits such as carbon storage and sequestration by marine coastal systems are questionable, because disturbance may cause the carbon to be released and thus turn the carbon sink into a carbon source. However, even though a quantification of ecological effects may be hardly possible, it is clear that coastal systems generate a variety of benefits and restoration projects often fail. This is why the precautionary principle is recommended.

Spill-over effects from which fishers could benefit have been demonstrated in a few cases on a small scale. This process takes time as there first needs to be an improvement within MPAs before effects can be expected outside MPAs. Fishers may also experience increased costs because of MPAs because they need to fish further away or because of increased competition outside the MPA, and it is not clear whether they will offset the increased costs. These spill-over effects therefore do not directly translate into economic benefits.

Benefits for recreation and tourism have been acknowledged by stakeholders in several studies. Based on these studies it is also clear that MPAs support a variety activities and jobs and generate substantial income. However, a difficulty is that there is often no baseline because these activities and their economic effects were not studied prior to MPA designation. This makes it difficult to distinguish between the effect of the site and the effect of the designation of that particular area as an MPA.

Multiple methods have been used to quantify or monetize these benefits in ex ante socio-economic analyses. These include: bio economic modelling, benefit transfers, stated preference methods, MCAs and the eco point valuation method.

Bio economic modelling is a tool which combines economic and ecological data in mathematical functions to predict future scenarios. These models have been applied in the context of MPAs and often focus on the relations between fish stocks and fisheries and sometimes include other benefits as well such as effects on recreation and tourism.

Benefit transfers use the value obtained in one site to predict the value of another site. Benefit transfers can be applied to a variety of ecosystem services as long as primary valuation data is available and the studied site is highly similar to the policy site (i.e. the site that will be valued). In order for this method to be applied more widely more primary valuation studies are needed as these are currently scarce in the marine context.

Stated preference methods have also been applied in the context of MPAs. Stated preference methods are based on surveys in which the willingness to pay (i.e. amount of money that people are willing to spend on a welfare gain) or willingness to accept (i.e. amount of money that people are willing to accept for the loss of a certain good) is obtained. These methods focus on the value of non-market benefits such as the value placed on conserving nature for example. Stated preference methods are highly disputed and it is questioned whether or not the results are reliable.

MCAs can be used to explore the trade-offs of pros and cons of different policy options. This is especially useful when different criteria cannot be expressed in one single unit, thereby acknowledging that it is difficult to produce a single right answer. Aggregation of different scores is considered to be difficult and a common pitfall is the use of too many criteria. Decision-making based on MCAs may therefore be more difficult.

Finally, the eco point valuation method has been applied in the context of MPAs. The eco point valuation method can be used to quantify biodiversity effects of different measures, based on area size, area quality and a weighing factor that can be applied when different areas are compared. The main advantage of the application of the eco point valuation method is that it allows to include the intrinsic value of nature in CBAs. Therefore, the eco point valuation method suits the overall goals of MPAs; the protection and recovery of marine ecosystems.

Based on the benefits and the valuation methods that have been assessed, the eco point valuation method is recommended when CBAs need to be performed in the context of MPAs. Even though MPAs are capable of generating many benefits, there are large uncertainties about the extent of the benefits that can be expected. This is due to the fact that marine ecosystems are very complex. Nevertheless, since nature quality contributes to the generation of many ecosystem services, it is crucial to get a thoroughly understanding of the effects of MPAs on marine ecosystems. Therefore, the eco point valuation method is considered to be the best (applicable) method that could be used to quantify these effects.

6.3 Application of the eco point valuation method on the Borkum Reef Ground

An attempt has been made to estimate the future benefits of a possible MPA in the Netherlands, the Borkum Reef Ground in order to test the applicability of the eco point valuation method. Based on this case study it is concluded that even though the eco point valuation could be useful it is not yet recommended, because essential knowledge on the potential ecological impacts of the designation of the MPA is missing, in this case, but possibly also in similar cases.

The eco point valuation method is based on area size, area quality and a weighing factor when multiple areas have to be compared. The latter is not the case here. The main difficulty is assessing area quality and quality gain. This is based on the presence of characteristic species for different habitats present within the area and the effects of protection on these species. In order to assess area quality, information is needed on what the maximum quality could be (i.e. a pristine area). However, this is very difficult to assess as the marine environment has been used intensively for decades. In order to determine a gain in eco points, an understanding of the effects of MPAs in the marine environment is needed. Based on the analyses of ecological effects it is clear that MPAs can have many ecological effects but these are also very difficult to predict beforehand. That is why applying the eco point valuation in order to quantify future biodiversity effects is currently not recommended in the context of MPAs as this would mean that uncertainties are added up which makes the results unreliable. Instead of quantifying or monetizing benefits a qualitative approach has been adopted in which the possible effects are described and knowledge gaps are highlighted. Based on this description it is clear that protecting the Borkum Reef Ground against bottom trawling will most likely benefit benthic organisms. The results also indicate that there are many human activities that have adverse effects on the marine environment. That is why only moderate effects have been predicted but large uncertainties remain.

6.4 Recommendations

From the above it appears that many factors influence MPA performance and it is likely that many MPAs are underperforming. Moreover, many effects go unnoticed as monitoring is insufficient. It is clear that MPAs provide benefits but there remain large uncertainties about their extent. It is also clear that biodiversity contributes to many benefits but the exact link between them is less clear. That is why the eco point valuation method which can be used to quantify biodiversity effects was recommended in the first place. However the application of this method on the case of the Borkum Reef Ground highlighted significant knowledge gaps. These knowledge gaps are related to the estimation of the current quality of marine areas and the possible effects of MPAs, which are obviously both needed to be able to determine the potential increase in quality of the ecosystem as a result of MPA designation.

For now, it seems that predicting the effects of MPAs on the marine ecosystem is very challenging. Therefore, quantifying or monetizing these effects is not yet recommended as the results may be very unreliable. Therefore, a qualitative approach in which the knowledge gaps are highlighted may be a better solution for the time being. This should be based on the best available data and could be strengthened by expert judgement.

Application of the eco point valuation is still recommended in the future as it takes into account the value of biodiversity and allows for the quantification of the effects on biodiversity, the driver of many ecosystem services. The main difficulty of applying this method is that it is very difficult to determine what a more pristine area could look like in terms of the presence and abundancy of different species in order to establish current area quality, relative to its pristine state.

It is highly recommended that a fraction of existing MPAs are designated as no-take MPAs. Multiple use MPAs are not as effective as no-take MPAs and the results may therefore be moderate. When effects are not optimal or non-existent, future MPA implementation may face more resistance. This has been the case after the establishment of the Plaice Box, which gave some disappointing results. It is therefore of utmost importance to understand MPA performance and the factors that influence them. Designation of more no-take MPAs will likely yield better results and support future implementation as well as contribute more significantly to the realization of conservation goals. These no-take zones can be used to get a better understanding of what a more pristine marine area could look like. In that way, the quality of current quality of marine areas can be determined and scored. This information is necessary in order to apply the eco point valuation method.

In order to predict the effects of protection connectivity should be taken into account as well. Assessing appropriate MPA size and connectivity between MPAs needs a customized approach as this highly depends on the species that need to be protected within MPAs. The assessment of the required level of connectivity in the marine environment is difficult, but could be enhanced by applying genetic research. In this way, gene flows can be assessed by comparing genetic similarity/dissimilarity between populations in different areas. This data can also be used to identify source and sink populations and determine which additional areas should be protected. When placed correctly, MPAs are more likely to generate significant effects which supports future designation. This information can be used to identify the importance of a particular area for certain species or species groups. In the case of the Borkum Reef Ground it is not exactly clear how important this area is for marine mammals such as the harbor porpoise and different fish species for example. This type of information is essential to predict, quantify and monetize effects of MPAs.

Furthermore, it is important to put in place robust monitoring schemes in order to assess MPA effects and compare MPAs to each other. The lack of monitoring data makes it extremely difficult to determine what the effects of MPAs could be. It is highly recommended to address this problem at an international level, for example at the level of RSCs and/or the EU. It is recommended to not only monitor presence of species, but also their abundancy, the temporal variations and the interactions between species groups. The interactions between benthos and predatory fish species for example, needs to be studied in more detail.

Combining the information on area quality and the effects of MPAs allows for application of the eco point valuation in the marine context in the future. For now, the first thing to do is to get a better understanding of the functioning of marine ecosystems and the effects of MPAs in order to be able to predict the potential benefits that they can provide.

7 References

Aanesen, M., C. Armstrong, M. Czajkowski, J. Falk-Petersen, N. Hanley, and S. Navrud. 2015. Willingness to pay for unfamiliar public goods: Preserving cold-water coral in Norway. Ecological Economics 112:53–67.

Abdulla, A., Linden, O. 2008. Maritime traffic effects on biodiversity in the Mediterranean Sea: Review of impacts, priority areas and mitigation measures. Malaga, Spain: IUCN Centre for Mediterranean Cooperation. 184 pp.

Abesamis, R. A., and G. R. Russ. 2005. Density-dependent spillover from a marine reserve: Long-term evidence. Ecological Applications 15:1798–1812.

Alban, F., Appe´re´, G., & Boncoeur, J. 2006. Economic Analysis of Marine Protected Areas. A Litterature Review. EMPAFISH Project, Booklet no 3. Murcia: Editum.

Alban F., Person J., Roncin N. and Boncoeur J., 2008. Analysis of SocioEconomic Survey Results. EMPAFISH Project. 139 pp.

Alexander, D., A. Wellard, K. De, R. Moliner, A. & Fletcher, L. Doria, M. Smith, B. Stoker, and D. Mortimer. 2016. Exploring the Components and Processes of Marine Ecosystems Critical to Ecosystem Service Generation. Marine Ecological Surveys Ltd – a report for the Joint Nature Conservation Committee.

Álvarez García, M. Á., J. González Álvarez, L. García de la Fuente, and A. Colina Vuelta. 2012. Valuing the benefits of designating a network of scottish MPAs in territorial and offshore waters:104.

Armstrong, C. W. 2007. A note on the ecological-economic modelling of marine reserves in fisheries. Ecological Economics 62:242–250.

Attrill MJ, Austen MC, Cousens SL, Gall SC, Hattam C, Mangi S, Rees A, Rees S, Rodwell LD, Sheehan EV, Stevens, TF. 2012. Lyme Bay – a case-study: measuring recovery of benthic species; assessing potential “spillover” effects and socio-economic changes, three years after the closure. Report 1: Response of the benthos to the zoned exclusion of bottom towed fishing gear in Lyme Bay, March 2012. Report to the Department of Environment, Food and Rural Affairs from the University of Plymouth-led consortium. Plymouth: University of Plymouth Enterprise Ltd. 82 pages

Ballantine, B. 2014. Fifty years on: Lessons from marine reserves in New Zealand and principles for a worldwide network. Biological Conservation 176:297–307.

Barbier EB, Hacker SD, Kennedy C, Koch EW, Stier AC, and Silliman BR. 2011. The value of estuarine and coastal ecosystem services. Ecological Monographs 81(2):169–193.

Bateman, I. J., G. M. Mace, C. Fezzi, G. Atkinson, and K. Turner. 2011. Economic analysis for ecosystem service assessments. Environmental and Resource Economics 48:177–218.

Bayraktarov, E., M. I. Saunders, S. Abdullah, M. Mills, J. Beher, H. P. Possingham, P. J. Mumby, and C. E. Lovelock. 2015. The cost and feasibility of marine coastal restoration. Ecological Applications 26:1055–1074.

Beare, D., A. D. Rijnsdorp, M. Blaesberg, U. Damm, J. Egekvist, H. Fock, M. Kloppmann, C. Röckmann, A. Schroeder, T. Schulze, I. Tulp, C. Ulrich, R. Van Hal, T. Van Kooten, and M. Verweij. 2013. Evaluating the effect of fishery closures: Lessons learnt from the Plaice Box. Journal of Sea Research 84:49–60.

Bergström, L., L. Kautsky, T. Malm, R. Rosenberg, M. Wahlberg, N. Åstrand Capetillo, and D. Wilhelmsson. 2014. Effects of offshore wind farms on marine wildlife - A generalized impact assessment. Environmental Research Letters 9.

Bertram, C., T. Dworak, S. Görlitz, E. Interwies, and K. Rehdanz. 2014. Cost-benefit analysis in the context of the EU Marine Strategy Framework Directive: The case of Germany. Marine Policy 43:307–312.

De Blaeij, A., and R. Verburg. 2011. Voor- en nadelen van het gebruik van natuurpunten bij het bepalen en monetariseren van natuureffecten:37.
Bos, O.G., Witbaard, R., Lavaleye, M. S. S., Moorsel, G. W. N. M., Teal, L. R., Hal, R. V., ... & Geelhoed, S. C. V. 2011. Biodiversity hotspots on the Dutch Continental Shelf: a marine strategy framework directive perspective.
Bos O.G., Paijmans A.J. 2012. Natuurverkenning naar de Borkumer Stenen. Project Aanvullende Beschermde Gebieden (http://edepot.wur.nl/240319). Report C137/12, IMARES
Bos, O.G., Glorius, S.T., Coolen, J.W.P., Cuperus, J., van der Weide, B.E., Aguera Garcia, A., van Leeuwen, P.W., Lengkeek, W., Bouma, S., van Hoppe, M. & H.M.L. van PeltHeerschap, (2014) Natuurwaarden Borkumse Stenen: project aanvullende beschermde gebieden. Report number C115/14, IMARES Wageningen UR, Wageningen.
Bos, F., Ruijs, A. 2019. Biodiversity in the Dutch practice of cost-benefit analysis. CPB Netherlands Bureau for Economic Policy Analysis, the Netherlands.
Bureau Waardenburg. 2020. Options for biodiversity enhancement in offshore wind farms. Knowledge base for the implementation of the Rich North Sea Programme. Bureau Waardenburg Rapportnr.190153. Bureau Waardenburg, Culemborg.
Carr, M. H., J. E. Neigel, J. A. Estes, S. Andelman, R. Robert, M. H. Carr, J. E. Neigel, J. A. Estes, S. Andelman, R. R. Warner, and J. L. Largier. 2003. Comparing Marine and Terrestrial Ecosystems : Implications for the Design of Coastal Marine Reserves Warner and John L . Largier Source : Ecological Applications , Vol . 13 , No . 1 , Supplement : The Science of Marine Reserves Published by : Wiley Stable. Ecological Applications 13:S90–S107.
CBD. 2020. History of the Convention. Retrieved from: https://www.cbd.int/history/.
Chae, D. R., P. Wattage, and S. Pascoe. 2012. Recreational benefits from a marine protected area: A travel cost analysis of Lundy. Tourism Management 33:971–977.
Claudet, J., C. W. Osenberg, L. Benedetti-Cecchi, P. Domenici, J. A. García-Charton, Á. Pérez-Ruzafa, F. Badalamenti, J. Bayle-Sempere, A. Brito, F. Bulleri, J. M. Culioli, M. Dimech, J. M. Falcón, I. Guala, M. Milazzo, J. Sánchez-Meca, P. J. Somerfield, B. Stobart, F. Vandeperre, C. Valle, and S. Planes. 2008. Marine reserves: Size and age do matter. Ecology Letters 11:481–489.
Claudet, J., D. Pelletier, J. Y. Jouvenel, F. Bachet, and R. Galzin. 2006. Assessing the effects of marine protected area (MPA) on a reef fish assemblage in a northwestern Mediterranean marine reserve: Identifying community-based indicators. Biological Conservation 130:349–369.
Compendium voor de Leefomgeving. 2017. Bodemfauna Noordzee en bodemvisserij. retrieved from: https://www.clo.nl/indicatoren/nl1251-bodemfauna-noordzee-en-boomkorvisserij.
Coolen, J. 2017. North Sea Reefs. Duikdenoordzeeschoon.Nl.
Coolen, J. W. P., O. G. Bos, S. Glorius, W. Lengkeek, J. Cuperus, B. van der Weide, and A. Agüera. 2015. Reefs, sand and reef-like sand: A comparison of the benthic biodiversity of habitats in the Dutch Borkum Reef Grounds. Journal of Sea Research 103:84–92.
Côté, I. M., I. Mosqueira, and J. D. Reynolds. 2001. Effects of marine reserve characteristics on the protection of fish populations: A meta-analysis. Journal of Fish Biology 59:178–189.
Davis, K. J., G. M. S. Vianna, J. J. Meeuwig, M. G. Meekan, and D. J. Pannell. 2019. Estimating the economic benefits and costs of highly-protected marine protected areas. Ecosphere 10.
Van Denderen, P.D. 2015 Ecosystem effects of bottom trawl fishing. Wageningen

Institute of Animal Sciences. Wageningen University. Wageningen.
Didderen, K., W. Lengkeek, P. Kamermans, B. Deden, E. Reuchlin-Hugenholtz, J. H. Bergsma, ..., and H. Sas. 2018. Pilot to actively restore native oyster reefs in the North Sea. Bureau Waardenburg Report 10-:1–33.
Dudley, N. (2013). Guidelines for Applying Protected Area Management Categories. Gland, Switzerland: IUCN. x + 86pp. WITH Stolton, S., P. Shadie and N. Dudley (2013). IUCN WCPA Best Practice Guidance on Recognising Protected Areas and Assigning Management Categories and Governance Types, Best Practice Protected Area Guidelines Series No. 21, Gland, Switzerland: IUCN. xxpp.
Van Duin, C., Peerdeman, V., Jaspers, H., Bucholc, A. 2017. Winning ophoogzand Noordzee 2018 t/m 2027. Sweco Nederland B.V.
Dureuil, M., Boerder, K., Burnett, K., Froese, R., Worm, B. 2018. Elevated trawling inside protected areas undermines conservation outcomes in a global fishing hot spot. Science 362:1403-1407.
van Duren, L. A., A. Gittenberger, A. C. Smaal, M. van Koningsveld, R. Osinga, J. A. Cado van der Lelij, and M. B. de Vries. 2016. Rijke riffen in de Noordzee:82.
EC. 2000. Directive 2000/60/EC of the European Parliament and of the council directive of 23 October 2000 establishing a framework for Community action in the field of water policy
EC. 2006. Communication from the Commission Halting the loss of biodiversity by 2010 and beyond Sustaining ecosystem services for human well-being. Retrieved from: https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELLAR:1b1ce3dc-f379-459f-8db4-00a9bc9921c1
EC. 2008. Directive 2008/56/EC of the European Parliament and of the Council of 17 June 2008 establishing a framework for community action in the field of marine environmental policy (MSFD).
EC. 2013. Regulation (EU) No 1380/2013 of the European Parliament and of the Council of 11 December 2013 on the Common Fisheries Policy, amending Council Regulations (EC) No 1954/2003 and (EC) No 1224/2009 and repealing Council Regulations (EC) No 2371/2002 and (EC) No 639/2004 and Council Decision 2004/585/EC
EC. 2014. The EU Birds and Habitats Directives. Retrieved from: https://ec.europa.eu/environment/nature/info/pubs/docs/brochures/nat2000/en.pdf
EC. 2017. Study on the economic benefits of Marine Protected Areas Literature review analysis. Luxembourg: Publications Office of the European Union. Retrieved from: https://op.europa.eu/en/publication-detail/-/publication/85897a77-b0c7-11e8-99ee-01aa75ed71a1.
EC. 2018. Study on the economic benefits of MPAs. Luxembourg: Publications Office of the European Union. Retrieved from: https://op.europa.eu/en/publication-detail/-/publication/dbe3d250-b0b5-11e8-99ee-01aa75ed71a1.
EC. 2019. Regional Sea Conventions. Retrieved from: https://ec.europa.eu/environment/marine/international-cooperation/regional-sea-conventions/index_en.htm.
EC. 2020a. Communication from the Commission to the European Pariament, the Council, the European Economic and Social Committee and the Committee of the Regions EU Biodiversity Strategy for 2030 Bringing nature back into our lives. Retrieved from: https://eur-lex.europa.eu/resource.html?uri=cellar:a3c806a6-9ab3-11ea-9d2d-01aa75ed71a1.0001.02/DOC_1&format=PDF.
EC. 2020b. Legislation: the Marine Strategy Framework Directive. Retrieved from: https://ec.europa.eu/environment/marine/eu-coast-and-marine-policy/marine-strategy-framework-directive/index_en.htm

Edgar, G. J., R. D. Stuart-Smith, T. J. Willis, S. Kininmonth, S. C. Baker, S. Banks, N. S. Barrett, M. A. Becerro, A. T. F. Bernard, J. Berkhout, C. D. Buxton, S. J. Campbell, A. T. Cooper, M. Davey, S. C. Edgar, G. Försterra, D. E. Galván, A. J. Irigoyen, D. J. Kushner, R. Moura, P. E. Parnell, N. T. Shears, G. Soler, E. M. A. Strain, and R. J. Thomson. 2014. Global conservation outcomes depend on marine protected areas with five key features. Nature 506:216–220.

EEA. 2015a. State of Europe's seas. Retrieved from: https://www.eea.europa.eu/publications/state-of-europes-seas

EEA. 2015b. Marine protected areas in Europe's seas an overview and perspectives for the future. Retrieved from: https://www.eea.europa.eu/publications/marine-protected-areas-in-europes.

EEA. 2017. Marine Regions and Subregions. Technical Document. Delineation of the MSFD Article 4 marine regions and subregions. Retrieved from https://www.eea.europa.eu/data-and-maps/data/europe-seas/marine-regions-and-subregions-1/technical-document/at_download/file.

Esmail, B., and D. Geneletti. 2018. Multi-criteria decision analysis for nature conservation: A review of 20 years of applications. Methods in Ecology and Evolution 9:42–53.

FAO. 2018. The State of World Fisheries and Aquaculture 2018 - Meeting the sustainable development goals. Rome. Licence: CC BY-NC-SA 3.0 IGO.

Federal Agency for Nature Conservation. n.d. Borkum Reef Ground SAC. Retrieved from: https://www.bfn.de/en/activities/marine-nature-conservation/national-marine-protected-areas/north-sea-eez/borkum-reef-ground-sac.html.

Fenberg, P. B., J. E. Caselle, J. Claudet, M. Clemence, S. D. Gaines, J. Antonio García-Charton, E. J. Gonçalves, K. Grorud-Colvert, P. Guidetti, S. R. Jenkins, P. J. S. Jones, S. E. Lester, R. McAllen, E. Moland, S. Planes, and T. K. Sørensen. 2012. The science of European marine reserves: Status, efficacy, and future needs. Marine Policy 36:1012–1021.

Fletcher, S., S. Rees, S. Gall, W. Dodds, and L. Rodwell. 2014. Assessing the socio-economic benefits of marine protected areas:1–145.

Follesa, M. C., R. Cannas, A. Cau, D. Cuccu, A. Gastoni, A. Ortu, C. Pedoni, C. Porcu, and A. Cau. 2011. Spillover effects of a Mediterranean marine protected area on the European spiny lobster Palinurus elephas (Fabricius, 1787) resource. Aquatic Conservation: Marine and Freshwater Ecosystems 21:564–572.

Forcada, A., C. Valle, P. Bonhomme, G. Criquet, G. Cadiou, P. Lenfant, and L. S. L. José. 2009. Effects of habitat on spillover from marine protected areas to artisanal fisheries. Marine Ecology Progress Series 379:197–211.

Foster, N. L., S. Rees, O. Langmead, C. Griffiths, J. Oates, and M. J. Attrill. 2017. Assessing the ecological coherence of a marine protected area network in the Celtic Seas. Ecosphere 8.

Fowler, A. M., A. M. Jørgensen, J. C. Svendsen, P. I. Macreadie, D. O. B. Jones, A. R. Boon, D. J. Booth, R. Brabant, E. Callahan, J. T. Claisse, T. G. Dahlgren, S. Degraer, Q. R. Dokken, A. B. Gill, D. G. Johns, R. J. Leewis, H. J. Lindeboom, O. Linden, R. May, A. J. Murk, G. Ottersen, D. M. Schroeder, S. M. Shastri, J. Teilmann, V. Todd, G. Van Hoey, J. Vanaverbeke, and J. W. P. Coolen. 2018. Environmental benefits of leaving offshore infrastructure in the ocean. Frontiers in Ecology and the Environment 16:571–578.

Franzese, P. P., E. Buonocore, L. Donnarumma, and G. F. Russo. 2017. Natural capital accounting in marine protected areas: The case of the Islands of Ventotene and S. Stefano (Central Italy). Ecological Modelling 360:290–299.

Furness R, Wade H, Masden E (2013) Assessing vulnerability of marine bird populations to offshore wind farms. Journal of Environmental Management 119: 56–66.

Gill, D. A., M. B. Mascia, G. N. Ahmadia, L. Glew, S. E. Lester, M. Barnes, I. Craigie, E. S. Darling, C. M. Free, J. Geldmann, S. Holst, O. P. Jensen, A. T. White, X. Basurto, L. Coad, R. D. Gates, G. Guannel, P. J. Mumby, H. Thomas, S. Whitmee, S. Woodley, and H. E. Fox. 2017. Capacity shortfalls hinder the performance of marine protected areas globally. Nature 543:665–669.
Glorius, S., J. Craeymeersch, T. Hammen van der, A. Rippen, J. Cuperus, B. Weide van der, J. Steenbergen, J., I. Tulp (2015). Effecten van garnalenvisserij in Natura 2000 gebieden. WMR rapport C013/15. 162 p.
Goñi, R., S. Adlerstein, D. Alvarez-Berastegui, A. Forcada, O. Reñones, G. Criquet, S. Polti, G. Cadiou, C. Valle, P. Lenfant, P. Bonhomme, A. Pérez-Ruzafa, J. L. Sánchez-Lizaso, J. A. García-Charton, G. Bernard, V. Stelzenmiiller, and S. Planes. 2008. Spillover from six western Mediterranean marine protected areas: Evidence from artisanal fisheries. Marine Ecology Progress Series 366:159–174.
Guidetti, P., and E. Sala. 2007. Community-wide effects of marine reserves in the Mediterranean Sea. Marine Ecology Progress Series 335:43–56.
Halkos, G., and G. K. Galani. 2012. The use of contingent valuation in assessing marine and coastal ecosystems’ water quality: A review. Unpublished; Munich Personal RePEc Archive.
Halpern, B.S. (2003) The impact of marine reserves: do reserves work and does reserve size matter? Ecological Applications 13(1): S117-S137.
Halpern, B. S., S. E. Lester, and J. B. Kellner. 2009. Spillover from marine reserves and the replenishment of fished stocks. Environmental Conservation 36:268–276.
Halpern, B. S., and R. R. Warner. 2002. Marine reserves have rapid and lasting effects. Ecology Letters 5:361–366.
Hastings, A., and L. W. Botsford. 1999. Equivalence in yield from marine reserves and traditional fisheries management. Science 284:1537–1538.
Hastings, A., and L. W. Botsford. 2006. Persistence of spatial populations depends on returning home. Proceedings of the National Academy of Sciences of the United States of America 103:6067–6072.
Heck, N., P. Dearden, and A. McDonald. 2011. Stakeholders’ expectations towards a proposed marine protected area: A multi-criteria analysis of MPA performance criteria. Ocean and Coastal Management 54:687–695.
HELCOM. 2007. HELCOM Baltic Sea Action Plan, HELCOM Ministerial Meeting Krakow, Poland.
HELCOM. 2016. Ecological coherence assessment of the Marine Protected Area network in the Baltic Sea. Retrieved from: https://www.helcom.fi/wp-content/uploads/2019/08/BSEP148.pdf.
HELCOM. n.d. About us. Retrieved from: https://helcom.fi/about-us/.
Hiddink, J. G., Jennings, S., Sciberras, M., Szostek, C. L., Hughes, K. M., Ellis, N., … Collie, J. S. (2017). Global analysis of depletion and recov-ery of seabed biota after bottom trawling disturbance. Proceedings of the National Academy of Sciences, 114, 8301–8306.
Hsieh, C., Reis, C., Hunter, J., Beddington, J., May, R.M., Sugihara, G. 2006. Fishing elevates variability in the abundance of exploited species. Nature, 443, pages859–862.
Huntington, H. P., R. Daniel, A. Hartsig, K. Harun, M. Heiman, R. Meehan, G. Noongwook, L. Pearson, M. Prior-Parks, M. Robards, and G. Stetson. 2015. Vessels, risks, and rules: Planning for safe shipping in Bering Strait. Marine Policy 51:119–127.
Hussain, S. S., A. Winrow-Giffin, D. Moran, L. A. Robinson, A. Fofana, O. A. L. Paramor, and C. L. J. Frid. 2010. An ex ante ecological economic assessment of the benefits arising from marine protected areas designation in the UK.

Ecological Economics 69:828–838.
IUCN-WCPA. 2008. Establishing Marine Protected Area Networks—Making It Happen. Washington, D.C.: IUCN-WCPA, National Oceanic and Atmospheric Administration and The Nature Conservancy. 118 p.
Jägerbrand, A. K., A. Brutemark, J. Barthel Svedén, and I. M. Gren. 2019. A review on the environmental impacts of shipping on aquatic and nearshore ecosystems. Science of the Total Environment 695:133637.
Jenkins, T. L., and J. R. Stevens. 2018. Assessing connectivity between MPAs: Selecting taxa and translating genetic data to inform policy. Marine Policy 94:165–173.
Jobstvogt, N., V. Watson, and J. O. Kenter. 2014. Looking below the surface: The cultural ecosystem service values of UK marine protected areas (MPAs). Ecosystem Services 10:97–110.
van Kooten, T., Deerenberg, C.M., Jak, R.G., van Hal, R. & Machiels, M.A.M. 2015. An exploration of potential effects on fisheries and exploited stocks of a network of marine protected areas in the North Sea. Report No. C093/14. IMARES, Wageningen, the Netherlands.
Laffoley D and Grimsditch G (Eds). 2009. The management of natural coastal carbon sinks. Gland, Switzerland: IUCN.
Leleu, K., B. Remy-Zephir, R. Grace, and M. J. Costello. 2012. Mapping habitats in a marine reserve showed how a 30-year trophic cascade altered ecosystem structure. Biological Conservation 155:193–201.
Leopold M.F., R.S.A. van Bemmelen & S.C.V. Geelhoed 2011. Zeevogels op de Noordzee. Achtergronddocument bij Natuurverkenning 2011. Wageningen, Wettelijke Onderzoekstaken Natuur & Milieu, WOt-werkdocument 257. 48 blz. 9 fig.; 2 tab.; 147 ref.
Lester, S. E., B. S. Halpern, K. Grorud-Colvert, J. Lubchenco, B. I. Ruttenberg, S. D. Gaines, S. Airamé, and R. R. Warner. 2009. Biological effects within no-take marine reserves: A global synthesis. Marine Ecology Progress Series 384:33–46.
Liefveld, W. M., K. Didderen, W. Lengkeek, M. Japink, S. Bouma, and M. M. Visser. 2011. Evaluating biodiversity of the North Sea using Eco-points. Testing the applicability for MSFD assessments.
Lindeboom, H. J., Witbaard, R., Bos, O. G., & Meesters, H. W. G. (2008). Gebiedsbescherming Noordzee. Habitattypen, instandhoudingsdoelen en beheersmaatregelen. Commissioned by PBL. WOt-Werkdocument, 114.
Luisetti, T., E. L. Jackson, and R. K. Turner. 2013. Valuing the European "coastal blue carbon" storage benefit. Marine Pollution Bulletin 71:101–106.
Macreadie, P. I., A. Anton, J. A. Raven, N. Beaumont, R. M. Connolly, D. A. Friess, J. J. Kelleway, H. Kennedy, T. Kuwae, P. S. Lavery, C. E. Lovelock, D. A. Smale, E. T. Apostolaki, T. B. Atwood, J. Baldock, T. S. Bianchi, G. L. Chmura, B. D. Eyre, J. W. Fourqurean, J. M. Hall-Spencer, M. Huxham, I. E. Hendriks, D. Krause-Jensen, D. Laffoley, T. Luisetti, N. Marbà, P. Masque, K. J. McGlathery, J. P. Megonigal, D. Murdiyarso, B. D. Russell, R. Santos, O. Serrano, B. R. Silliman, K. Watanabe, and C. M. Duarte. 2019. The future of Blue Carbon science. Nature Communications 10:1–13.
Mangi SC, Gall SC, Hattam C, Rees S, Rodwell LD. 2012. Lyme Bay – a case-study: measuring recovery of benthic species; assessing potential "spillover" effects and socio-economic changes; 3 years after the closure. Report 2: Assessing the socio-economic impacts resulting from the closure restrictions in Lyme Bay. Report to the Department of Environment, Food and Rural Affairs from the University of Plymouth-led consortium. Plymouth: University of Plymouth Enterprise Ltd. 96 pages.
Marine Scotland Science. 2016. Scottish Marine Protected Areas Socioeconomic

Monitoring 2016 Report. Retrieved from: https://www.gov.scot/publications/scottish-marine-protected-areas-socioeconomic-monitoring/.

McClanahan, T. R., and S. Mangi. 2000. Spillover of exploitable fishes from a marine park and its effect on the adjacent fishery. Ecological Applications 10:1792–1805.

McLeod E, Chmura GL, Bouillon S, et al. 2011. A blueprint for blue carbon: toward an improved understanding of the role of vegetated coastal habitats in sequestering CO2. Front Ecol Environ 9: 552–60.

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

McVittie, A., and D. Moran. 2010. Valuing the non-use benefits of marine conservation zones: An application to the UK Marine Bill. Ecological Economics 70:413–424.

MedPAN and SPA/RAC, 2019. The 2016 status of Marine Protected Areas in the Mediterranean. Retrieved from: http://d2ouvy59p0dg6k.cloudfront.net/downloads/medpan_forum_mpa_2016___brochure_a4_en_web_1_.pdf.

Merino, G., F. Maynou, and J. Boncoeur. 2009. Bioeconomic model for a three-zone marine protected area: A case study of Medes Islands (northwest Mediterranean). ICES Journal of Marine Science 66:147–154.

Micheli, F., A. Saenz-Arroyo, A. Greenley, L. Vazquez, J. A. Espinoza Montes, M. Rossetto, and G. A. de Leo. 2012. Evidence that marine reserves enhance resilience to climatic impacts. PLoS ONE 7.

Ministry of Infrastructure and the Environment & Ministry of Economic Affairs. 2012. Marine Strategy for the Netherlands part of the North Sea 2012-2020, Part 1. Retrieved from: https://www.noordzeeloket.nl/beleid/europese/achtergrond/documenten-mariene/.

Ministry of Infrastructure and the Environment & Ministry of Economic Affairs. 2015. Policy Document on the North Sea 2016-2021. Retrieved from: https://www.government.nl/documents/policy-notes/2015/12/15/policy-document-on-the-north-sea-2016-2021-printversie.

Moland, E., E. M. Olsen, H. Knutsen, P. Garrigou, S. H. Espeland, A. R. Kleiven, C. André, and J. A. Knutsen. 2013. Lobster and cod benefit from small-scale northern marine protected areas: Inference from an empirical before-after control-impact study. Proceedings of the Royal Society B: Biological Sciences 280:1–9.

Murawski, S. A., S. E. Wigley, M. J. Fogarty, P. J. Rago, and D. G. Mountain. 2005. Effort distribution and catch patterns adjacent to temperate MPAs. ICES Journal of Marine Science 62:1150–1167.

Nellemann, C., Corcoran, E., Duarte, C. M., Valdés, L., De Young, C., Fonseca, L., Grimsditch, G. (Eds). 2009. Blue Carbon. A Rapid Response Assessment. United Nations Environment Programme, GRID-Arendal, www.grida.no

Noordzeeloket. n.d.a. Military use. Retrieved from: https://www.noordzeeloket.nl/en/functions-and-use/militair-gebruik/.

Noordzeeloket. n.d.b. Noordzee beleidskaart. Retrieved from: https://www.noordzeeloket.nl/@167243/noordzee/.

OECD. 2015. The economics of marine protected areas. OECD publishing, Paris.

OECD. 2016. Marine Protected Areas Economics, management and effective policy mixes. OECD publishing, Paris.
Olsen EM, Johnson D, Weaver P, Goñi R, Ribeiro MC, Rabaut M, Macpherson E, Pelletier D, Fonseca L, Katsanevakis S, Zaharia T (2013). Achieving Ecologically Coherent MPA Networks in Europe: Science Needs and Priorities. Marine Board Position Paper 18. Larkin, KE and McDonough N (Eds.). European Marine Board, Ostend, Belgium.
ONE-Dyas B.V. n.d. GEMS. Retrieved from: https://onedyas.com/project/gems/.
OSPAR. 2018. Status Report on the OSPAR Network of Marine Protected Areas. Retrieved from: https://www.ospar.org/documents?v=40944.
OSPAR. 2020. The North-East Atlantic Environment Strategy. Retrieved from: https://www.ospar.org/convention/strategy.
OSPAR. n.d. Contracting Parties. Retrieved from: https://www.ospar.org/organisation/contracting-parties.
van Oostenbrugge, H., D. Slijkerman, K. G. Hamon, O. G. Bos, M. A. M. Machiels, O. van de Valk, N. T. Hintzen, E. Bos, J. T. van der Wal, and J. W. P. Coolen. 2015. Effects of seabed protection on the Frisian Front and Central Oyster Grounds. LEI report 2015-145.
Pauly, D., Christensen, V., Guenette, S., Pitcher, T.J., Sumaila, R., Walters, C.J., Watson, R., Zeller, D. 2002. Towards sustainability in world fisheries. Nature, 418(6898):689-95.
Pendleton, L., D. C. Donato, B. C. Murray, S. Crooks, W. A. Jenkins, S. Sifleet, C. Craft, J. W. Fourqurean, J. B. Kauffman, N. Marbà, P. Megonigal, E. Pidgeon, D. Herr, D. Gordon, and A. Baldera. 2012. Estimating Global "Blue Carbon" Emissions from Conversion and Degradation of Vegetated Coastal Ecosystems. PLoS ONE 7.
Plummer, M. L. 2009. Assessing benefit transfer for the valuation of ecosystem services. Frontiers in Ecology and the Environment 7:38–45.
Prellezo, R., P. Accadia, J. L. Andersen, B. S. Andersen, E. Buisman, A. Little, J. R. Nielsen, J. J. Poos, J. Powell, and C. Röckmann. 2012. A review of EU bio-economic models for fisheries: The value of a diversity of models. Marine Policy 36:423–431.
Rabaut, M., U. Braeckman, F. Hendrickx, M. Vincx, and S. Degraer. 2008. Experimental beam-trawling in Lanice conchilega reefs: Impact on the associated fauna. Fisheries Research 90:209–216.
RAC/SPA. n.d. The MAP. Retrieved from: https://www.rac-spa.org/map.
Rees, S. E., M. J. Attrill, M. C. Austen, S. C. Mangi, and L. D. Rodwell. 2013. A thematic cost-benefit analysis of a marine protected area. Journal of Environmental Management 114:476–485.
Rees, S. E., S. C. Mangi, C. Hattam, S. C. Gall, L. D. Rodwell, F. J. Peckett, and M. J. Attrill. 2015. The socio-economic effects of a Marine Protected Area on the ecosystem service of leisure and recreation. Marine Policy 62:144–152.
Rees, S. E., S. J. Pittman, N. Foster, O. Langmead, C. Griffiths, S. Fletcher, D. E. Johnson, and M. Attrill. 2018. Bridging the divide: Social–ecological coherence in Marine Protected Area network design. Aquatic Conservation: Marine and Freshwater Ecosystems 28:754–763.
Rijkswaterstaat. 2019. Kader Ecologie en Cumulatie 3.0 t.b.v. uitrol van windenergie op zee 2030 Deelrapport A: Methodebeschrijving
Rijkswaterstaat. n.d.a. Missie Rijkswaterstaat. Retrieved from: https://www.rijkswaterstaat.nl/over-ons/onze-organisatie/onze-missie/index.aspx.
Rijkswaterstaat. n.d.b. Beheer en ontwikkeling rijkswateren. Retrieved from: https://www.rijkswaterstaat.nl/water/waterbeheer/beheer-en-ontwikkeling-rijkswateren/index.aspx.

Rijkswaterstaat. n.d.c. Organisatiestructuur. Retrieved from: https://www.rijkswaterstaat.nl/over-ons/onze-organisatie/organisatiestructuur/index.aspx.

Rijkswaterstaat. n.d.d. Rijkswaterstaat Water, Verkeer en Leefomgeving. Retrieved from: https://www.rijkswaterstaat.nl/over-ons/onze-organisatie/organisatiestructuur/water-verkeer-en-leefomgeving/index.aspx.

Rijkswaterstaat. n.d.e. Over SEE. Retrieved from: https://www.rwseconomie.nl/over-see.

Roberts, C. M., and J. P. Hawkins. 2000. Fully-protected marine reserves : a guide. Page Usa Wwf.

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

Roberts, C. M., J. P. Hawkins, J. Fletcher, S. Hands, K. Raab, and S. Ward. 2010. Guidance on the size and spacing of marine protected areas in England. Environment Department, University of York, York:84.

Roberts, C. M., B. C. O'Leary, D. J. Mccauley, P. M. Cury, C. M. Duarte, J. Lubchenco, D. Pauly, A. Sáenz-Arroyo, U. R. Sumaila, R. W. Wilson, B. Worm, and J. C. Castilla. 2017. Marine reserves canmitigate and promote adaptation to climate change. Proceedings of the National Academy of Sciences of the United States of America 114:6167–6175.

Roncin, N., F. Alban, E. Charbonnel, R. Crec'hriou, R. de la Cruz Modino, J. M. Culioli, M. Dimech, R. Goñi, I. Guala, R. Higgins, E. Lavisse, L. Le Direach, B. Luna, C. Marcos, F. Maynou, J. Pascual, J. Person, P. Smith, B. Stobart, E. Szelianszky, C. Valle, S. Vaselli, and J. Boncoeur. 2008. Uses of ecosystem services provided by MPAs: How much do they impact the local economy? A southern Europe perspective. Journal for Nature Conservation 16:256–270.

Rossi, V., E. Ser-giacomi, M. Dubois, P. Monroy, and M. Hidalgo. 2014. Lagrangian Flow Networks : a new framework to study the multi-scale connectivity and the structural complexity of marine populations.

Russo, T., A. Parisi, G. Garofalo, M. Gristina, S. Cataudella, and F. Fiorentino. 2014. SMART: A spatially explicit bio-economic model for assessing and managing demersal fisheries, with an application to italian trawlers in the strait of sicily. PLoS ONE 9.

Saarikoski, H., J. Mustajoki, D. N. Barton, D. Geneletti, J. Langemeyer, E. Gomez-Baggethun, M. Marttunen, P. Antunes, H. Keune, and R. Santos. 2016. Multi-Criteria Decision Analysis and Cost-Benefit Analysis: Comparing alternative frameworks for integrated valuation of ecosystem services. Ecosystem Services 22:238–249.

Sas, H. Didderen, K., van der Have, T., Kamermans, P., van den Wijngaard, K., Reuchlin, E. 2019. Recommendations for flat oyster restoration in the North Sea.

Shears, N. T., R. V. Grace, N. R. Usmar, V. Kerr, and R. C. Babcock. 2006. Long-term trends in lobster populations in a partially protected vs. no-take Marine Park. Biological Conservation 132:222–231.

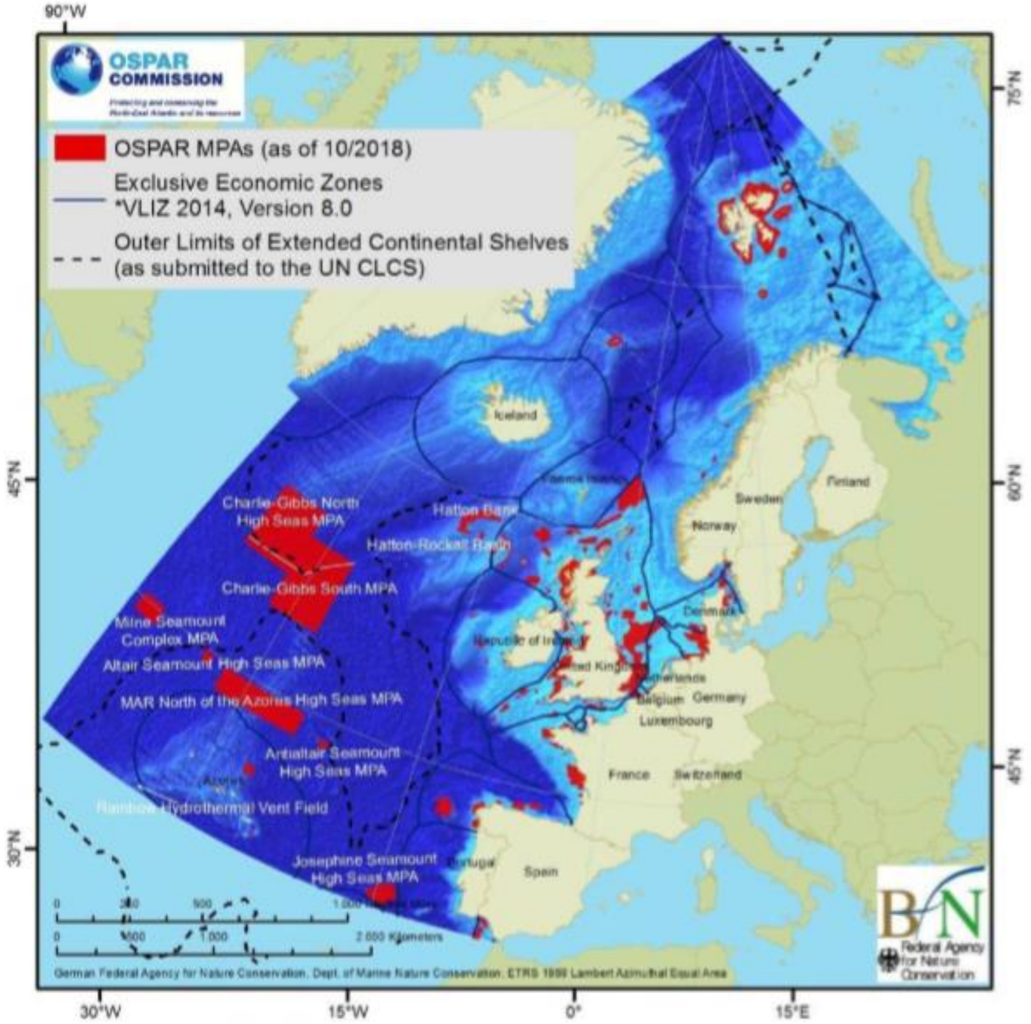
Sijtsma F.J., A.V. Hinsberg, S. Kruitwagen and F. Dietz , 2009, Natuureffecten in de MKBA's van projecten voor integrale gebiedsontwikkeling. PBL Netherlands Environmental Assessment Agency, the Netherlands.

Statistics Netherlands. (2016). Economic description of the Dutch North Sea and Coast: 2005, 2010, 2014. The Hague: Statistics Netherlands.

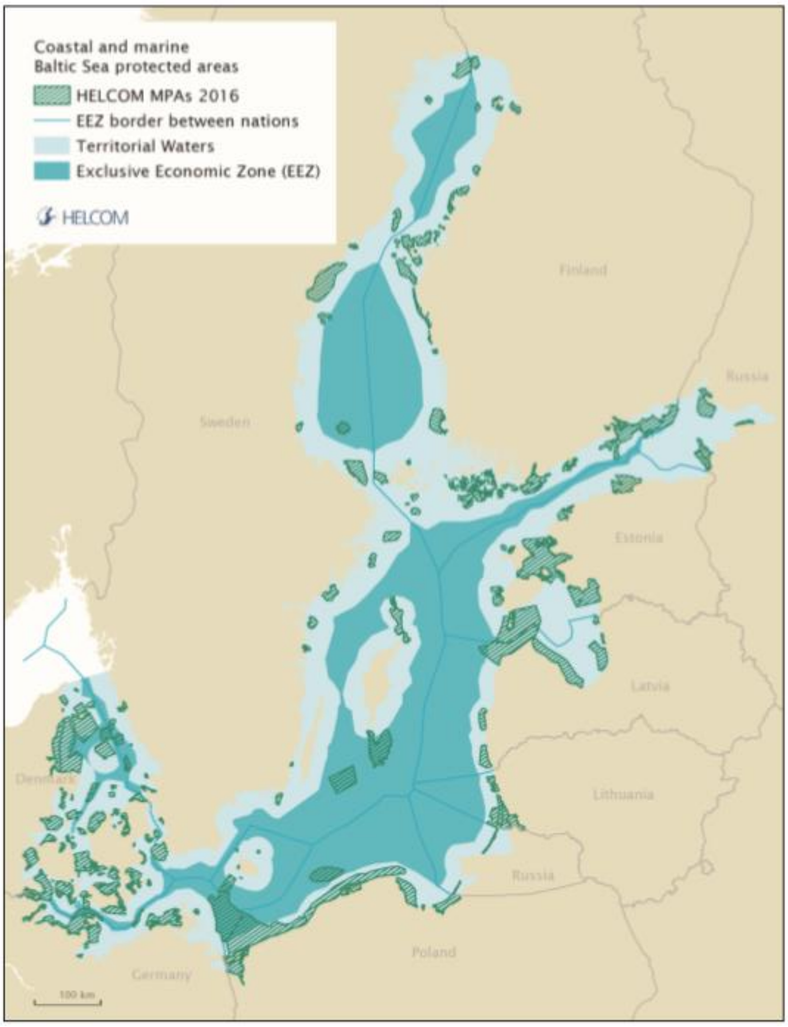
Steenbergen, J., Ulleweit, J., Machiels, M., Nijman, R., Panten, K., van Helmond, E. 2015. Discards Sampling of the Dutch and German Brown Shrimp Fisheries in

2009 – 2012. WMR rapport.
Stolwijk, H. (2004). Kunnen natuur- en landschapswaarden zinvol in euro's worden uitgedrukt? CPB memorandum. Nr. 5/2004/04; 20 juli 2004.
Tamis, J. ., C. . Karman, P. de Vries, R. . Jak, and C. Klok. 2011. Offshore olie- en gasactiviteiten en Natura 2000:149.
Teal L.R., 2011. The North Sea fish community: past, present and future. Background document for the 2011 National Nature Outlook. Wageningen, Wettelijke Onderzoekstaken Natuur & Milieu, WOtwerkdocument 256. 64 p. 27 Figs.; 8 Tabs.; 100 Refs.
The Benyon review panel. 2020. Into highly protected marine areas. Retrieved from: https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/890484/hpma-review-final-report.pdf.
This Fish. n.d. Shrimp by Trawl. Retrieved from: https://this.fish/fishery/shrimp-bottom-trawl-british-columbia/.
Thompson, K., K. Miller, P. Johnston, and D. Santillo. 2017. Storage of carbon by marine ecosystems and their contribution to climate change mitigation Greenpeace Research Laboratories Technical Report (Review) 03-2017.
Torres, C., and N. Hanley. 2016. Economic valuation of coastal and marine ecosystem services in the 21stcentury: an overview from a management perspective. Marine Policy 75:99–107.
Townsend, M., K. Davies, N. Hanley, J. E. Hewitt, C. J. Lundquist, and A. M. Lohrer. 2018. The challenge of implementing the marine ecosystem service concept. Frontiers in Marine Science 5:1–13.
UNEP. 1995. Special protocol concerning specially protected areas and biological diversity in the Mediterranean. Retrieved from: https://www.rac-spa.org/sites/default/files/protocole_aspdb/protocol_eng.pdf.
Vandeperre, F., R. M. Higgins, J. Sánchez-Meca, F. Maynou, R. Goñi, P. Martín-Sosa, A. Pérez-Ruzafa, P. Afonso, I. Bertocci, R. Crec'hriou, G. D'Anna, M. Dimech, C. Dorta, O. Esparza, J. M. Falcón, A. Forcada, I. Guala, L. Le Direach, C. Marcos, C. Ojeda-Martínez, C. Pipitone, P. J. Schembri, V. Stelzenmüller, B. Stobart, and R. S. Santos. 2011. Effects of no-take area size and age of marine protected areas on fisheries yields: A meta-analytical approach. Fish and Fisheries 12:412–426.
Vassallo, P., C. Paoli, A. Rovere, M. Montefalcone, C. Morri, and C. N. Bianchi. 2013. The value of the seagrass Posidonia oceanica: A natural capital assessment. Marine Pollution Bulletin 75:157–167.
Venkatachalam, L. 2004. The contingent valuation method: A review. Environmental Impact Assessment Review 24:89–124.
Virtanen, E. A., M. Viitasalo, J. Lappalainen, and A. Moilanen. 2018. Evaluation, gap analysis, and potential expansion of the Finnish Marine Protected Area network. Frontiers in Marine Science 9:1–19.
Vrooman, J., van Sluis, C., van Hest, F., 2018. Gebiedsbescherming op de Nederlandse Noordzee. De stand van zaken in relatie tot visserij. Stichting De Noordzee, Utrecht
Wageningen University Research. 2019. Scholaanvoer in 2018 sterk afgenomen; garnalenvangsten bijna verdubbeld. retrieved from: https://www.agrimatie.nl/ThemaResultaat.aspx?subpubID=2232&themaID=2857#:~:text=kg.,kg%20garnalen%20werden%20gevangen.
WWF. 2000. Planning for Representative Marine Protected Areas: A Framework for Canada's Oceans. Report prepared for World Wildlife Fund Canada, Toronto
WWF. 2009. Silent Invasion – The spread of marine invasive species via ships' ballast water. WWF International, Gland.

Annex 1 OSPAR MPAs in the North East Atlantic



Annex 2 HELCOM MPAs in the Baltic Sea



Annex 3 IUCN categories of protected areas

IA Strict nature reserve	Strictly protected for biodiversity and also possibly geological/ geomorphological features, where human visitation, use and impacts are controlled and limited to ensure protection of the conservation values.
IB Wilderness area	Usually large unmodified or slightly modified areas, retaining their natural character and influence, without permanent or significant human habitation, protected and managed to preserve their natural condition.
II National park	Large natural or near-natural areas protecting large-scale ecological processes with characteristic species and ecosystems, which also have environmentally and culturally compatible spiritual, scientific, educational, recreational and visitor opportunities.
III Natural monument or feature	Areas set aside to protect a specific natural monument, which can be a landform, sea mount, marine cavern, geological feature such as a cave, or a living feature such as an ancient grove.
IV Habitat/species management area	Areas to protect particular species or habitats, where management reflects this priority. Many will need regular, active interventions to meet the needs of particular species or habitats, but this is not a requirement of the category.
V Protected landscape or seascape	Where the interaction of people and nature over time has produced a distinct character with significant ecological, biological, cultural and scenic value: and where safeguarding the integrity of this interaction is vital to protecting and sustaining the area and its associated nature conservation and other values.
VI Protected areas with sustainable use of natural resources	Areas which conserve ecosystems, together with associated cultural values and traditional natural resource management systems. Generally large, mainly in a natural condition, with a proportion under sustainable natural resource management and where low-level non-industrial natural resource use compatible with nature conservation is seen as one of the main aims.

Annex 4 Connectivity assessment method of HELCOM