Optimal management of marine resources:
Spatial planning of multiple uses by multiple actors

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This research was conducted under the auspices of the SENSE graduate school

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Thesis

submitted in the fulfilment of the requirements for the degree of doctor
at Wageningen University
by the authority of the Rector Magnificus
Prof. dr. M.J. Kropff
in the presence of the
Thesis Committee appointed by the Academic Board
to be defended in public
on Friday 7 October 2011
at 1.30 p.m. in the Aula

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Optimal management of marine resources: Spatial planning of multiple uses by multiple actors.
xii + 168 pages.

Thesis, Wageningen University, Wageningen, NL, (2011)
With summaries in Dutch and English

ISBN 978-94-6173-026-8

La Mer, A bercé mon cœur pour la vie
From: La mer, Charles Trenet (1913-2001)

Contents

Dankwoord

Het ondernemen van een promotietraject is als het maken van een lange reis per schip, waarbij de bestemming nog verre van bekend is, maar waar je vooral inscheept vanwege de reis, en niet vanwege de bestemming.

Hoewel wellicht cliché, lijkt me dat er geen betere metafoor is dan deze, aangezien mijn proefschrift over de zee gaat. Nu de reis dan is volbracht en het proefschrift er ligt, is het tijd om een aantal mensen hartelijk te bedanken, omdat ze mij op (een deel van) deze reis hebben begeleid.

Allereerst de mensen die mij op mijn reis over de woelige academische baren hebben gegidst: mijn promotoren Ekko van Ierland en Jan Stel. Heel erg bedankt voor de steun en begeleiding de afgelopen vier jaar bij het bedenken en schrijven van mijn proefschrift. Zoals dit dankwoord waarschijnlijk laat zien, ben ik nog steeds lang van stof, maar het is dankzij jullie al een stuk beter dan het was. Ook mijn co-promotor Hans-Peter Weikard verdient een warme dank voor alle prettige discussies, modelbesprekingen en natuurlijk het overhalen om een PhD milieu-economie te gaan doen. Tenslotte ook mijn dank aan de andere co-auteurs in de afgelopen vier jaar: Rolf Groeneveld en Arjan Ruijs.

Naast de gidsen wil ik alle medereizigers bedanken, die hun eigen reis aan het maken waren en zijn, maar met wie ik lange tijd ben meegereisd. Kamergenoten Kelly, Erik & Michael bedankt voor de fijne tijd op kamer 1110 met leuke gesprekken, ervaringen en natuurlijk het tafeltennissen. Ook bedankt aan de mede-reizigers op andere kamers: Karianne, Diana & Thomas. Also thanks to my foreign co-travellers on the PhD journey with whom I shared a room: Petronella, Xavier and Huong, and those with whom I shared experiences but not rooms: Enoch, Irene, Joseph, Loan, Ma, Miyuki, Morteza, Yonky, and Vo. I really had a good time at ENR and our little PhD drinks and parties contributed greatly to that.

During my travels I also travelled with other colleagues that had already completed their PhD journey, but were travelling along in the ENR journey. Justus, thanks for getting me into environmental economics, along with Hans-Peter, and for allowing me to continue my journeys in this field. Other colleagues with whom I had a great time as well as conversations filled with genuine laughter also deserve a warm thank you: Edwin, El Hadji, David, Demet, Jeroen, Geerte, Rob, Petra, Silke, Thomas, Wiepke & Xueqin.

Verder natuurlijk mijn dank aan de dames, die ervoor zorgden dat de schepen vlot over de oceaan konden varen. Wil, Sjoukje en Gré hartelijk dank voor alle administratieve ondersteuning, maar natuurlijk ook voor de sociale ondersteuning met fijne koffie- en theepauzes.

Tijdens de reis kun je niet 24 uur bezig zijn met navigeren en varen. Het is dan ook heerlijk dat er zoveel zorgzame en vriendelijke mensen zijn, die voor je klaar staan, met wie je veel lol kunt hebben en die afleiding brengen waar dat nodig is. Dus vrienden Frank & Frank, bands de RABS en Ontzetting, de Heeren van de spellenklup, en dispuut Noiam: ontzettend bedankt.

Loek & Marjo, Marije, Pepijn en Roosmarijn (en Johannes, Nellie en Jeroen) natuurlijk bedankt voor onze warme en fijne familie. Het is goed om zo af en toe in Burgum met z'n allen een thuishaven te hebben. In Dordrecht, Arnhem en Davos zorgen ook Erika en Gerard, met mijn nieuwe(re) zusjes Esther en Christine voor fijne vaste aanlegplekken.

Tenslotte natuurlijk mijn eigen thuishaven waar ik altijd weer met plezier thuiskom. Mirjam en Sam, wat is het fantastisch dat jullie er zijn. Laten we nog lang samen (boot)tochtjes maken en samen thuiskomen.

Maarten

Chapter 1: Introduction

1.1. Background

"The sea is common to all, because it is so limitless that it cannot become a possession of any one, and because it is adapted for the use of all, whether we consider it from the point of view of navigation or of fisheries."(Grotius, 1609).

Although, even today, the surface of the moon is better known and mapped than the ocean floor, the oceans are no longer considered as limitless as in the days of Grotius. Human activities such as fisheries and pollution are now clearly taking their toll in ocean space. According to the Food and Agriculture Organisation (2009) many fish stocks are fully exploited or in decline, and we are dealing with increasing pollution both in air and water (ICES, 2003), while economic activities in ocean space are rapidly expanding.

One of the main reasons why we are so rapidly depleting the ocean's resources and polluting ocean space, has already been pointed out by Aristotle in 350 BC: "For that which is common to the greatest number has the least care bestowed upon it." (Aristotle, 350 BC). This argument was popularized some 2000 years later as "the Tragedy of the Commons" by Hardin (1968). A more formal economic analysis was introduced by Gordon (1954) and Scott (1955); they argue that open access to a resource leads to depletion, but under private property a resource will be well managed.

In 1960, the UN recognized that measures had to be taken to manage ocean space and its resources. A series of conferences lead, in 1982, to the new Law of the Sea as codified in the United Nations Conference on the Law of the Sea III, also known as UNCLOS III. It came into force on 16 November 1994, and has currently been signed by 161 countries (UN, 2010).

By UNCLOS III, at least the following zones occur in the marine part of states: The territorial zone (12 nautical miles, nm) which fully falls under national jurisdiction. Then comes the Exclusive Economic Zone (EEZ). Its border is determined as a 200 miles zone from the coastline or by the edge of the continental shelf. In this zone states have the exclusive right to exploit the resources in a sustainable way. Beyond the national EEZs lie the High Seas, comprising 38% of ocean space (VLIZ, 2011). Here the notion of the Common Heritage of Mankind is applicable (Borgese, 1998; Stel and Loorbach, 2003). In practice however, the open access approach as pleaded for by Grotius still persists for the High Seas of ocean space.

It would be naïve, however, to think that with the assignment of property rights or rights to exploit resources in a sustainable way, as in the EEZs, all problems would simply be solved. After all, ecosystems are not governed by human boundaries, they are open and highly dynamic systems. This results in two main problems: The first problem is that we are often dealing with transboundary common goods as in the case of shared fish stocks, and public goods issues as in the case of mitigation of transboundary pollution. The shared characteristic of these issues is that in both cases other countries cannot be excluded from enjoying the benefits. In addition for public goods such as mitigation of transboundary pollution, the benefit is non-rivalrous. Both characteristics lead to free-riding and under-provision of these goods from a global point of view.

The second problem is the coordination and planning of economic activities within the EEZ of a single country. An increasing number of economic activities takes place, or are being planned in EEZs. Some of these activities may be incompatible with each other, whereas others may be perfectly compatible. Policy making for the separate activities, however, often takes place on a sectoral basis, with little regard for other activities, resulting in inefficient and suboptimal policy outcomes. Fishing, for example, is still unrestricted in Natura 2000 areas in the Dutch North Sea, although quota exist on the total catch.

1.2. Objective and research questions

Ocean space is used for a large number of economic activities, and new human activities are being proposed. To use the marine environment in a sustainable way, we have to manage these human activities. In this way we can continue to enjoy the goods and services provided by the ocean system sustainably, without damaging the system irreversibly.

Several instruments can be applied for the management of human activities in ocean space, but of course each instrument has its strengths and weaknesses. These depend on both the activity we wish to manage and on the state of the ocean system. Moreover, the advantages and disadvantages are related to the scale of the analysis and the problem at hand. A good instrument at country level may perform similar, better or worse at inter-country level. The analysis of these two broad problems forms the basis of this thesis. The aim of the thesis is therefore:

To contribute to the optimal management of marine resources, by developing models for optimal spatial planning of offshore wind farms and by developing models to investigate the economic incentives associated with the planning of Marine Protected Areas both in EEZs and the High Seas .

In the thesis I have chosen to study three activities in the ocean space, one new and two traditional ones, being: offshore wind farms, fisheries and nature conservation. As instruments I consider marine spatial planning and a specific tool within marine spatial planning: Marine Protected Areas (MPAs). Effort restrictions are also used in some of the chapters. More specifically the following questions are addressed in this thesis:

1. How can spatial planning of new uses of ocean space improve the ecosystem management in the marine environment?
2. How does the multiple use nature of Marine Protected Areas (MPAs) affect the incentives of countries to assign such areas?
3. How does the perceived uniqueness of species and their distribution over ecosystems owned by different countries affect the MPA assignment of countries?
4. How does the assignment of MPAs in the High Seas influence the formation of Regional Fisheries Management Organizations?

1.3. Methodology

To address the questions above I use both constrained optimization techniques and game theory. While considering these questions we move through different scale levels. Question 1 is a problem at the local scale, i.e. the EEZ of a single country. Questions 2 and 3 involve the EEZs of several countries. Question 4 is defined at the scale of the High Sea. As a consequence, it concerns all countries of the world, through the international negotiations in the context of the UN.

In Chapter 2 I address research question one. Spatial planning is interpreted as a constrained optimization problem. Consequently we formulate a constrained optimization model where a central planner's perspective is applied to plan economic activities. The objective of the planner is to maximize society's welfare, by generating an optimal spatial plan of economic activities. Planning economic activities, however, is constrained by both spatial economic and ecological restrictions. Restrictions in space are incompatible other economic activities and differing costs and benefits of locations. Ecological restrictions are restrictions to protect habitats or species. A number of scenarios and associated plans are generated by applying different restrictions, e.g. a minimum number of birds protected, or disallowing wind farms in ecologically sensitive areas. These plans are then evaluated on their merits.

For studying research questions two to four I use game theory. Game theory is a mathematical method to analyze strategic interactions among agents. Agents choose a strategy from their possible set of strategies, such that they maximize their payoffs given the behavior of other agents. Because MPAs constitute, at least partly, public goods and fish stocks constitute a common pool resource, strategic interaction between countries is likely to occur. Therefore game theory is an appropriate method to analyze such situations.

Chapter 3 deals with research question two. The chapter investigates the incentives of countries to assign an MPA, when they account for the fisheries benefits only, for the biodiversity conservation benefits only or for both. I devise a game theoretic model in which countries can cooperate or free-ride on each other's contribution, and derive the optimal MPA size under the fisheries only, conservation only or the combined scenario.

Chapter 4 addresses research question three. I devise a two player game in which countries decide to allocate MPAs in ecosystems that are available to them. Species are distributed over the ecosystems, but a number of species occur in multiple ecosystems, creating overlap between ecosystems. I investigate the MPA size chosen in each ecosystem, if countries cooperate, behave strategically, or ignore the contributions in biodiversity protection by the other country.

Chapter 5 deals with the fourth research question. I investigate how the assignment of an MPA influences the stability of Regional Fisheries Management Organisations (RFMOs) through adapting an existing RFMO formation game such that MPAs can be accommodated. I then study how the introduction of such an MPA influences the potential stability of the formed coalitions.

1.4. Institutional setting and scale

1.4.1. Background

When looking at the management of ocean space, one inevitably runs into issues of scale. The reason is that at different scales different problems occur, because different rules and regulations apply. In this thesis I look at various scale levels and move from local problems, involving one country, to regional problems involving a few countries, to international problems involving a large number of countries. The increase in complexity due to larger numbers of countries involved necessitates an increased abstraction of reality to keep the models and results traceable and meaningful.

The laws and policy at the various levels are framed by the Law of the Sea Legislation (UNCLOS III). UNCLOS III constitutes international law in the marine domain. It defines among others, the territorial sea, EEZ and High Seas. In this thesis I will ignore the difference between the EEZ and the territorial sea, because the studied problems that apply at the EEZ level apply equally at the territorial level[1].

In the next section I describe the institutional setting at the different scale

[1]The last version of the Law of the Sea and its amendments, as well as the current signatories can be found at: http://www.un.org/Depts/los/index.htm. For a more thorough review of UNCLOS III see e.g. Churchill and Lowe (1999)

levels. I define the local scale as the EEZ of a single country. The problems addressed at this scale comprise coordination and optimization problems within one country's EEZ. Issues such as: "How can we arrive at an optimal spatial configuration for multiple users?" are addressed at this level. As an example we will describe the institutional setting in which the Netherlands operates in the North Sea, as it clearly illustrates the setting in which a typical Western state operates.

I define the regional scale at the level of what is called a 'regional sea', i.e. a sea that is fully claimed by EEZs, such as the Baltic and North Sea. Problems addressed at this level comprise coordination problems among countries such as: "How can we manage shared fish stocks best?" "What is the influence of multiple uses on the optimal allocation of fishing effort and nature conservation if countries cooperate, free-ride or ignore each other?" Here we will scale up the Dutch example to the EU level.

Finally, I define the international level as the High Seas. These areas are governed by international law, through the International Seabed Authority and Regional Fisheries Management Organisations. In the High Seas the notion of Common Heritage of Mankind is applied. Their management is complicated by the fact that international law is hard to enforce, and governance at this level is carried out by means of voluntary international agreements. The problems addressed at this level comprise typical international environmental agreement problems, such as: "How to build a coalition of countries that manage the fish stocks of the sea in a sustainable way?"

1.4.2. Local level

The local scale is defined here as the EEZ of a single country; in our case the Dutch EEZ in the North Sea. At the local scale the Dutch have access to a part of the North Sea (including a part of the Wadden Sea). This part can be divided into the territorial waters where the Netherlands has full jurisdiction, and the EEZ where it has the rights to exploit the resources sustainably. From an economic perspective, however, both are the same, because with regard to the economic exploitation the same set of rules and regulations apply[2].

The Dutch North Sea is governed by a plethora of rules and regulations, depending upon the administrative sector. Fisheries, nature, oil and gas extraction and energy generation at the North Sea are the responsibility of the

[2]The administrative situation described here is the one that exists since 2010. Before 2010 a number of ministries existed that have now merged, making the situation before 2010 even more complicated. Most notably for the North Sea: the Ministry of Agriculture, Nature and Food Quality and the Ministry of Economic Affairs merged into the new 'Ministry of Economic Affairs, Agriculture and Innovation'. The Ministry of Transport, Public Works and Water Management and the Ministry of Housing, Spatial Planning and the Environment merged into the 'Ministry of Infrastructure and the Environment'.

Ministry of Economic Affairs, Agriculture and Innovation. In addition both fisheries and nature conservation are subject to European policies, most notably the Bird and Habitats Directive for nature and the European Common Fisheries Policy for fisheries. Furthermore the Netherlands is a signatory to a number of international environmental agreements obliging it to reduce pollution. These agreements include the OSPAR convention to reduce pollution from dumping, land-based pollution and non-polluting activities that adversely affect the seas.

Extraction of oil, gas and minerals as well as energy generation are also the responsibility of the Ministry of Economic Affairs, Agriculture and Innovation. The ministry grants licenses for the extraction of these resources in specific areas. Companies can apply for these licenses. If approved, they can, subject to further regulations, extract resources or build offshore wind parks. The extraction of aggregates, such as sand and gravel, is the responsibility of another ministry: the Ministry of Infrastructure and the Environment. The procedure for aggregates is similar to that of other non-renewable resources, i.e. companies apply for licenses with which they are granted to extract aggregates at certain locations, subject to further regulations.

Navigation at the North sea is also governed by the Ministry of Infrastructure and the Environment, but is further subject to international regulations through UNCLOS III which defines the freedom of navigation, the International Convention for the Safety of Life at Sea, which lies out navigational rules and ships routing, and the International Convention for the Prevention of Pollution from Ships and the Bonn agreement which try to reduce pollution from shipping.

Concerning the further (spatial) planning of the North Sea, the Ministry of Infrastructure and the Environment is also responsible for the law on spatial planning. In practice it states that the government should come up with a spatial vision for the North Sea. This has been realized with the Integral Management Plan North Sea 2015, which was formulated by an inter-ministerial committee, IDON. It contains an integral plan concerning the health, economics and spatial planning of the North Sea (IDON, 2005). Furthermore the European Union has formulated the Marine Strategy Directive, obliging each country to formulate a Marine Strategy for its EEZ. This strategy should be an integrated spatial plan that follows a holistic ecosystem-based approach (European Commission, 2008). This view was re-affirmed in the Maritime Policy of the EU where the Marine Strategy Directive is seen as its environmental pillar (European Commission, 2006; van Hoof and van Tatenhove, 2009). Both the policy and the directive are the responsibility of the Ministry of Infrastructure and the Environment.

1.4.3. Regional level

The Dutch EEZ in the North Sea borders with the EEZs of the United Kingdom, Belgium and Germany. With these countries, the Netherlands consequently has to deal when it comes to transboundary issues. At this level most activities are governed by European Directives and policy as well as international law. The relevant European legislation is the Marine Strategy Directive, the Bird and Habitats Directive, and the Water Framework Directive. The main policies are the Common Fisheries Policy and the Maritime Policy.

The Marine Strategy Directive obliges all countries of the EU to formulate a marine strategy. Such a strategy should be an integrated strategy that should "culminate in the execution of programmes of measures designed to achieve or maintain good environmental status" and "while being specific to its own waters, reflects the overall perspective of the marine region or subregion concerned" (European Commission, 2008, preamble). Although the Directive specifies that countries should cooperate, both with EU members and non-EU members it leaves the details to the countries involved in the formulation of the strategies. The Bird and Habitats Directive and Water Framework Directive suffers from similar problems. Both, in principle, oblige countries to designate a network of protected areas (including Marine Protected Areas) and define river basin management plans, but both do not explicitly specify the details (European Commission, 1992, 2000). In practice the transformation of these directives into legislation by individual countries leaves room for different interpretations (van Hoof and van Tatenhove, 2009) which consequently can evolve into transboundary issues in e.g. spatial planning and the planning of Marine Protected Areas in particular.

As mentioned in the previous section, countries like the Netherlands have the rights to exploit the resources in its EEZ, but in a sustainable way. However, some fish stocks are shared with the aforementioned countries and consequently the exploitation of fish stock has been regulated with the Common Fisheries Policy of the EU. The ministers responsible for the fisheries at a national level agree annually upon quota for most stocks. As such the total allowable catch is regulated among EU countries. The EU also bargains with non-EU-members such as Norway, Russia and the Faro Islands over transboundary stocks (EU, 2010). Because the setting of quota is the outcome of a bargaining process and because the EU as a whole has to bargain with non-members, some of the transboundary shared stock problems remain.

The Integrated Maritime Policy is not yet in force. It aims to be an encompassing policy that reaches across sectors, integrating the CFP and using the Marine Strategy Directive as its environmental pillar. It recognizes the need for ecosystem based planning at the regional scale, but also acknowledges that the marine economic sectors should be strengthened and that planning should be across sectors, incorporating the views of all stakeholders (European

Commission, 2006). As pointed out by van Hoof and van Tatenhove (2009), the main challenge of the Maritime Policy will be to realize this integration of stakeholders across sectors and countries.

In order to gather the required data necessary for these kind of policies the EU has started to install the Global Monitoring for the Environment and Security system (GMES), a system that uses local and European data and information systems to collect long term environmental data on the oceans (Ryder and Stel, 2003).

1.4.4. High Sea level

In Europe, the Dutch North Sea does not border with any High Seas, but the Netherlands is partner to the several international treaties governing this part of ocean space. As such it is bound to rules and regulations that have been put into international law.

The main piece of legislation governing economic activities in the High Seas is the United Nations Convention on the Law of the Sea (UNCLOS III) (UN, 1982) and its amendments: the Agreement relating to the implementation of Part XI of the United Nations Convention on the Law of the Sea of 10 December 1982 (UN, 1994) and The United Nations Agreement for the Implementation of the Provisions of the United Nations Convention on the Law of the Sea of 10 December 1982 relating to the Conservation and Management of Straddling Fish Stocks and Highly Migratory Fish Stocks (UN, 1995). These conventions together provide the framework in which resources can be extracted from the High Seas.

Concerning fisheries these agreements state that fishing in the High Seas should be regulated by Regional Fisheries Management Organizations (RFMOs). Note that “Regional” in Regional Fisheries Management Organisations does not refer to ‘regional seas’ but to regions in the world oceans. Thus RFMOs govern e.g. the North-East Atlantic region or Western and Central Pacific region. The membership of RFMOs is open and consequently any country in the world can join.

These RFMOs set quota for fisheries in their area. Moreover, fishing by nations should be sustainable and based upon the precautionary principle. Fishing nations should have due regard for the measures taken by coastal states to protect their fish stocks, and all nations should try to reach cooperative agreements in case of disputes (UN, 1982, 1995).

Concerning the extraction of other resources such as minerals or oil and gas, these have been stated to be the Common Heritage of Mankind. The returns of extracting these resources should benefit mankind as a whole. For this purpose and for further regulations the UN have installed the Seabed Authority, that grants licenses, takes care of the division of benefits and should promote transfer of technology (UN, 1982, 1994). These rules of course only apply to countries that have signed up to the conventions. UNCLOS has been signed by 161 countries of the world, but the two other conventions were signed by considerably fewer countries. Interestingly, the USA, which has one of the largest EEZs in the world, has not ratified UNCLOS III, but has ratified the Agreement on Migratory Fish Stocks (UN, 2010).

Another important treaty, that nearly all countries have signed is the Convention on Biological Diversity (CBD), and more specifically the CBD of 2006. In this convention the parties to the Convention have agreed to conserve at least 10% of the seas, generally interpreted as MPAs. This decision was reaffirmed in the conference in 2010 (CBD, 2010). Obviously, this does not mean that 10% of the High Seas should be protected, but given that the High Seas constitute 38% of the world seas (VLIZ, 2011) a one on one translation would mean that at least 26% of the High Seas should be protected. Currently, however, the total area covered by MPAs is only 1.17% of ocean space (Spalding et al., 2010), so the targets are far from being reached.

Pollution is mainly regulated through the MARPOL conventions, but the older annexes hardly apply to the High Seas as they are based on a distance to land approach, that does allow for dumping in the High Seas (Ardron et al., 2008). The newer annexes in contrast have not been ratified by all countries and thus cannot completely mitigate the pollution they are intended for (IMO, 2010).

1.5. Emergence of new sea users and new management instruments

1.5.1. Offshore wind farms and marine spatial planning

One of the oldest use of the ocean space is fisheries. Man has been fishing since prehistoric times, first by picking up shells and small creatures from the beaches and gradually moving into other ways of fishing, such as angling, and fishing from boats. Navigation is a use of ocean space that may be even older. Man reached Australia approximately 40.000 years ago by boat. Navigation is intimately linked with trade. Traders have been exploring the world seas in search of new wares and markets since ancient history. Extraction of non-renewable resources in contrast has a long tradition for some resources and less for others. Salt extraction goes back to ancient times, but aggregate extraction (sand and gravel) and oil and gas extraction have only begun in the second half of the last century. Oil exploration in the North Sea for example started in the 1960s (Olsgard and Gray, 1995). Similarly, most aggregates were originally mined on land but since the 1940's aggregate extraction has also been carried out at sea (Smith, 2000).

Thus ocean space has become a lot busier over the ages and is becoming busier still. One of the most recently emerged activities on the seas is the extraction of wave and wind energy as renewable energy supply. Our increasing demand for energy, and renewable energy in particular, has led us to look at the seas as a potential source of energy. The steady and strong winds on the seas offer a large energy potential for offshore wind parks. Wind power has been used for centuries, but was never very interesting for the generation of electricity. Nowadays the increase in turbine size combined with technological progress makes onshore wind electricity a feasible economic option. Offshore wind farms are generally not economically feasible yet, but may become feasible if positive social and ecological effects of offshore wind farms and negative social and ecological effects of fossil fuels are accounted for (Snyder and Kaiser, 2009; OECD, 2010).

Within the ecological effects the main negative effect comprises mortality of animals, particularly birds (Exo et al., 2003; Drewitt and Langston 2006). Other effects include habitat alteration and disturbance by noise and electromagnetic fields (Elliott, 2002; Petersen and Malm, 2006). Positive effects may occur as well. Offshore wind farms are functioning as artificial reefs attracting fish and offering new habitat types. They can also act as de facto marine reserves if fishing is prohibited (Wilhelmsson et al., 2006; Fayram and de Risi, 2007). Socially some gains are expected from removing the wind farms from the direct view, mitigating the "Not In My Back Yard" effect but whether this really applies is questionable (Firestone and Kempton, 2007; Haggett, 2008).

From the economic perspective the offshore environment offers a few other advantages and disadvantages: on the one hand strong, steady winds increase the potential energy generation and less turbulent winds and a reduction in wind shear reduce equipment costs. On the other hand the harsh environment, the foundations and the cabling increase the costs of offshore wind farms (Henderson et al., 2002; Mathew, 2006).

Most, if not all, of these effects have a spatial component. Obviously local wind speeds, depth and distance from the shore are important revenue and cost considerations when planning offshore wind farms. Similarly both positive and negative ecological effects differ with location as some environments are more vulnerable than others or are more susceptible to improvement by offshore wind farms. It is therefore hardly surprising that the emergence of offshore wind farms as a new activity was one of the main drivers behind the start of the Marine Spatial Planning processes in several European countries (Douvere et al., 2007).

Marine Spatial Planning is a holistic approach that applies ecosystem-based sea use management to allocate parts of ocean space to specific uses in order to achieve optimal economic, ecological and social objectives. Because it is cross-sector it can be used to identify synergies and conflicts between different uses (Douvere et al., 2007; Douvere, 2008). Further advantages of Marine Spatial Planning include a more rational site selection for development and conservation, more efficient use of marine resources, and a more strategic and proactive framework for decision making (Gilliland and Laffoley, 2008).

The need for spatial considerations in offshore wind farms has been recognized sector-wise and several models exist taking spatial economic differences into consideration (e.g. Kooijman et al., 2001). Spatial ecological aspects have also been researched (e.g. Garthe and Hüppop, 2004, van der Wal et al., 2006) but for Marine Spatial Planning a more integrated model is necessary. Such a model would take other economic activities into account as well as the spatial ecological effect. The main social effects can be addressed by generating several scenarios with such a model by, and together with, stakeholders. The scenarios generated by the different stakeholders can then form the basis for discussion among stakeholders and policymakers. In Chapter 2 such a model is designed and applied to the Dutch North Sea.

1.5.2. Multiple use Marine Protected Areas at the regional level

As mentioned in the previous section the designation of offshore wind farms was one of the main drivers behind the start of MSP in Europe. Another main driver was nature conservation, especially the habitat and birds directives (Douvere et al., 2007; Douvere, 2008). These directives require the designation of protected areas, including Marine Protected Areas (European Commission, 1992). As a result Marine Protected Areas became an important instrument within MSP in Europe, and as such they are included in the Marine Strategy Directive (European Commission, 2008) and mentioned in the Maritime Policy (European Commission, 2006).

An essential tool within MSP are therefore Marine Protected Areas (MPAs). The interpretation of the term “protected” basically defines what the goal of such areas is e.g. fisheries management, nature conservation, tourism, or something else. The use of MPAs as a fisheries management tool has been criticized as inefficient, especially under open access (Hannesson, 1998; Anderson, 2002) and overly optimistic if fishermen behavior is ignored (Smith and Wilen, 2004), but also promoted as hedge against uncertainty (Lauck et al., 1998; Sumaila, 2002; Sumaila et al., 2007) and useful if a habitat-effect occurs (Schnier, 2005a,b; Armstrong, 2007; Armstrong and Falk-Petersen, 2008). For conservation, its effects have generally been shown to be positive (Halpern and Warner, 2002; Lubchenco et al., 2003; Halpern et al., 2010) although they obviously cannot protect species from more mobile threats such as pollution and oil spills.

One of the reasons that MPAs are gaining momentum in MSP and marine conservation is exactly because they can serve different goals at the same time and a number of authors have investigated possible synergies such as between nature conservation and tourism (Brown et al., 2000; Boncoeur et al., 2002; de

Groot and Bush, 2010; Thur, 2010), nature conservation and fisheries management (Tundi Agardy, 1994; Gell and Roberts, 2003; Meester et al., 2004) and nature conservation and scientific research (Lindeboom, 1995). All of these studies, however, focus on the local scale, whereas some of these benefits e.g. nature conservation and fisheries benefits, are public goods, i.e. their benefits are non-excludible at the regional level whereas the costs are borne by single countries.

The non-excludability of benefits of MPAs at the regional scale has also been recognized, but always for single uses, usually fisheries. Ruijs and Janmaat (2007) investigate the location of an MPA in a transboundary fishery and how it is affected by fish migration and non-cooperative behavior. Sumaila (2002) studies how MPAs can serve as a hedge against shocks in fish growth in a transboundary fisheries. The public good aspect of MPAs with regard to nature conservation has received little attention, but Busch (2008) has derived a number of general conditions for transboundary parks to be superior over isolated parks, such that countries will assign transboundary parks instead of isolated ones.

The fact that MPAs serve multiple goals may at the regional scale affect incentives of countries to assign MPAs, depending on which uses of the MPA they account for. In Chapter 3 I investigate how the incentives of countries change depending whether they account for the fisheries benefits, the nature conservation benefits or both.

1.5.3. Marine Protected Areas configurations at the regional level

As argued in the previous section MPAs are an important instrument for marine conservation and MSP, but at the regional scale they suffer from free-riding problems. Even if all uses are accounted for as investigated in Chapter 3, free-riding problems persist. In that Chapter, however, I only investigate the size of MPAs and ignore the spatial aspect of MPAs.

The importance of the spatial configuration of MPAs has been emphasized by several authors, both with regards to the fisheries and nature conservation. In fisheries for example, Costello and Kaffine (2010) show that MPAs are created endogenously when spatial territorial user rights are used in the fisheries. Sanchirico (2004) studies the importance of connectivity and how it affects which patches in a meta-population are to be designated as MPA. Sanchirico and Wilen (2001) identify ecological structures necessary for MPAs to increase both harvest and stock. Smith and Wilen (2003, 2004) show the importance of taking the spatial effort redistribution of fishermen into account when designing MPAs.

Spatial configuration has attracted similar attention in marine nature conservation. The literature on these subjects comprises mainly reserve site selection problems. Meester et al. (2004) identify MPA configurations for a given number of MPA sites that meet multiple targets concerning species representation, fisheries effort and reserve shape. Ball and Possingham (2000) designed MARXAN, a program that calculates MPA configurations given specific minimum representation targets of species, while minimizing the costs of the full reserve, or the boundary length of the full reserve. Game et al. (2008) investigate spatial configurations to minimize the probability of losing biodiversity in reserves through a catastrophe.

In contrast to fishermen, the influence of spatial configuration on the economic incentives of countries has received relatively little attention. In the fisheries literature Ruijs and Janmaat (2007) address the issue, although they only consider a single MPA. In marine conservation the problem has hardly been studied so far, but some work has been done in the terrestrial domain (e.g. Rodrigues and Gaston, 2002 and Jantke and Schneider, 2010). These papers, however, only consider cost-effectiveness in a single country versus cost-effectiveness under full cooperation and ignore strategic incentives.

In Chapter 4 I explore the incentives and how the MPA configuration is influenced by the distribution of species over ecosystems. I also point out the important differences in configuration between the situation where countries ignore the contribution to protection by other countries and the situation where free-riding on these contributions occurs. The former situation is similar to the approach by Rodrigues and Gaston (2002) and Jantke and Schneider (2010), the latter is the approach usually taken by economists when studying public goods.

1.5.4. Marine Protected Areas at the High Sea level

In the previous sections I have outlined the importance of MPAs and the incentives associated with these MPAs at the regional sea level. MPAs, however, are also increasingly called for as an instrument for management of the High Seas. MPAs are seen as the way forward in the High Seas, especially to protect vulnerable habitats, such as sea mounts and as a management tool for High Sea fisheries (Sumaila et al., 2007; North-East Atlantic Fisheries Commission, 2009; IUCN, 2010). Several proposals have been made to designate at least 10% of the High Seas as MPA (Sumaila et al., 2007).

The High Sea is governed by other mechanisms and institutions then regional seas and as a consequence the incentives, and ways to assign MPAs, are also very different from the regional seas. Because no property rights exist in the High Seas, MPAs cannot be assigned by a country; they have to be assigned through an international agreement. Some partial MPAs have been designated on the High Seas, e.g. by the North-East Atlantic Fisheries Commission that outlawed trawling in certain areas (North-East Atlantic

Fisheries Commission, 2009).

The only way to ensure that an MPA is indeed a protected area, i.e. an area where certain activities such as fishing are outlawed, is by unanimous agreement, because if a country does not agree and permits these activities the MPA is no longer a protected area. Therefore an MPA agreement must be acceptable and beneficial to all potential users of the area where the MPA is to be installed.

Regional Fisheries Management Organisations (RFMOs) regulate the High Sea fisheries in certain parts of the High Seas. Although countries are obliged by UNCLOS III to join the appropriate RFMO if they wish to fish in the area managed by the RFMO, they cannot be excluded if they fish there without joining the RFMO. Another way to fish outside the RFMO membership would be to choose to resign from UNCLOS. When fishing without joining the appropriate RFMO countries are said to be involved in Unregulated Fishing. It can be beneficial to a country not to join an RFMO and engage in unregulated fishing, because in this way the country is not bound to Total Allowable Catches, and it can profit from the effort reductions by the RFMO members.

The problem then, to get countries to join an RFMO can be thought of as an international environmental agreement where a coalition of countries forms an RFMO, whereas the other countries free-ride on the effort reductions by this RFMO. In recent literature (e.g. Pintassilgo et al., 2010) this has been analyzed as a coalition formation game, that studied the stability of coalitions to reduce fishing effort.

If MPAs would be assigned in the High Seas, they would change effort decisions. Since effort decisions are in part dictated by RFMO membership the MPAs may also change RFMO membership. In Chapter 5 I use the fisheries model of Chapter 3 to check the influence of MPAs on this coalition formation.

1.6. Reading guide

The remaining chapters of this thesis answer the research questions phrased in section 1.2 and the last chapter draws some overall conclusions. The scale level of analysis of the research questions moves in the same direction as the description of the institutional setting in section 1.4. Moreover the research questions run parallel to the description of new users and management instruments in section 1.5.

Chapter 2 analyzes the local scale and the new sea user offshore wind farms as well as Marine Spatial Planning of this activity in the ocean space. A spatial optimization model is formulated and applied to the Dutch North Sea. A number of scenarios with different spatial ecological restrictions are explored.

Chapter 3 analyzes the strategic incentives associated with the assignment of MPAs when they have multiple uses. With the help of a game theoretic model I study the incentives for cooperation and free-riding when accounting for fisheries, conservation or both.

In Chapter 4 I study how the distribution of species over ecosystems and how different ways of accounting for the contributions of others affects the MPA assignment in ecosystems. I develop a game theoretic model that incorporates the distribution of species over ecosystems into the decisions on MPA size in different ecosystems. Three different regimes are considered: full cooperation, free-riding and conservation autarky, i.e. ignoring the contribution by others.

In Chapter 5 I study the effect of MPAs in the High Seas on the formation of Regional Fisheries Management Organisations. I adapt the fisheries model from Chapter 3 to a High Sea setting and combine it with a game theoretic model to analyze the stability of coalitions.

Chapter 2*: Spatial planning of offshore wind farms: a windfall to marine environmental protection?

2.1. *Introduction*

Wind energy is one of the current major candidates for renewable energy generation. Compared to fossil fuels it has the advantage that it is CO2 neutral when generating energy. In fact greenhouse gas emissions only take place during the construction, maintenance and decommissioning phases (Lenzen and Munksgaard, 2002). Moreover, if one accounts for the subsidies to other energy sources, and takes current carbon credit prices as a proxy for the damage costs of carbon emissions, wind energy is competitive with regular power sources (The Economist, 2008).

Total wind power installed in the EU by the end of 2007 was 56,535 MW, or 3.7 % of its total energy demand (European Wind Energy Association, 2008). This amount is significantly higher then the target set for 2010 by the EU, which was 40,000 MW (European Commission, 1997).

This success story has a flip side: wind farms often meet local resistance. Considerations such as equity, fairness and landscape intrusion lay at the basis of such resistance (Christensen and Lund, 1998; Wolsink, 2000; Ek, 2005; Wolsink, 2007). Moreover, turbines cause noise, shadow flickering, electromagnetic fields and disturbance of animals and habitat, by causing collisions with birds and bats, and acting as barriers against migration and foraging (Burton et al., 2001; Mathew, 2006; van der Wal et al., 2006). Because these effects vary strongly with location the spatial dimension is pivotal in tradeoffs between wind energy and its environmental effects.

Wind parks are located offshore to avoid landscape intrusion and noise. The offshore environment has other advantages as well, such as stronger and steadier winds and large continuous areas, enabling the establishment of large wind farms. Offshore winds are less turbulent, thus decreasing the fatigue load and increasing the lifetime of the project. Finally the reduced occurrence of wind shear allows shorter towers (Henderson et al., 2002; Mathew, 2006). Disadvantages of offshore wind farms are higher investment costs for foundation, the distance to the main electrical grid, and improved equipment needed because of the harsh environment, which causes quick corrosion and makes maintenance difficult (Henderson et al., 2002; Mathew, 2006).

Ecological effects of locating wind farms offshore can be both detrimental and beneficial. Wind farms negatively affect the marine environment through

*This chapter is based on the paper: Maarten J. Punt, Rolf. A. Groeneveld, Ekko C. van Ierland, Jan H. Stel, 2009. Spatial planning of offshore wind farms: a windfall to marine environmental protection? *Ecological Economics* **69**(1): 93-103.

avian collisions (Exo et al., 2003; Drewitt and Langston, 2006), underwater noise (Koschinski et al., 2003; Wahlberg and Westerberg, 2005; Thomsen et al., 2006) and electromagnetic fields (Gill, 2005; Petersen and Malm, 2006; Öhman et al., 2007). There are positive effects too on local biodiversity as the turbines can act as artificial reefs and no-take zones, and there is evidence of spill-over effects (Petersen and Malm, 2006; Wilhelmsson et al., 2006; Fayram and de Risi, 2007). The impacts on biodiversity and ecosystem functions should therefore be considered in location choice.

Wind farms yield the highest net revenues if located in areas with high wind speeds (which increases energy generation) and areas with low average seafloor depths at closest proximity to the shore (both of which diminish costs).

In this chapter we analyze the problem of finding the optimal location for offshore wind farms, by considering both economic and ecological aspects. We develop and use a spatially explicit model that includes energy generation as well as the effects on bird and fish species. The model maximizes the revenues from wind farms under constraints for ecological impacts related to bird collisions and impacts on fish stocks.

Few of the large number of economic and ecological models of spatial planning of offshore wind parks have integrated economic and ecological considerations in one framework. The model of Kooijman et al. (2001) calculates costs based on an engineering model and a GIS module that covers the North Sea. Elkinton et al. (2005) focus on the layout of the offshore wind parks. Planning systems for businesses have been developed by Resoft (2008), Garrad Hassan (2008), EMD (2008) and BMT renewables (2008). All of these models calculate costs, but none of them consider the effects on species or ecosystems explicitly, although the models of Garrad Hassan, Resoft and EMD include modules for landscape intrusion. Ecological models focus mainly on the effects of wind farms on birds. These include sensitivity maps by Garthe and Hüppop (2004), a turbine specific collision risk model (Tucker, 1996) and a spatial planning model (van der Wal et al., 2006). Eliott (2002) has formulated a conceptual model but does not quantify relationships. Moreover, none of these models consider economic choices explicitly.

The contribution of this chapter is that it considers both spatial economic choices and local ecological effects. We present a modeling framework that spatially allocates offshore wind farms, taking into consideration spatial variations in wind speed, distance to the shore and sea floor depth, and presence and dispersal of bird and fish populations. We demonstrate how the model can be applied to the Dutch EEZ, where concerns of renewable energy, biodiversity and fisheries rank high on the political agenda. The aim of the chapter is to illustrate some of the choices faced in spatial planning of offshore wind parks as well as the potential of the model in analyzing this problem.

The chapter is organised as follows: the next section presents a more detailed analysis of offshore wind farms and their effects. It continues with the formulation of the model and the illustration of how it can be applied to the Dutch EEZ. Finally, discussion and conclusions are presented.

2.2. *Integrated assessment of offshore wind farms*

Earlier conceptual integrated assessment models of offshore wind energy (e.g. Elliott, 2002) were based on the Drivers-Pressures-States-Impacts-Responses (DPSIR) framework. Another conceptual model, the Scene model, has been suggested by Rotmans (1998), and constitutes the inventory of social, economic and natural stocks and their relations.

We use the framework of Hein (2005) to describe the ecosystem and economic system and their relationships in a more explicit way then the DPSIR framework (Figure 2.1). It constitutes on the one hand the economic system, and its underlying processes: consumption and production and on the other hand the ecosystem that is driven by ecological processes. Several flows between both systems exist: the economic system pollutes and intervenes in the ecosystem whereas the ecosystem delivers ecosystem goods and services to the economic system. Interactions between systems influence the state of both, and systems may change over time as indicated by the time direction in Figure 2.1. New states will show different relationships within and between the economic system and the ecosystem.

To build a conceptual model of the economic and ecological aspects of wind farms we first identify the drivers in both systems. Next we analyze the

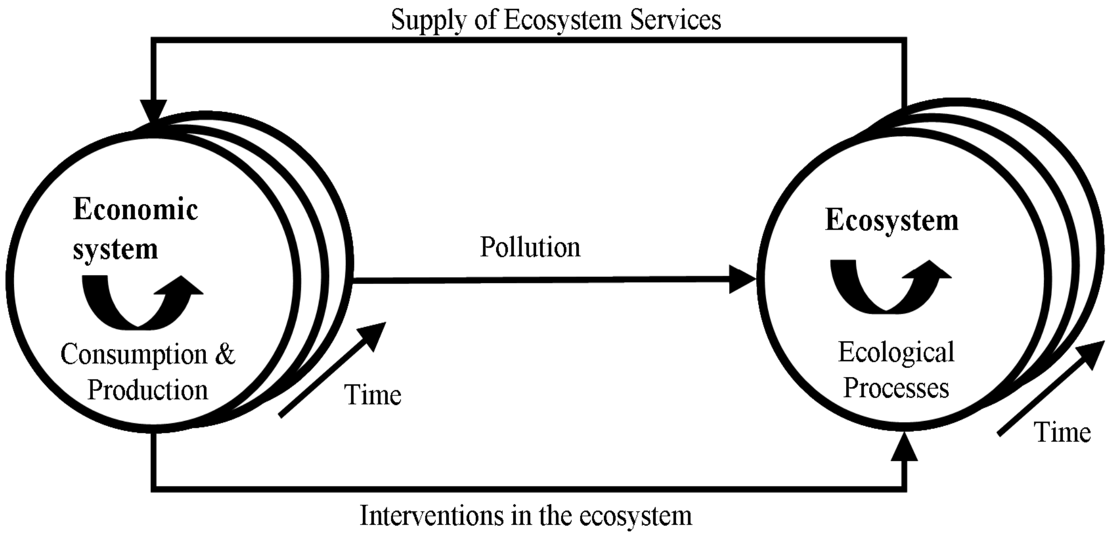


Figure 2.1: The interactions between the economic system and the ecosystem (cf. Hein, 2005)

feedbacks between both systems, first the affected ecosystem services, then the pollution effects and finally the interventions in the ecosystem.

The driving factor in the economic system that underlies the planning and implementation of offshore wind farms is the consumption of energy. Some of the energy consumption may be supplied by wind farms, depending on the costs. OECD and IEA (2005) estimate that the generation costs of electricity with conventional plants (coal, gas, nuclear) range between 15.7 and 60.4 US$(2003)/MWh., whereas wind power ranges between 31.1 and 94.3 US$(2003)/MWh. Note that these prices exclude technology specific subsidies and corporate taxes.

When building a wind park the net economic benefits are dependent upon the amount of electricity that can be sold and the costs, which mainly consist of the investment costs such as the costs of the turbines and infrastructure, and the variable costs that mainly constitute operation and maintenance costs (O&M costs). These costs differ between locations depending upon turbine types, wind speed, sea floor depth at the site and distance from the shore (Kooijman et al., 2001; Noord et al., 2004; Mathew, 2006).

The marine ecosystem is mostly driven by ecological processes such as photosynthesis, recruitment, mortality, predation and decomposition. The ecosystem services provided depend on these processes and the resulting species stocks. The goods and services provided by marine ecosystems have been inventoried by Beaumont et al. (2007). The goods and services affected by wind farms are raw materials, food provision, cultural services, option use services and biologically mediated habitat.

Pollution by offshore wind parks consists of emissions during construction, maintenance and decommissioning, sediment plumes and drill cuttings due to foundation works and cabling, and underwater noise (Elliott, 2002; Gill, 2005; Petersen and Malm, 2006). Total CO_2 emission ranges from 8.1 to 123.7 g CO_2/ kWh over the lifetime of a project (Lenzen and Munksgaard, 2002). The newest fossil fuel energy plants by comparison emit between 344 and 846 g CO_2/kWh (Metz et al., 2005). To our knowledge no studies yet exist that investigate the effects of sediment plumes and drill cuttings caused by wind farms but de Groot (1996) and Breuer (2004) have studied the effects for sand extraction and oil exploration on the local environment. Both smother habitats and the drill cuttings can cause chemical poisoning, depending on which materials are used with the drilling. Most of these effects are local and the benthos recovers reasonably quickly.

More permanent than the above mentioned forms of pollution are underwater noise and the electromagnetic fields that are caused by the cables. Although the cables are well isolated, the formation of electromagnetic fields cannot be prevented completely (Gill, 2005; Öhman et al., 2007). The evidence on the impacts of cables is inconclusive but the impacts seem to be minimal

(Petersen and Malm, 2006; Öhman et al., 2007). The same applies for underwater sound; it leads to avoidance of wind farms areas, masking of communication signals and hearing loss for various marine species. Wahlberg and Westerberg (2005) find that cod and Atlantic salmon detect wind farm noise from distances of 7-13 km and 0.4-0.5 km respectively, and may change their migration routes accordingly. These figures, however, are highly uncertain due to complicating factors such as reflection, transmission speed, wind speed and water density. For porpoises and seals the detection distances are 40 and 360 meters, respectively. Porpoises experience the noise, but are not disturbed whereas seals avoid the noise (Koschinski et al., 2003).

The interventions in the ecosystem consist of physical barriers that are formed by the wind parks for migration and foraging routes of birds, and possibly fish species and marine mammals. The direct effects on birds have been researched extensively, both on and offshore (see e.g. Hiscock et al., 2002, Exo et al., 2003, Garthe and Hüppop, 2004, Drewitt and Langston, 2006 and Madders and Whitfield, 2006). The most important effects are collision, avoidance and changed migration routes. The number of collisions of birds with turbines depends on location, bird species and weather (Exo et al., 2003; Garthe and Hüppop, 2004; Drewitt and Langston, 2006). Drewitt and Langston (2006) reviewed collision literature and found a large range: 0.01 to 23 birds killed per turbine per year.

Furthermore the presence of turbines may also influence the local hydrography of a place through the concrete foundations and the cabling. Previously available habitat is destroyed by the foundations and cabling, and the local habitat is changed to hard substrate. Recolonization may occur depending on the neighborhood (Elliott, 2002; Hiscock et al., 2002; Gill, 2005; Petersen and Malm, 2006; van der Wal et al., 2006).

A more positive intervention in the ecosystem is the creation of an artificial reef, and if fishing is prohibited in the area, a no-take zone. Wind farms have been shown to act as artificial reefs that increase numbers of fish and other benthos species (Petersen and Malm, 2006; Wilhelmsson et al., 2006; Fayram and de Risi, 2007).

2.3. Model setup

2.3.1. Introduction

We focus our modeling efforts on the most important economic and environmental effects of wind farms. We use a supply side model that assumes that all energy generated can be sold at a fixed price. Costs and revenues are calculated on a one-year basis. We include fish and birds as indicator species. The effects of wind farms on species are modeled based on the impacts on habitat quality and dispersal options. In the proposed model, for simplicity, we do not consider the sound and electromagnetic field disturbances of fish.

The model consists of both an economic and an ecological system. In both systems variables and parameters vary with the location under consideration. The model maximizes the revenues from the sale of wind generated electricity, considering ecological and planning restrictions, such as minimum numbers of species and sites reserved for shipping or marine conservation. This section explains the mathematical structure of the model.

2.3.2. The economic system

The model maximizes the revenues of generating wind energy in a particular area, under a set of ecological and policy constraints. Formally[1]:

$$\begin{aligned} &\max R_{tot} = \max \sum_i R\{(T_1 .... T_i)\} = \sum_i ((p e_i - c_i) T_i) \\ &\text{s.t.} \quad 0 \le T_i \le T_i^{\max} \qquad \forall i \end{aligned} \tag{2.1}$$

where $R_i(T_i)$ represents the revenues (in thousand of 2007 Euro) of wind energy generation in cell i as a function of the number of turbines in cell i (T_i). $T_i^{\max}$ is the maximum number of turbines in a cell determined by the spatial, design and ecological constraints, such as available area, space requirements for the separate turbines and policy measures. The revenues in each cell i are calculated as the energy produced (in Megawatt hour) in cell i multiplied with the price of energy p (in thousand of 2007 Euro per MWh) minus the total costs of energy production (in thousand of 2007 Euro) in that cell.

Energy production, E_i, is a function of the potential energy production per turbine and the number of turbines in that cell:

$$E_i = e_i \times T_i \quad \forall i \tag{2.2}$$

where e_i is the energy potential of a turbine in cell i. The energy production E_i is generated at a cost C_i. The cost function has the following form:

$$C_i = c_i \times T_i \quad \forall i \tag{2.3}$$

where c_i represents the total cost per wind turbine. Total costs (c_i) are equal to the amortized investment costs (f_i) and operation & maintenance costs (v_i). The costs vary with the power of the turbine, the average seafloor depth in the cell and the distance to the nearest grid connection. Total revenues are thus determined by the energy potential and the local costs in each cell. In the absence of further constraints the model would fill each cell where placing

[1]To distinguish between parameters and variables we use capitals for variables and lower case for parameters. Indices are noted as subscripts.

turbines is profitable, with the maximum number of turbines. This concludes the economic part.

2.3.3. The ecosystem

Ecological effects of offshore wind farms are described by a spatially explicit model that includes spatial variation in abundance of birds and fish, as well as dispersal of individuals.

Many spatial ecological models are based upon the theory of island biogeography of MacArthur and Wilson (1967) or the metapopulation theory of Levins (1969) or reserve selection models as the one first derived by Kirkpatrick (1983). These models have been extended to model both terrestrial reserves and species (e.g. Pulliam et al., 1992, Day and Possingham, 1995, and Moilanen and Cabeza, 2002) and marine reserves and species (e.g. Ball and Possingham, 2000, Beattie et al., 2002, and Sanchirico, 2004).

Reserve selection models use algorithms such as simulated annealing and branch-and-bound, and are specifically concerned with the selection of reserves such that either a given numbers of species (typically 50-500) is preserved under minimal costs, or that as many species as possible are conserved under a given budget. Their focus on species richness makes this approach unsuitable for our model. Metapopulation models use relatively few patches with subpopulations and directed movement. As the modeled environment is considered to be very open and the population evenly spread out, a model of random dispersal movement combined with local habitat quality, such as the one by Hof and Bevers (1998) may describe the environment better. In this chapter we use the version as modified by Groeneveld (2004). The model is a steady state model that shows the final distribution of a population as an end result of processes such as birth, death and migration. Such models are often used in ecology and to a lesser extent in economics (Hanski, 1994; Hof and Bevers, 1998, 2000; Moilanen and Hanski, 1998; Groeneveld et al., 2005; Polasky et al., 2005, 2008; Groeneveld and Weikard, 2006). The temporal dimension is contracted in such models as the main interest is the final spatial distribution and survival and not so much the path towards the equilibrium.

The original model assumes random dispersal, implying that the probability that a species occurs in a cell is a function of the connectivity to adjacent suitable cells. The original non-linear formula to calculate this probability as given by Hof and Bevers (1998) is:

$$P_i = 1 - \left[\prod \left(1 - r_{ij} H_j \right) \right] \quad \forall i \tag{2.4}$$

with r_{ij} the probability that cell i is connected to cell j, and $H_j \in \{0,1\}$ the habitat suitability of cell j. Hof and Bevers approach (2.4) with a set of linear equations:

$$P_{i,s} \leq \sum_j \left(d_s \cdot H\left(T_j\right)_{j,s} \right) \quad \forall i,s \text{ with } j \text{ adjacent to } i,\ i \neq j \tag{2.5}$$

$$P_{i,s} \leq H\left(T_i\right)_{i,s} \quad \forall i,s \tag{2.6}$$

where s denotes either birds or fish and in which parameter d_s is chosen to approach equation (2.3) as closely as possible (for details see: Hof and Bevers, 1998). We follow the modification of Groeneveld (2004) and make the domain of H_i continuous over the interval: [0,1]. $P_{i,s}$ represents the probability that an individual of a species reaches that cell and survives. Since the model is a linear maximization problem at least one of equations (2.5) and (2.6) is satisfied as an equality. Hence $P_{i,s}$ is equal to the minimum of the two RHSs of equations (2.5) and (2.6), i.e. either the habitat suitability of cell *i,* which may have increased or decreased after the introduction of the turbines, or to a weighted sum of the habitat suitability of its neighboring cells, depending on which of the two is less.

The habitat suitability of a cell is measured on scale of 0-1, with 1 being ideal and 0 being unsuitable. The original habitat suitability of cell *i* (before the introduction of the turbines) lies somewhere between zero and one and is modified by the number of turbines. The final $H(T_i)_{i,s}$ is a function of the number of turbines in that cell (T_i) but the relationship between $H(T_i)_{i,s}$ and T_i differs between species. Birds are bothered by the turbines, so that each additional turbine makes that cell less attractive to birds. For fish we assume that the positive artificial reef working and the absence of fisheries more than offset possible negative effects of electromagnetic fields and noise. Examples of suitability functions are shown in Figure 2.2.

As $P_{i,s}$ is the probability of an individual reaching cell *i*, multiplying the potential maximum number of individuals with this probability gives the number of survivors. The total number of individuals of a species *s* in cell *i*, $N_{i,s}$, is therefore calculated as:

$$N_{i,s} = P_{i,s} \times n_{i,s} \quad \forall i,s \tag{2.7}$$

with $n_{i,s}$ the potential maximum number of birds and fish in that cell.

2.3.4. The integrated system

Both systems are integrated into one model, with the number of turbines T_i as a link between both systems. T_i is the basic decision variable of the model as it determines costs and revenues in the economic side of the model and habitat suitability in the ecological part of the model (see Figure 2.2). In turn, habitat suitability affects P_i through (2.5) and (2.6) and hence the number of birds and fish in a cell. In this way the model integrates the economic and ecological side

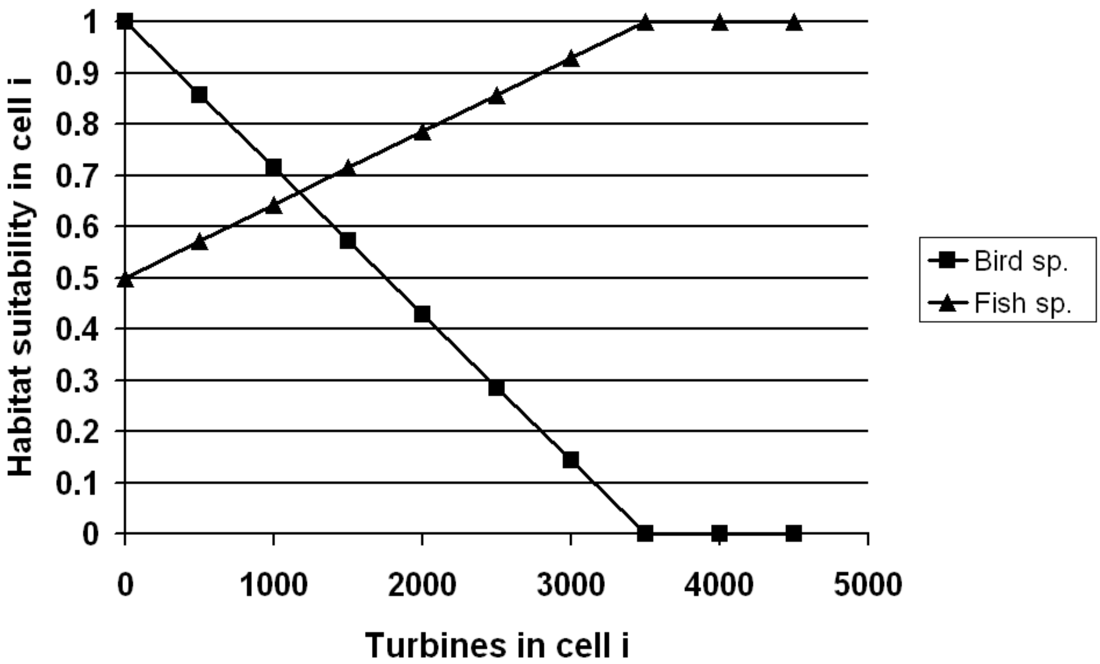


Figure 2.2: Hypothetical habitat suitability for a bird and fish species as a function of turbines in cell i (T_i) (Intercept values without turbines: 1 and 0.5, slopes -2.86*10-4 and 1.43*10-4 for bird and fish species respectively)

of the problem. The model maximizes profits under the restriction that minimum numbers of birds and fish are maintained:

$$N_s = \sum_i N(T_i)_{i,s} \geq N_s^{\min} \quad \forall s \tag{2.8}$$

Apart from the ecological restrictions, no further feedbacks from the ecosystem to the economic system exist, and the model is a constrained optimization problem.

2.4. Demonstration for the Dutch Exclusive Economic Zone in the North Sea

2.4.1. Introduction

We now demonstrate the model for the Dutch Exclusive Economic Zone (EEZ) in the North Sea, in order to show its potentials. The results are meant to be indicative of the capabilities of the model, rather than providing an actual policy advice.

The North Sea is a shallow sea (90 m. on average) that is surrounded by several industrialized countries. It is very rich in biodiversity with 1500 species of nematodes, 700 taxa of macrofauna in the benthos, 224 species of fish, 28 species of birds, 2 seal species, harbour porpoises, 2 dolphin species and a whale species (Ducrotoy et al., 2000). It is also one of the most intensively used seas, surrounded by countries with strongly developed maritime sectors (see Table 2.1).

Offshore wind farms are a relatively new activity in the North Sea. So far Denmark has seven offshore wind farms in operation and two under construction, the UK has seven in operation and seven under construction, and the Netherlands have two wind parks (European Wind Energy Association, 2008).

Parameter values of the Dutch North Sea used in the model are shown in Table 2.2. The North Sea atlas (www.noordzeeatlas.nl) provides spatial information on the average seafloor depth and densities of several bird and fish species in the EEZ. Although these densities are not the maximum densities of birds and fish that could occur in the North Sea, they can be taken as a general measure of the number of individuals present before the implementations of wind farms.

We use the ICES squares divided by four as cell unit for the grid in the model. This means that each cell is 33 x 28 km. This is a compromise between the spatial scale of the fish data (ICES squares) and bird data (5 x 5 km). The cells are shown in Figure 2.3. To mitigate edge effects on the population models we assume that the edge cells in the Dutch EEZ are connected to their neighbors in other EEZs and that these have ideal habitat.

The bird density in number/m^2 was converted in absolute numbers by averaging the density in one cell and multiplying with its area. The fish catch per unit of effort (CPUE) was taken as relative biomass measure and converted to absolute numbers by assuming that total spawning stock biomass (as estimated by ICES) of the species in the Dutch EEZ was directly proportional to the area of the Dutch EEZ in the total North Sea. The CPUE was then normalized to one and multiplied with the total spawning stock biomass in the

Table 2.1: Economic impact of maritime sectors in selected North Sea countries

Countries	Economic impact (% of GDP)	Turnover (€bln)	Direct employment	Indirect employment
Denmark	11.5	12.4	70,100	-
Netherlands	3.7	14.7	137,000	56,0000
Norway	20.0	21.4	192,000	-
UK	3.5-4.9	23.7	250,000	173,000

Source: Marine Institute (2006)

Table 2.2: Basic parameter values in the model for the Dutch EEZ.

Parameter	Value	Source
Price of energy (p)	0.0825 thousand €/MWh	Eurostat
Energy potential per turbine (e_i)[1]	8.8 - 12.7 GWh/annum	Windspeeds: ECN (2004). Turbine technical specifications: Mathew (2006).
Amortized investment costs (f_i)[2]	300 - 500 thousand €/ annum	Turbine prices: Kooijman (2002) as cited in Noord et al. (2004) and compensated for inflation. Modifying percentages for infrastructure: Mathew (2006)
Operation & Maintenance costs (v_i)[2]	100 - 200 thousand €/ annum	Kooijman (2002) as cited in Noord et al. (2004) Modifying percentages for O&M: Mathew (2006)
Total costs (c_i=f_i+v_i)	400 - 700 thousand €/ annum	
Connectivity between cells (d)	0.3 (corresponding r_{ij} = 0.5)	Hof and Bevers (1998)
Original number of fish in cell i (s_i)	$40*10^3$ - $900*10^3$	Noordzee atlas (www.noordzeeatlas.nl)
Original number of birds in cell i (b_i)	50 – 12000	Noordzee atlas (www.noordzeeatlas.nl)
Maximum number of turbines	3500	Mathew (2006)
Intercept value of fish	0.5	
Slope value of fish	1/7000	
Intercept value of birds	1	
Slope value of birds	-1/3500	

[1]The parameter e_i is calculated on the basis of the Rough Capacity Factor (RCF) of a cell. The calculation of the RCF was taken from Mathew (2006). It is determined by wind speed and turbine performance characteristics. The RCF is multiplied with the generator power (2 MW) and number of hours in a year to get e_i.
[2]The investment costs of a turbine consist of turbine purchase (variable with depth and distance), and are multiplied by a percentage to account for infrastructure. The investment costs are multiplied by an annuity factor to calculate the costs on a yearly basis. This is parameter f_i. Yearly O&M costs (v_i) are calculated as a percentage of the purchase costs of the turbine.

Dutch EEZ. Biomass was converted to numbers assuming an average plaice (*Pleuronectes platessa*) weighs 1 kg.

For bird species we chose for illustrative purposes the combined summer dataset of the razorbill (*Alca torda*) and guillemot (*Uria aalge*). The dataset is combined as the two species cannot be separated when surveyed although most of the counts comprise guillemots (Noordzee atlas, 2008). For simplicity the flocks are referred to as razorbills. Both species have relatively high sensitivity to wind farms (Garthe and Hüppop, 2004) therefore the negative impacts on this species is among the highest in the ecosystem. The negative impacts on other bird species is probably lower or of a similar level. The slope used for the linear function that describes habitat suitability as a function of number of turbines (cf. Figure 2.2) equals a maximum of 3 birds killed per year per turbine, which is within the range found by Drewitt and Langston (2006). The number was chosen such that, when a cell is filled with the maximum number of turbines as defined by turbine characteristics, the habitat suitability for birds of that cell is worst case[2]. The fish species we chose is plaice (*Pleuronectes platessa*) of age class 4 years and older, as it is a widespread demersal fish and offshore wind farms have been shown to have a positive effect on demersal fish (Wilhelmsson et al., 2006). Furthermore juvenile plaice is not a target species of the fisheries and are concentrated in the Wadden Sea area. In our illustration we assume that filling a cell completely with wind turbines doubles the habitat suitability for fish in that

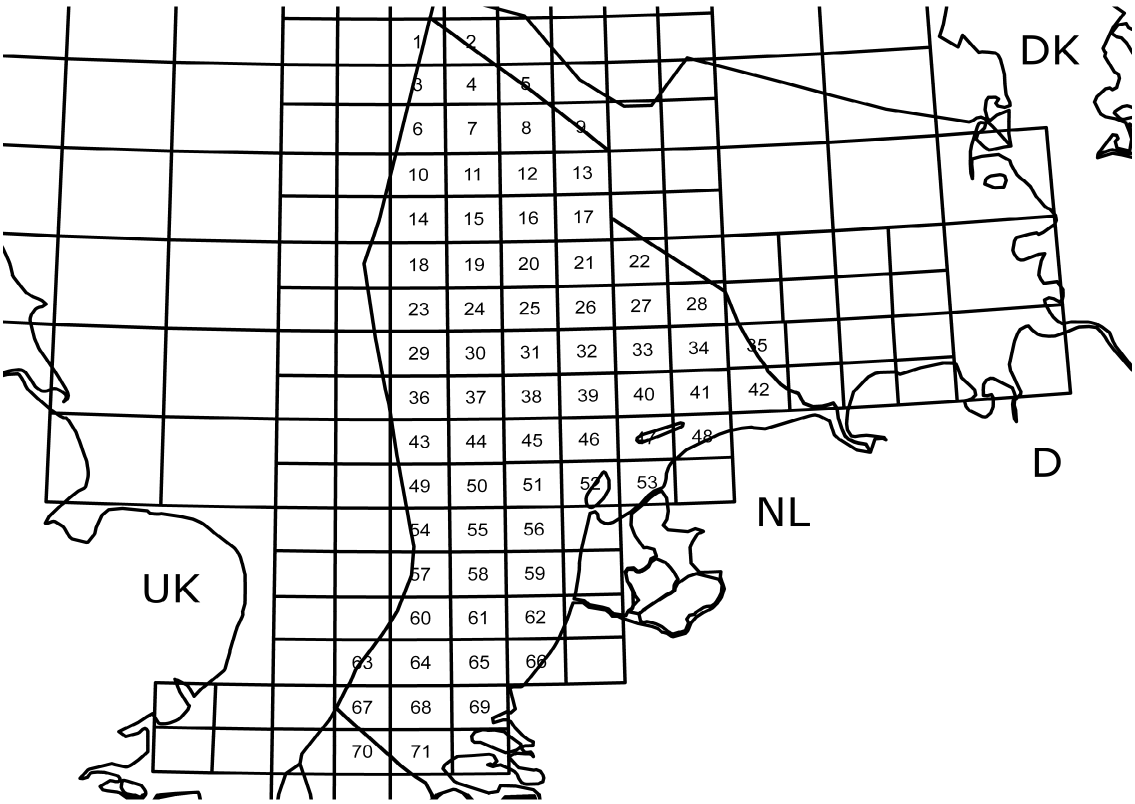


Figure 2.3: The Dutch EEZ and the numbered cells used in the model. From left to right UK: United Kingdom, NL: The Netherlands, D: Germany, DK: Denmark.

[2]The range found by Drewitt and Langston (2006) was between 0.01 and 23 birds per turbine per year. A number higher than three would give similar results, however, as it would still reduce the bird population to zero for the cells that are completely filled with turbines. A small difference may occur for cells that are not completely filled.

cell (i.e. the original habitat suitability of a cell before the turbine introduction is not ideal due to e.g. fishing pressure). Since the probability of species occurrence is mostly determined by local habitat suitability this causes the original fish population in that cell to double and reach its potential maximum. Although both assumptions of worst case habitat suitability for birds and best case suitability for fish when the maximum number of turbines is placed are both arbitrary, they serve to illustrate extreme cases. They can be relaxed in a sensitivity analysis and real data can be inserted when results of empirical ecological studies become available.

In addition to the usual restriction on the total numbers of birds and fish, one further restriction is applied in the model to make it more realistic. For each cell the maximum available area was calculated by excluding land, islands, the Wadden sea area, shipping lanes, anchorages and terrain reserved for military exercises, and the maximum number of turbines in each cell was modified accordingly.

2.4.2. The scenarios

One baseline simulation and four scenarios were run with the model. In the baseline simulation no turbines are built, and the model is calibrated such that the potential maximum number of species $n_{i,s}$, and the species probability $P_{i,s}$ replicate the original ecological data. The four scenarios have different restrictions imposed for ecological targets, and all scenarios are restricted to a total maximum installed capacity of 22,000 MW. This equals the current installed capacity of all power plants in the Netherlands (Eurostat, http://epp.eurostat.ec.europa.eu). This target can be thought of as a scenario in which the Netherlands would want to take most of its electricity from wind power to drastically reduce CO_2 emissions. Even an installed capacity of 22,000 MW, however, cannot completely replace current energy plants due to constraints in wind availability and demand peaks (Mathew, 2006). In the current model the 22,000 MW target can, in absence of further restrictions, in principle be met by four cells with turbines.

The first scenario is the pure economic scenario. The only restriction to location choice is the available marine area, i.e. areas not claimed by other uses. In the second scenario we account for razorbills in the area. Placement of turbines is still free but there is a restriction that 99% of the total razorbill population modeled under the baseline simulation should survive. This scenario implies a strong conservation effort on razorbills, which mimics a strong emphasis on species specific conservation. In the third scenario this restriction is changed to a minimum survival of 90% of the baseline simulation-modeled razorbill populations in cells with special ecological values. Cells were deemed to have special ecological value if they were either part of the Plaice Box, or designated as possible marine reserves under the integrated management plan for the Dutch North Sea (IDON, 2005). This scenario mimics a more spatially orientated conservation strategy that is more focused on habitat preservation. The fourth scenario has the restriction that 70% of the modeled baseline simulation razorbill population should survive and furthermore an index which is made up of fish and birds weighing 1 vs. 100, respectively, should be at least 105% of the value in the baseline simulation. It is possible to reach a higher value of the multi-objective index since there is a trade-off between birds and fish in this case. This scenario can be thought of as implementing multi-objective targets, where society weighs fish against birds, and then imposes a target. The scenarios and their restrictions are shown in Table 2.3.

2.4.3. Results

The main results of the different scenario runs are shown in Table 2.4. Figure 2.4 shows the distribution of razorbills and plaice as modeled by the baseline simulation and Figure 2.5 shows the location of the turbines in the four different scenarios as well as the corresponding changes in species populations, indexed on the baseline simulation.

To accentuate the trade-off we have also formulated gained and forgone existence values for fish and birds. This value was calculated as the difference between the baseline simulation multiplied with the price. Beattie et al. (2002) use market prices as indicators of existence value and we follow that approach. The price in (€/kg) was applied by assuming that the average plaice weighs 1 kg. For the razorbills we used the value of Brown (1992) which is a

Table 2.3: The baseline simulation, the modeled scenarios and their restrictions

Scenarios	Restrictions
Baseline simulation	No turbines
Scenario 1: Economic	Turbine numbers only restricted by cell space and maximum installed capacity.
Scenario 2: Bird protection	99% of the total modeled baseline simulation bird population should survive. Restrictions on cell space and maximum installed capacity.
Scenario 3: Habitat protection	90% of the modeled baseline simulation bird population in cells with special ecological value should survive. Restrictions on cell space and maximum installed capacity.
Scenario 4: Multi-objective target	The multi-objective index should be at 105% of its maximum. 70% of the total modeled baseline simulation population should survive. Restrictions on cell space and maximum installed capacity.

replacement cost. Both values were converted to Euro and adjusted to current price level.

From Table 2.4 and Figure 2.5 we can see that the different policies have quite different effects. In scenario 1 where no ecological restrictions are in place the most profitable locations are those with high wind speeds, which are the three cells in the top of the EEZ and one a bit more to the right. Since these cells do not contain any shipping lanes they can be filled with the absolute maximum number of turbines. Under the current parameter values wind speed is clearly more important than the costs of the location (related to depth and distance) as the cells with the highest wind speed are also the most expensive ones, and they are still chosen. The birds in these locations of course suffer maximum damage, i.e. they cannot live in these areas anymore, whereas the fish population enjoys the full benefits of protection and doubles. Total bird stocks are not too strongly affected due to the medium low original number of birds in this area and the fact that so few cells are needed.

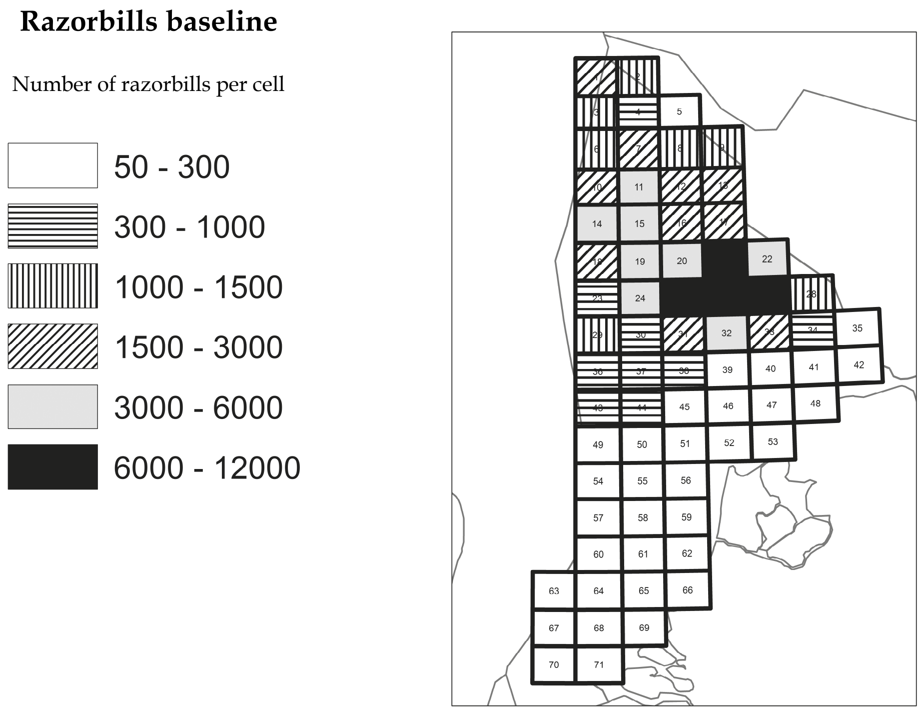
The restriction that 99% of the razorbill population should survive in scenario 2 changes the allocation of turbines as well as the main results. The cells selected now are less profitable but a greater bird population survives, as the original bird populations in these cells is low. The revenues of wind energy generation fall by 7%, but the razorbill population increases by 4% and the plaice stock increases by 12%, compared to the economic scenario. Whether this justifies the fall in revenues is a matter of preferences.

In scenario 3 some of the cells used originally in the economic scenario are part of the areas with special ecological value. The restriction of protecting 90% of the modeled baseline simulation razorbill population in those cells is

Table 2.4: Modeled revenues, numbers of razorbills and plaice and the existence value associated with the gains and losses in razorbill and plaice populations.

	Baseline simulation	S 1: Economic	S 2: Razorbill protection	S 3: Habitat protection	S 4: Multi-Objective
Revenues (million 2007 €)	-	4330	4028	4180	4104
Total number of razorbills (thousands)	119.13	113.09	117.94	113.21	114.59
Total number of plaice (millions)	21.54	19.84	22.31	22.78	23.66
Forgone existence value razorbills (million 2007 €)[1]	-	3.24	0.64	3.17	2.43
Gained existence value plaice (million 2007 €)[1]	-	0.12	0.30	0.49	0.84

[1]For an explanation of monetary values used and calculations see section 2.4.3.



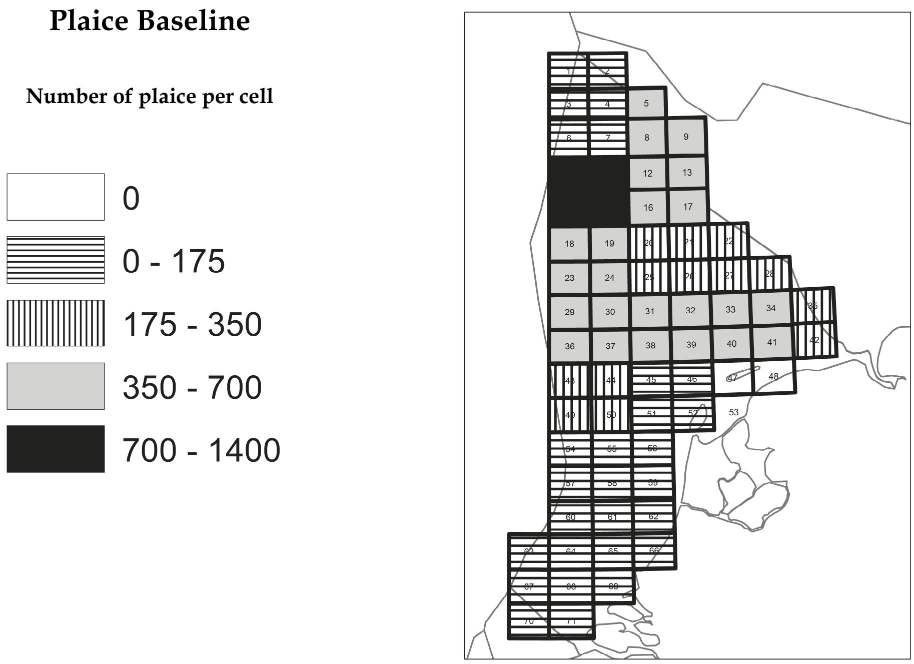


Figure 2.4: Modeled baseline simulation distribution of razorbills and plaice in the Dutch EEZ.

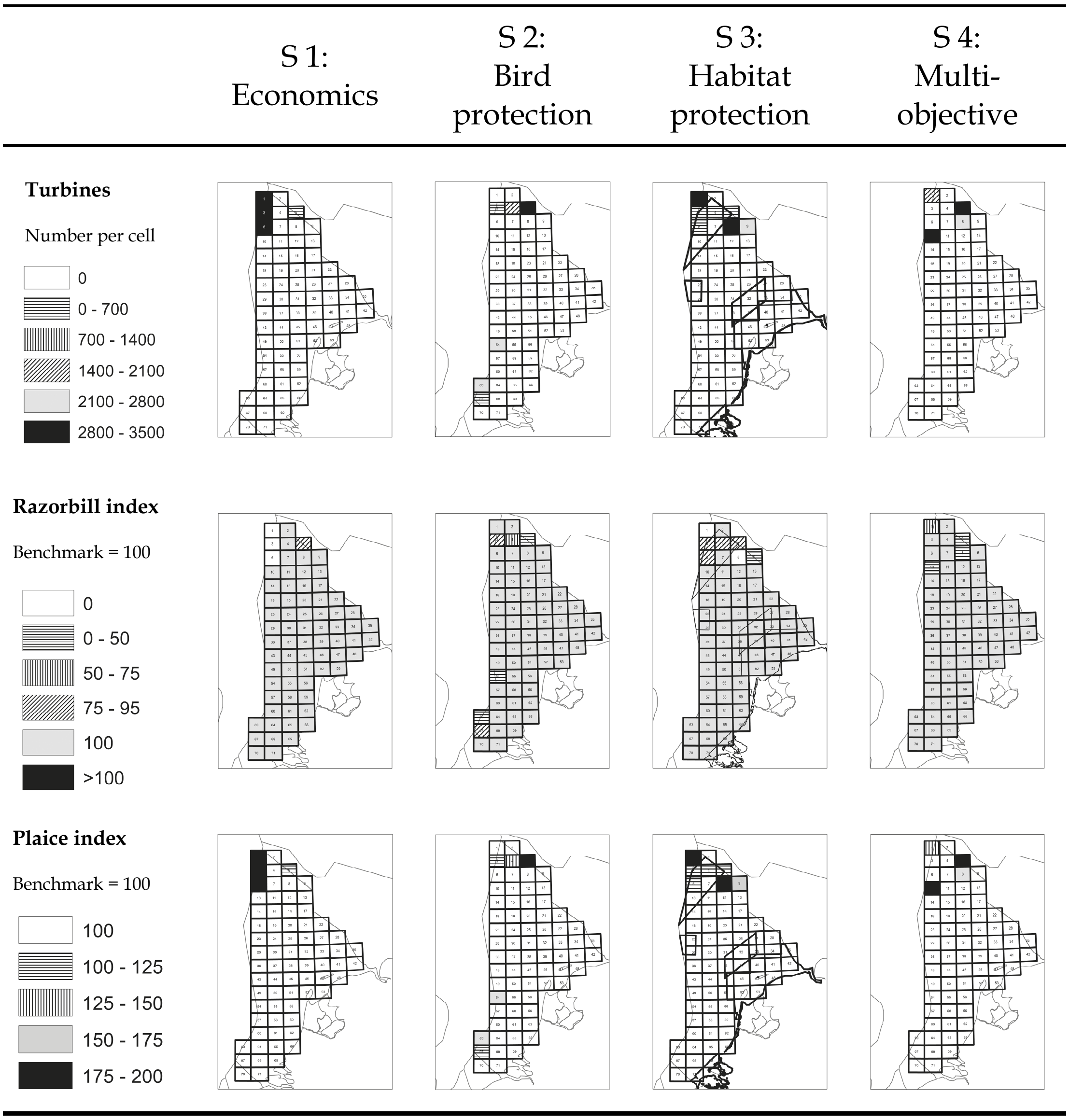


Figure 2.5: Number of turbines, and indices of razorbill and plaice in different scenarios with a total installed capacity of 22000 MW in the Dutch EEZ

now the main restriction that drives the model results. This restriction does not allow building more than a small number of turbines in the cells that are part of an area with special ecological value. Hence the turbines can still be built in the most profitable area but have to be more spread out. This also affects revenues, razorbill and plaice numbers, but the changes are relatively small: revenues fall by 3% whereas razorbills and plaice populations increase by 0.1% and 4% respectively, compared to the economic scenario. If we look at the change in existence value lost and gained we see the same pattern. At current prices, the change in existence value would hardly affect the total social revenues as the revenues of wind energy are much larger than gained and forgone existence values.

Under the multi-objective scenario the turbines are again placed in the Northern area of the Dutch EEZ. The reduction in razorbill numbers under the multi-objective scenario is so small that it hardly affects the index. Moreover the increase in number of plaice makes up for the decrease in bird numbers. The spatial configuration as well as the revenues, razorbill and plaice numbers are similar to those under the economic scenario. The multi-objective is reflected in the configuration: turbines are now placed in cells with a reasonable profitability, a higher general plaice population and a medium low bird population. The revenues are in between the habitat protection scenario and bird protection scenario, for the razorbill population it is better than the habitat protection scenario, but worse than the bird protection scenario, and it is better for the plaice population in both cases.

2.4.4. Sensitivity analysis

We applied a sensitivity analysis on several parameters to check the robustness of the model results, mainly focusing on the economic scenario. Connectivity, however, was checked against the minimum birds scenario as this spatial component is most likely to affect the bird populations in the neighborhood, possibly making it more difficult to sustain the required number of birds.

We examined the effect of turbine capacity, investment costs, marginal distance costs and connectivity. Turbine capacity determines the rough capacity factor in each cell (together with wind speed), the maximum number of turbines in a cell, and the slope of the bird and fish habitat parameters. Alternative specifications of turbine characteristics (for 1.5, 3.5 and 5 MW turbines) influenced total revenues, the maximum number of turbines installed in a cell. The results for the different turbines are shown in Table 2.5.

Smaller turbines give smaller total revenues because maximum power output is lower, hence they produce less power. Larger turbines are better as they are higher and intercept stronger winds, so power output and revenues

Table 2.5: Main model results for revenues razorbill and plaice numbers in scenario 1, when using different turbine specifications.

Turbine type	Total revenues (billion 2007 Euro)	Total number of razorbills (thousands)	Total number of plaice (million)
1.5 MW	4.29	112.93	22.21
2 MW[1]	4.33	113.09	21.87
3.5 MW	5.66	113.70	21.76
5 MW	6.37	113.50	21.76

[1]Turbines used in the original model

are better. From the environmental protection perspective larger turbines may be a more promising strategy, as less turbines are needed so there are fewer razorbills that are disturbed (see also Figure 2.2). Due to the inherent structure of the model, which causes the habitat suitability for birds to be at its minimum when the maximum number of turbines is placed in a cell, more birds are killed per turbine. This may cause differences if cells are not completely filled with turbines. This explains the small difference in bird numbers between the 5 MW and 3.5 MW turbines.

Another important point is the investment costs. In the current model setting the turbine price increases with depth and distance from the shore, and infrastructure costs are calculated as an additional 30% of the turbine price. We analyzed the changes under a range of multipliers from 1.1 to 2.0 in steps of 0.1. The allocation of turbines is relatively robust against these changes. The allocation is the same for the range 1.1-1.3, one cell is changed for the range 1.4-1.7. For the range 1.8-2.0 the allocation is gradually shifted to the south (Figure 2.6c). A high RCF cannot compensate any more for high costs at this point. These cells have relatively low costs whereas their RCF is higher than in other cells with similar costs. In the current model settings increasing the discount rate or decreasing the project life time have similar effects.

In a similar analysis we increased the marginal distance costs over a range of 10-90%. The results were a bit stronger than in the previous analysis, i.e. the turbine allocation switched to cells close to the shore (See Figure 2.6d).

We checked the connectivity effects with a range of r_{ij} over [0.1-0.8] in steps of 0.1. We approached each value of r_{ij} with a d and ran the model. For the smallest r_{ij} = 0.1 the model becomes unfeasible, because the ecological model is unable to replicate the baseline for such low values of connectivity. Clearly connectivity should be assumed to be higher. For r_{ij} = [0.4-0.8] the model

a) b) c) d)

Figure 2.6: Turbine distributions in the Dutch EEZ under: a) the original economic scenario, b) the economic scenario with turbines of 3.5 MW, c) the economic scenario with an 80% increase in investment costs, d) the economic scenario with an 80% increase in marginal distance costs

results do not change. There is a small change in the allocation of turbines and total number of birds for r_{ij}=0.2. Cell 5 is cleared and instead cells 2 and 11 are used, this is done to save nearby local populations such as the one in cell 7. The total number of birds compared to higher values of connectivity is reduced by 1.7%. For a connectivity of 0.3 this reduction is 0.09%, and allocation of turbines does not change.

The population equations used in the model are highly stylized. The spatial component mainly consists of the state of adjacent cells. Therefore the effect of the turbines is mainly local, especially for higher degrees of connectivity, which assures that turbine-free neighboring cells supply the necessary immigration. This high degree of connectivity is justified as marine ecosystems are often regarded as being more open than terrestrial systems (Carr et al., 2003).

2.5. Discussion & conclusions

In this chapter we presented a modeling framework to spatially allocate offshore wind farms in order to maximize revenues while accounting for other economic activities, and protecting birds and fish populations (i.e. razorbill and plaice in our example). Four different scenarios were analyzed for the Dutch EEZ, and we conducted a sensitivity analysis.

The analyses show that the allocation of turbines in the Dutch EEZ is quite robust against changes in parameters. Basically two regions are preferred from an economic perspective. The first region comprises the top cells of the EEZ, the second region lies just south of the middle of the EEZ. The two regions reflect a difference in emphasis on the RCF and costs. The cells of the second region overlap with selected locations in a similar economic study to build turbines in the Dutch North Sea (Ministerie van Economische Zaken, 2004). Some of the cells that are attractive from a cost perspective are quite full already due to shipping lanes and other uses. Although the model adjusts the number of turbines for those uses, the actual number of turbines that can be realized in those cells may be lower.

Neither region is very important to bird conservation, as both have low original bird populations. However, different bird species have different requirements and the underlying distribution of individuals influences the number of birds in a cell, thus the results do not necessarily hold for other bird species.

The model shows that careful spatial planning of turbines may prevent the turbines from causing major harm to bird populations, while increasing local fish stocks. This finding is similar to that of Petersen and Malm (2006), who argued after a review that with proper siting and careful construction reef effects would outweigh possible negative effects.

Whether these increases in fish stocks have a spill-over effect remains to be seen. The current model allows for negative spill-overs but not for positive ones. So far, however, the evidence regarding spill-over effects of MPAs differs between species (Murawski et al., 2005).

The scenario where MPAs are partially closed for wind farms clearly differs from the other three scenarios, because the conservation policy is more spatially orientated. One could argue that wind farms should not be placed in special ecological areas at all. However the small number of turbines in those cells could be interpreted as being placed on the edge of the reserve. They increase fish stocks by basically extending the reserves. Whether or not such arrangements would be beneficial for marine environmental protection would also depend on the other species present in the reserve such as birds, mammals and the local benthos. The bird species we chose to model does not generally have large subpopulations in the special areas and is therefore less influenced by the wind farms in the reserves.

This chapter has shown the necessity of considering both economic and ecological spatial effects of planning offshore wind farms. The choice of location includes trade-offs between revenues, costs and species that are only apparent with a spatial model. Even though the model is highly stylized, the current version provides an indication of how the spatial trade-offs interact, in addition to the absolute minimum effects that need to be considered when making location choices. These factors and trade-offs should now be further explored in order to reap the windfall effects from the spatial planning of offshore wind farms.

Chapter 3*: Planning Marine Protected Areas: a multiple use game

3.1. Introduction

The marine environment supplies several goods and ecosystem services to society. Yet it is also under increasing pressure from a variety of human activities, such as fisheries, oil & gas exploration and shipping. To extract goods and services sustainably and to protect vulnerable ecosystems we need to manage human activities in the marine domain.

The European Commission is at the forefront in safeguarding and exploiting its Exclusive Economic Zones (EEZs). This is reflected in its policies and new initiatives such as the Common Fisheries Policy (EU, 2009a), the Water Framework Directive (EU, 2009b), the Maritime Policy (European Commission, 2007) and the Marine Strategy Directive (MSD) (European Commission, 2008). All of these call for ecosystem management at a regional sea level, i.e. between countries sharing a common sea, such as the North Sea. The EU MSD explicitly calls for the formulation of integrated marine strategies by its member states, which should "apply an ecosystem-based approach to the management of human activities while enabling the sustainable use of marine goods and services" (European Commission, 2008 article 1.1 and 1.2). Consequently we need to consider regional seas as entities surpassing boundaries of individual countries.

The necessary integration of management plans both within countries as well as between countries, is far from being reached. The management plans of individual member states' Exclusive Economic Zones (EEZs) are not always in concordance with each other (Stel, 2003; Douvere et al., 2007) and planning and policy making at national level is often fragmented since responsibilities for different activities are divided between different organizations, institutions and ministries (Stel, 2002). The fragmentation favors an attitude in which effects of human activities are considered in isolation, whereas the effects are actually interdependent and cumulative (Elliott, 2002; ICES, 2003; de la Mare, 2005).

To manage our seas sustainably we need to develop tools and policy instruments that are capable of achieving goals that have been set at EU level, and mitigate fragmentation at lower levels. Marine Protected Areas (MPAs) may be such a tool. They have been proposed for fisheries management for a long time (Guénette et al., 1998) and more recently also as a tool to tackle

*This chapter is based on the paper: Maarten J. Punt, Hans-Peter Weikard, Rolf A. Groeneveld, Ekko C. van Ierland and Jan H. Stel, 2010. Planning Marine Protected Areas: a multiple use game. *Natural Resource Modeling* **23**(4): 610-646.

biodiversity conservation, ecosystem restoration, regulation of tourist activities and as an example of integrated coastal management if all of these are included (Jones, 2002).

Over the last two decades several studies have advocated MPAs as a fisheries management tool, claiming that MPAs increase yields through spillover, especially in heavily fished areas (e.g. Polacheck, 1990, Bohnsack, 1993 and Holland and Brazee, 1996). The initial optimism of these studies was later dimmed by several modeling studies that showed that earlier papers largely ignored the effects on effort displacement (Hannesson, 1998; Anderson, 2002), spatial heterogeneity in fishermen behavior (Smith, 2004; Smith and Wilen, 2004) and (spatial) heterogeneity in fish behavior (Holland, 2000). All effects were shown to be able to reduce or negate the positive spillover effects of MPAs. The main reasons for still advocating MPAs for fisheries management that remained were uncertainties and shocks (Lauck et al., 1998; Charles, 2001, Sumaila, 2002; Sumaila et al., 2007; Kvamsdal and Sandal, 2008).

However this spatial heterogeneity was also shown to be able to make MPAs worthwhile in certain cases especially if MPAs increase growth rates (e.g. Sanchirico and Wilen, 2001 and Sanchirico, 2004). Fishing may have a destructive effect on habitats (Jennings and Kaiser, 1998; Jennings et al., 2001; Armstrong and Falk-Petersen, 2008) and hence areas that are protected and where no fishing occurs may have a positive effect on the growth rate of the fish stock outside the MPA through habitat enhancement and the preservation of the nursery function of the MPA (Rodwell et al., 2002; Schnier, 2005a; Armstrong, 2007; Armstrong and Falk-Petersen, 2008). Moreover studying MPAs only as a fisheries management tool ignores other important aspects of MPAs such as species conservation and other uses (but see: Brown et al., 2000, Boncoeur et al., 2002, Villa et al., 2002, Dalton, 2004 and Ngoc, 2010).

The multiple uses of MPAs and their impacts on the marine ecosystem require full cooperation among all countries that share a regional sea such as the North Sea. On the one hand, strategic interaction and sub-optimal policy outcomes may occur, because in general no central authority exists that can enforce cooperation on these issues. On the other hand, if various functions of MPAs (e.g. in terms of fisheries and nature conservation) are linked, the advantages of cooperation may increase in such a way that self-enforcing agreements can be formed.

In this chapter we will analyze the problem of multiple use MPAs by multiple countries using game theory. This provides insights into the functioning of MPAs as a policy instrument for the MSD and the Maritime Policy Directive. More specifically, we will examine the size of MPAs that countries adopt, when they account for effects of MPAs on fisheries and species conservation separately or jointly and investigate the effect of playing cooperatively versus playing non-cooperatively on a single issue when multiple issues are at stake.

In game theory strategic interaction between countries has been investigated in a general fisheries context (Munro, 1979; Levhari and Mirman, 1980; Hämäläinen et al., 1985; Vislie, 1987; Hannesson, 1997; Arnason et al., 2000; Bjørndal and Lindroos, 2004; Kronbak and Lindroos, 2007). The related problems of public good provision, international environmental agreements and enforcement have also been considered both in terms of transboundary pollution abatement and coalitions (Mäler, 1989; Mäler and de Zeeuw, 1998; Finus, 2003; Finus et al., 2006; Weikard et al., 2006a; Nagashima et al., 2009) and in terms of possible coalitions for fisheries management (Pintassilgo, 2003; Pintassilgo and Lindroos, 2008; Pintassilgo et al., 2010). Game theoretic treatments of (marine) protected areas have received less attention so far. Sanchirico and Wilen (2001) and Beattie et al. (2002) study strategic interaction, both among fishermen and between fishermen and policymakers. Sumaila (2002) devises a computational model of assigning an MPA as a differential game between two agents. Ruijs and Janmaat (2007) have studied strategic positioning of MPAs. They consider two countries, the location of MPAs and the effect of different migratory regimes in a differential game. Busch (2008) derives some general conditions from game theory for terrestrial transboundary reserves to be superior over isolated reserves.

Our chapter contributes to the literature as it considers the combination of multiple uses of MPAs with multiple agents: we examine cooperation and defector incentives when multiple agents operate in a multiple use setting, i.e. by considering impacts on fisheries and nature conservation. Furthermore the fisheries MPA model is improved to accommodate the habitat enhancement effects of MPAs, and a conservation game is introduced, that uses standard ecological functions to model species richness and consequent conservation benefits.

In section 3.2. we present a game theoretic model that investigates the issue of multiple use MPAs in a multiple country setting by developing two separate models of MPAs in a multiple country setting: one model for the fisheries case and one model for the species conservation case. Next we combine the models and investigate the impacts of combining fisheries and nature conservation goals on MPA size. In section 3.3. we provide a numerical example to illustrate our results and in section 3.4. we discuss the limitations of the model, draw conclusions and discuss their implications.

3.2. Model description

3.2.1. Model background

In our model we consider a regional sea, such as the North Sea, that is completely claimed by a number of countries. These countries have divided the sea into Exclusive Economic Zones (EEZs) of equal size. The sea contains only one fish species that is of commercial interest for fisheries, and this fish species consists of a single stock. The other fish species, as well as mammals and benthos are assumed to have no commercial value, only existence value. We ignore effects of time and space, to focus exclusively on basic mechanisms. Furthermore we confine our interest to the steady state and neglect transition paths.

Our fisheries model describes optimal harvesting of fish by a number of countries in a common sea with a single stock. We analyze the impact of establishing an MPA in a context where each country has a fixed share in the fishing area. In the fisheries model the cost of protection is a reduction in catchability proportional to MPA size[1]. We assume that from a fisheries' perspective, assigning an MPA does not cause any costs other than opportunity costs of forgone harvest and additional effort costs. For simplicity monitoring and enforcement costs are neglected. The gains of a country consist of an increase in growth rate of the shared stock owing to an increase in habitat quality in the protected area. Such an increase cannot be reached by conventional harvest restrictions because it would only reduce overall fishing pressure but not release one area completely from fishing pressure, to recover habitat. This habitat effect of MPAs is a public good since a single country bears the cost while all countries benefit.

Our nature conservation model describes conservation efforts by a number of countries in the same common sea. We analyze the impact of the total size of an MPA on the number of species protected and resulting costs and benefits. Each country's MPA is a contribution to the total protected area, but benefits derived from the extra species protected by this MPA accrue to all countries, making this another public goods issue.

3.2.2. Marine Protected Areas as fisheries management tool

Our fisheries model assumes that there is only one fish stock that is worth harvesting from a commercial point of view. We consider a set N of n symmetric countries that harvest from a single stock, ignoring other stocks and

[1]In the current model setting we assume a direct proportionality between reduction in harvest and MPA size. In further research other specifications such as concave or convex reductions can be explored. However, such specifications would not alter the basic line of reasoning.

species. If countries cooperate, they maximize the sum of their profits. If they defect, they optimize their own profits. Country i's profits Π_i depend on harvest and incurred costs. We use a supply side model that assumes that the full harvest can be sold at a fixed unit price p:

$$\Pi_i = pH_i - C_i \qquad \forall i \in N \tag{3.1}$$

where H_i is total harvest of country i and C_i are total costs of country i.

We assume that the stock is uniformly distributed over the fishing grounds and we model the growth of the stock with a modified Schaefer production function. We assume that the sea is completely claimed by EEZs and that all EEZs are of equal size. The total size of the sea is normalized to one, and consequently each country has an EEZ of size $\frac{1}{n}$. In absence of MPAs each country's fishing ground is its EEZ.

We assume that in equilibrium harvest equals the growth of the fish stock. The size of the MPA affects both the harvest and the growth rate of the fish stock but for simplicity we assume that location of the reserve does not matter.

The total growth of the fish stock is modeled with a modified logistic growth function scaled such that the carrying capacity, i.e. the maximum stock size, equals one. This growth function is:

$$G(X,M) = R(M)X(1-X) \tag{3.2}$$

with $R(M)$ the internal growth rate of the stock $0 \le X \le 1$, and $0 \le M \le 1$ the total MPA.

Setting aside a share of fishing ground as an MPA is assumed to have a positive effect on the growth rate R through enhancing the growth rate in the MPA. We model this enhancement as a coupled production function: the internal growth rate is r_b in the unprotected area and $(r_b + r_M)$ in the protected area. This assumption is based on increased recruitment that is achieved by increasing spawning biomass through the absence of fishing in sensitive areas. Similar functional forms for Marine Protected Area modeling have been used by e.g. Schnier (2005a,b), who also modifies the growth rate and Armstrong (2007) who modifies the carrying capacity as an effect of MPAs. If we further assume that both stock and carrying capacity are proportional to protected area we get:

$$G(X,M)=r_b(1-M)X\left(1-\frac{(1-M)X}{(1-M)}\right)+(r_b+r_M)MX\left(1-\frac{MX}{M}\right)$$
$$=(r_b+r_M M)X(1-X) \tag{3.3}$$
$$=\left(r_b+r_M\sum_i M_i\right)X(1-X)$$

with $0 \leq M_i \leq \frac{1}{n}$ is the size of the MPA of an individual country i. The sum of individual MPAs equals total MPA.

To model harvest we use a modified Schaefer harvest function:

$$H=Q(M)EX=\sum_{i=1}^{N}\left(Q_i(M_i)E_i\right)X \tag{3.4}$$

with Q total catchability, Q_i catchability in the EEZ of country i, E total effort level and E_i effort level of country i. In a standard Schaefer function catchability is a parameter. In our model we go beyond this, and assume catchability is a combination of a technical coefficient and the fishable area. Therefore catchability Q is a decreasing function of M. We use the following functions for $Q(M)$ and $Q_i(M_i)$:

$$Q(M)=q_o-q_M M$$
$$Q_i(M_i)=q_o-q_M\left(\tfrac{n-1}{n}+M_i\right) \tag{3.5}$$

where q_o is original catchability and q_M is marginal catchability reduction due to area that cannot be fished because it is owned by other players ($\frac{n-1}{n}$), or protected (M_i).

Costs are assumed to be constant per unit of effort:

$$C_i=c_E E_i. \tag{3.6}$$

Full cooperation on MPAs for fisheries

When players fully cooperate they maximize the sum of total profits:

$$\max \Pi = \max \sum_{i\in N} \Pi_i \tag{3.7}$$

with Π denoting total profits.

To obtain the steady state of the model we set total growth equal to harvest and solve for effort level[2]:

[2]The model can also be solved for stock. We chose to use effort here because the solutions are more straightforward.

$$H = G(M,X) \Rightarrow R(M)X(1-X) = Q(M)EX \Leftrightarrow X = 1 - \frac{Q(M)E}{R(M)}. \tag{3.8}$$

Using harvest function (3.4) and substituting the equilibrium stock given in (3.8) we get the objective function:

$$\Pi(E,M) = pH - c_E E = pQ(M)E\left(1 - \frac{Q(M)E}{R(M)}\right) - c_E E. \tag{3.9}$$

Taking the First Order Condition with respect to effort we obtain[3]:

$$\frac{\partial \Pi}{\partial E} = pQ(M)\left(1 - 2\left(\frac{Q(M)E}{R(M)}\right)\right) - c_E = 0. \tag{3.10}$$

If we substitute the equilibrium stock given in (3.8) into equation (3.10), we get:

$$pQ(M)(2X-1) = c_E. \tag{3.11}$$

Equation (3.11) displays standard components of a FOC in a static fisheries model. The left hand side is the marginal benefit: an increase in effort increases catch if $X > \frac{1}{2}$, and this additional catch is valued at *p*. The right hand side is the marginal cost of effort.

Similarly, the first order condition with respect to *M* is given by:

$$\frac{\partial \Pi}{\partial M} = \left(\frac{(q_o - q_M M)E}{(r_b + r_M M)}\right)\left(\left(\frac{pr_M(q_o - q_M M)E}{(r_b + r_M M)}\right) + 2pq_M E\right) - pEq_M = 0. \tag{3.12}$$

If we substitute the equilibrium stock given in (3.8) into equation (3.12) we get:

$$pr_M(1-X)^2 = pq_M E(2X-1). \tag{3.13}$$

This clearly illustrates the effect of MPAs. The left hand side of (3.13) is the marginal benefit of an MPA: an extra unit of protected area increases growth and consequently harvest by $r_M(1-X)^2$ which is valued at *p*. The right hand side shows the marginal cost of such an area as forgone harvest in an extra unit of protected area, also valued at *p*.

Solving equation (3.10) and (3.12) simultaneously gives a cubic equation which can be factorized into quadratic and linear parts and solved. Given the

[3]We present the interior solution of the model, although these solutions, too, may be corner solutions. Corner solutions arise if the bounded variables such as stock, effort and Marine Protected Area size exceed their bounds under the solution. Solutions that are corner solutions under our parameter assumptions are not analyzed.

Table 3.1: Signs of derivatives of equilibrium MPA under full cooperation and Nash equilibrium with respect to parameters in the fisheries game

Derivative	$\partial M/\partial p$	$\partial M/\partial c_E$	$\partial M/\partial r_M$	$\partial M/\partial r_b$	$\partial M/\partial q_M$	$\partial M/\partial q_o$
Full cooperation	+	-	Und.	-	Und.	Und.
Nash equilibirum	+	-	Und.	-	-	Und.

Und. = Undetermined. All relations are derived from the implicit function theorem. Assumptions used for signs above: All parameters >0, 0≤*M*≤1, 0 ≤ M_i ≤1/*n*, 0≤*Q*(*M*)≤1, $q_o \geq q_M$.

assumption that all parameters are strictly positive, we find two corner solutions where effort is zero and one interior solution:

$$M^* = \frac{(2pq_o + c_E)r_M - \sqrt{c_E r_M \left(r_M(8pq_o + c_E) + 8pq_M r_b\right)}}{2pq_M r_M},$$

$$E^* = \frac{\left[\begin{array}{l}\left(\left(2p^2q_o^2 + 6pc_Eq_o + c_E^2\right)r_M^2 + \left(4p^2q_Mq_o + 6pc_Eq_M\right)r_br_M + 2p^2q_M^2r_b^2\right)\sqrt{r_M} + \\ \left((2pq_o + c_E)r_M^2 + 2pq_Mr_br_M\right)\sqrt{c_E\left((8pq_o + c_E)r_M + 8pq_Mr_b\right)}\end{array}\right]}{\left[\begin{array}{l}\left(\left(-6pc_Eq_Mq_o - c_E^2q_M\right)r_M - 6pc_Eq_M^2r_b\right)\sqrt{r_M} + \\ \left((2pq_Mq_o + c_Eq_M)r_M + 2pq_M^2r_b\right)\sqrt{c_E\left((8pq_o + c_E)r_M + 8pq_Mr_b\right)}\end{array}\right]}. \qquad (3.14)$$

To evaluate the influence of individual parameters on equilibrium MPA we use the implicit function theorem and first order conditions (3.10) and (3.12) to determine the signs of their derivatives. The derivatives are shown in Appendix 3.A., the signs in Table 3.1.

Most signs of derivatives can be determined with exception of the derivatives with respect to r_M and q_o. The signs that can be determined have the expected sign. An increase in price of fish (*p*) would make harvesting more worthwhile, and thus the protected area is increased to increase harvest. An increase in cost of effort (c_E) makes a protected area more expensive because the protected area decreases the effectiveness of effort through the catchability.

Consequently with a smaller MPA, less effort is required, thus as effort becomes more expensive MPA size is reduced.

Similarly an increase in r_b or q_M decreases protected area as they make the MPA more costly by increasing forgone harvest necessary to protect an area to obtain the same growth bonus. The signs of $\partial M/\partial r_M$ and $\partial M/\partial q_o$ are mainly determined by the difference between price of fish and cost of effort and the parameters q_M and q_o. If $p >> c_E$ and $Q(M)$ is not too small, both signs of the derivatives are positive. This is in line with expectations: if the price of fish is large relative to effort costs, an MPA pays off: the added growth bonus and thus extra fish to catch is more valuable than the relatively minor costs of extra effort needed to catch that fish. A larger growth bonus (r_M) makes the MPA more valuable, because it increases extra available catch.

Similarly a larger original catchability (q_o) makes the MPA even cheaper because the reducing effect on harvest of the MPA is smaller.

The Nash equilibrium for MPAs for fisheries

In Nash equilibrium each individual player wishes to maximize his own fisheries profits, given that other players maximize their profits. We assume that the sum of harvest of all players is equal to the growth of the stock. The harvest depends on choices of effort and MPA size:

$$\sum_i H_i(M_i, E_i) = G(M, X) \Rightarrow \sum_{i=1}^{N} \left(Q_i(M_i)E_i\right)X = R(M_i, M_{-i})X(1-X) \Leftrightarrow$$

$$X = 1 - \frac{Q_i(M_i)E_i + (n-1)Q_j\left(M_j\right)E_j}{R\left(M_i, M_{-i}\right)} \quad \forall i \in N, i \neq j \tag{3.15}$$

where $M_{-i} = \sum_{j \in N \setminus \{i\}} M_j$.

An individual player's optimization problem is:

$$\max \Pi_i = \max\left(pE_i\left(1 - \frac{Q_i(M_i)E_i + (n-1)Q_j\left(M_j\right)E_j}{R\left(M_i, M_{-i}\right)}\right) - c_E E_i\right) \quad \forall i \in N. \tag{3.16}$$

Maximizing individual profit functions in (3.16) is not the same as in an open access regime. Even though countries optimize their own effort, no new entrants are allowed, thus rent is not driven to zero. The first order condition with respect to E_i is:

$$\frac{\partial \Pi}{\partial E_i} = pQ_i(M_i)\left(1 - \frac{Q_i(M_i)E_i + (n-1)Q_j(M_j)E_j}{R(M_i, M_{-i})}\right) - \frac{pE_i(Q_i(M_i))^2}{R(M_i, M_{-i})} - c_E = 0. \tag{3.17}$$

If we substitute the equilibrium stock given in (3.15) into equation (3.17), we get:

$$pQ_i(M_i)\left(2X - 1 + \left(\frac{(n-1)Q_j(M_j)E_j}{R(M_i, M_{-i})}\right)\right) = c_E \tag{3.18}$$

which displays standard components of a FOC in a static fisheries model. The left hand side is the marginal benefit: an increase in effort increases catch if X is larger than ½ minus the catch of other players, and this additional catch is valued at p. The right hand side is marginal effort cost. From a comparison between (3.11) and (3.18) it is clear that in Nash equilibrium the stock is smaller, through the catch by other players.

The FOC with respect to MPA is:

$$\begin{aligned}\frac{\partial \Pi_i}{\partial M_i} = {} & -pq_M E_i\left(1 - \frac{\left[\begin{array}{l}\left(q_o - q_M\left(\frac{n-1}{n} + M_i\right)\right)E_i \\ +(n-1)\left(q_o - q_M\left(\frac{n-1}{n} + M_j\right)\right)E_j\end{array}\right]}{r_b + r_M\left(M_i + (n-1)M_j\right)}\right) \\ & + pE_i\left(q_o - q_M\left(\tfrac{n-1}{n} + M_i\right)\right) \times \\ & \left(\frac{q_M E_i}{\left(r_b + r_M\left(M_i + (n-1)M_j\right)\right)} + \frac{\left[\begin{array}{l}r_M\left(q_o - q_M\left(\frac{n-1}{n} + M_i\right)\right)E_i \\ +(n-1)\left(q_o - q_M\left(\frac{n-1}{n} + M_j\right)\right)E_j\end{array}\right]}{\left(r_b + r_M\left(M_i + (n-1)M_j\right)\right)^2}\right).\end{aligned} \tag{3.19}$$

If we substitute the equilibrium stock given in (3.15) into equation (3.19) we get:

$$pr_M(1-X)\left(1 - X - \frac{(n-1)Q_j(M_j)E_j}{R(M_i, M_{-i})}\right) = pq_M E_i\left(2X - 1 + \frac{(n-1)Q_j(M_j)E_j}{R(M_i, M_{-i})}\right). \tag{3.20}$$

This is similar to the full cooperation case but it now includes terms for other players. The left hand side is the marginal benefit of an MPA: an extra unit of protected area increases growth and consequently harvest by:

$$r_M(1-X)\left(1-X-\frac{(n-1)Q_j\left(M_j\right)E_j}{R\left(M_i,M_{-i}\right)}\right)$$

where the last term accounts for catch of other players. The extra harvest is valued at p. The right hand side shows the marginal cost of such an area in the form of forgone harvest (again with a modifier for other players) in that extra unit of protected area, also valued at p. A player cannot reap all extra harvest from an MPA, because some of it is harvested by others, and therefore the incentive to assign an MPA is lower. This can also be seen from the comparison between (3.13) and (3.20).

Solving equations (3.17) and (3.19) simultaneously for n symmetric players we get another cubic equation which can be factorized into quadratic and linear parts and solved[4]. Similar to the case of full cooperation we find three solution pairs, with two corner solutions, given that all parameters are strictly positive and $q_o \geq q_M$. The interior solution for each individual MPA is:

$$M_i^* = \frac{\left(2p\left(nq_o-(n-1)q_M\right)+c_E n^2\right)}{2npq_M} - \frac{\sqrt{nc_E\left(\left(4p\left(n(n+1)q_o-\left(n^2-1\right)q_M\right)+c_E n^3\right)+4p(n+1)q_M\frac{r_b}{r_M}\right)}}{2npq_M}. \tag{3.21}$$

We apply the same procedure as under full cooperation, using the implicit function theorem and equations (3.17) and (3.19) to establish the effect of individual parameters on equilibrium MPA. The results are the same as for full cooperation (Table 3.1, and Appendix 3.A.).

If we take the solutions from full cooperation and the Nash equilibrium we can obtain the difference between the respective protected areas and evaluate the effect of parameters on this difference.

$$M^{FC}-M^{N} = \frac{\sqrt{c_E n r_M\left(\left(4p(n+1)\left(n\left(q_o-q_M\right)+q_M\right)+c_E n^3\right)r_M+4(n+1)pq_M r_b\right)}}{2pq_M r_M} - \frac{\sqrt{r_M c_E\left(8p\left(q_o r_M+q_M r_b\right)+c_E r_M\right)}-(n-1)\left(2p\left(q_o-q_M\right)+c_E n+c_E\right)r_M}{2pq_M r_M} \tag{3.22}$$

where M^{FC} is total area set aside under full cooperation and M^N is total area set aside in Nash equilibrium. Given our previous assumptions (all parameters

[4] The equilibrium effort is a complex and long expression and adds little. It is available from the author upon request.

strictly positive, and $n \geq 2$), it can be shown that parameter r_b increases the difference in MPA between full cooperation and Nash equilibrium. Unfortunately, the effects of other parameters are ambiguous, and hence we have to resort to simulations. The same holds for the difference in payoffs between full cooperation and Nash equilibrium[5].

3.2.3. Marine Protected Areas for conservation

Conservation is a main goal of Marine Protected Areas. In this game we measure conservation success as species richness attained in a Marine Protected Area. We do not model populations of all species independently but instead use the species-area relationship to determine the number of species that a reserve contains. The species-area relationship (SPAR) has first been put forward by Arrhenius (1921) and is a curve of the general form $S = kA^z$ with S number of species, A area and k and z two positive parameters. It is explained by either the passive-sampling effect (MacArthur and Wilson, 1967) or the habitat diversity hypothesis (Williams, 1943).

The use of this relationship has been criticized for its scale-independent application and extrapolation (Leitner and Rosenzweig, 1997; Rosenzweig, 2005), and its ambiguous role in conservation decisions in the Single Large or Several Small (SLOSS) debate (Simberloff and Abele, 1976). Despite these criticisms SPARs can still be used as a predictor of local species richness, if one accounts for scale (Neigel, 2003; Rosenzweig, 2005). Furthermore even though SPARs may support either several small reserves or a single large reserve, depending on parameters, in our case we assume a uniform sea, implying that the same species would be protected in several small reserves, and therefore we apply a SPAR on a single large reserve[6].

For mathematical convenience we transform the species-area relationship into a log-log relationship, i.e. $\ln S = \ln k + z \ln A$. We further assume that costs rise linear with area set aside and that countries benefit from the (log) number of species in the total area set aside. Each country has an incentive to set some area aside, but given that others will also set aside some area, each will set aside less in Nash equilibrium. Assuming that all areas are interlinked and hence form one protected area, total benefits of species conservation are:

$$D(M) = b_P\left(\ln(S)\right) - c_P M = b_P\left(\ln(k) + z\ln(M)\right) - c_P M. \tag{3.23}$$

[5] The difference in payoff is a complex and long expression, shedding no light on the effect of parameters, therefore we do not show it here. It is available from the author upon request.
[6] In a more elaborate analysis a more complex specification on the relation between area, species richness and the benefits of nature conservation can be used. However, this would not change the fundamentals of the current analysis.

Here b_P represents marginal global benefits of protection of the log number of species, ln S, and k and z are positive parameters of the SPAR and c_P is the cost of protecting an area, such as monitoring and enforcement costs or opportunity costs for other uses.

Full cooperation on MPAs for conservation

Under full cooperation global welfare is maximized, accounting for benefit generation in all countries. The countries maximize:

$$D(M) = b_P\left(\ln(k) + z\ln(M)\right) - c_P M \tag{3.24}$$

with M again total protected area. This results in the following first order condition:

$$\frac{\partial D}{\partial M} = \frac{b_P z}{M} - c_P = 0 \Leftrightarrow M^* = \frac{b_P z}{c_P}. \tag{3.25}$$

The full cooperation optimal MPA size is independent of the number of players, as they consider the protection of the full sea, and only one optimum exists, although the solution is silent on how this should be reached. Given symmetric costs and benefits a fair solution would be that each country's MPA is $\frac{1}{n}$ M^*.

The Nash equilibrium for MPAs for conservation

In Nash equilibrium, each country maximizes its individual welfare, assuming all other countries maximize theirs. We assume each country gets benefits proportional to their EEZ size, $1/n$. Hence global marginal benefits of protection accrue to players in equal shares. Each country hence maximizes:

$$D_i(M_i) = {}^1\!/_n\, b_P\left(\ln(k) + z\ln\left(M_i + (n-1)M_j\right)\right) - c_P M_i, \quad \forall i \in N, i \neq j. \tag{3.26}$$

For an interior solution, the first order condition becomes:

$$\frac{\partial D}{\partial M_i} = \frac{b_P z}{n\left(M_i + (n-1)M_j\right)} - c_P = 0 \Leftrightarrow M_i^* = \frac{b_P z}{n c_P} - (n-1)M_j, \qquad \forall i \in N. \tag{3.27}$$

Again we see the strategic setting of the game from the first order condition. The additional benefit of an extra unit of area is in the numerator, but it is scaled by total area already protected as shown in the denominator. The costs of protecting one unit extra are c_P.

We obtain for a symmetric solution:

$$M_i^* = \frac{b_P z}{n^2 c_P} \tag{3.28}$$

which approaches zero for $n \to \infty$ clearly illustrating the sub-optimality of this Nash equilibrium.

The difference in MPA size assigned between full cooperation and Nash equilibrium is:

$$M^{FC} - M^N = \frac{b_P}{c_P} z\left(1 - \frac{1}{n}\right). \tag{3.29}$$

The difference in MPA size between full cooperation case and Nash equilibrium is increasing in number of players, in relative gains (b_p/c_p) of the MPA and slope of the SPAR (z). The difference in payoff between full cooperation and Nash equilibrium is:

$$D^{FC} - D^N = b_p z\left(\ln(n) - 1 + \tfrac{1}{n}\right). \tag{3.30}$$

For gains of an MPA we see a similar pattern: the difference increases with gains of a protected area and with increasing numbers of players. Consequently, if we assume that cooperation is easier if differences in payoff are smaller, then in the nature conservation case chances of cooperation decrease with an increase in b_p, z or n.

3.2.4. Combining the games

Marine Protected Areas affect fisheries as well as nature conservation. In order to see how equilibria change if we take both into account, we now combine the games. Payoffs now reflect both gains from fisheries and conservation. The optimal MPA size, for full cooperation and Nash equilibrium may change as well. Formally, this is different from issue linkage as in e.g. Folmer et al. (1993), Barrett (1997) or Buchner et al. (2005), because we are not dealing with two separate problems that are addressed with two different strategic instruments. In our game two separate problems are addressed with one instrument: the size of the MPA.

When countries combine both problems, the objective function under full cooperation becomes:

$$W(E,M) = pQ(M)E\left(1 - \frac{Q(M)E}{R(M)}\right) - c_E E + b_P\left(\ln(k) + z\ln M\right) - c_P. \tag{3.31}$$

Whereas the objective function of an individual country is:

$$W_i(E_i, M_i) = pE_i\left(1 - \frac{Q_i(M_i)E_i + (n-1)Q_j(M_j)E_j}{R(M_i, M_{-i})}\right) - c_E E_i + \\ {}^1\!/\!_n\, b_P\left(\ln(k) + z\ln\left(M_i + (n-1)M_j\right)\right) - c_P M_i, \qquad \forall i \in N, i \neq j. \tag{3.32}$$

The first order conditions of both problems are respectively:

$$\frac{\partial W}{\partial M} = \left(\frac{(q_o - q_M M)E}{(r_b + r_M M)}\right)\left(\left(\frac{pr_M(q_o - q_M M)E}{(r_b + r_M M)}\right) + 2pq_M E\right) - pEq_M + \frac{b_P z}{M} - c_P = 0 \tag{3.33}$$

$$\frac{\partial W_i(M_i)}{\partial M_i} = pE_i\left(q_o - q_M\left(\tfrac{n-1}{n} + M_i\right)\right) \times \\ \left(\frac{q_M E_i}{\left(r_b + r_M\left(M_i + (n-1)M_j\right)\right)} + \\ \frac{r_M\left(\left(q_o - q_M\left(\tfrac{n-1}{n} + M_i\right)\right)E_i + (n-1)\left(q_o - q_M\left(\tfrac{n-1}{n} + M_j\right)\right)E_j\right)}{\left(r_b + r_M\left(M_i + (n-1)M_j\right)\right)^2}\right) \\ - pq_M E_i\left(1 - \frac{\left(\left(q_o - q_M\left(\tfrac{n-1}{n} + M_i\right)\right)E_i + (n-1)\left(q_o - q_M\left(\tfrac{n-1}{n} + M_j\right)\right)E_j\right)}{\left(r_b + r_M\left(M_i + (n-1)M_j\right)\right)}\right) \\ + \frac{\left({}^1\!/\!_n\right)b_P z}{M_i + (n-1)M_j} - c_P = 0. \tag{3.34}$$

In these two FOC we recognize the first order conditions of the separate problems. The first order conditions with respect to effort are, of course, the same as under (3.10) and (3.18). Solving (3.10) and (3.18) for effort and substituting the results into (3.33) and (3.34) yields two fourth order polynomials that cannot be solved analytically. The derivatives with respect to parameters that can be determined with the implicit function theorem are also inconclusive on effects of parameters, unless we assume specific values. Therefore we resort to simulations for further analysis.

The effect of the combination of the games is an MPA that is intermediate to the equilibrium MPAs of the separated games. As the two separate games are both public goods games the combined game is also a public goods game. We now show the effect of these combined functions in a numerical example.

3.3. Numerical example

We now present a numerical example with two countries, as an illustration of the more general case. Suppose we have a sea that is fully claimed as EEZs and shared equally between two countries. Initial parameter settings are shown in Table 3.2. The current parameter values are arbitrary values, selected to illustrate the functioning of the model. In a sensitivity analysis we study impacts of changing these parameter values.

3.3.1. Fisheries: Full cooperation versus Nash

Solving the countries' maximization problem for M, for given parameters using (3.10) and (3.12) gives a value of 38.6% of the full sea. How this is distributed between countries is not relevant from the perspective of full cooperation. In our setting with symmetric countries a natural choice would be to share the burden of conservation equally and set aside 19.3% of the full sea.

In the non-cooperative game countries optimize their private income. As can be seen from Table 3.3 MPA size in each country is much smaller under Nash equilibrium than under full cooperation. Each country designates about 5.6% of the full sea as MPA, and hence 11.2% of the total area is protected.

As can be seen from Table 3.3, outcomes of full cooperation are much more favorable in terms of payoff for both countries than playing Nash. Full cooperation cannot be reached however, unless some bargaining solution is enforced.

Table 3.2: Arbitrary parameter values for the numerical example

Parameter		Value	Unit
p	Price	25	Euro per unit of harvest
c_E	Cost per unit of effort	5	Euro per unit of effort
r_b	Basic growth rate	0.2	-
r_M	Growth bonus MPAs	0.8	-
b_p	Benefits of protection	1	Euro per log of species number
k	Species-area curve constant	2	-
z	Species-area curve exponent	0.2	-
c_P	Costs of protection	0.25	Euro per unit of MPA

Table 3.3: Single player outcomes under full cooperation and Nash equilibrium in the separate fisheries game, the separate conservation game and the combined game

	Fisheries game		Conservation game		Combined game	
Variable	Full cooperation	Nash Equilibrium	Full cooperation	Nash Equilibrium	Full cooperation	Nash Equilibrium
M_i^*	0.193	0.056	0.4	0.2	0.2	0.08
Π_i	0.73	0.25	0	0.16	0.72	0.25
H_i	0.06	0.04	0	0.05	0.06	0.035
E_i	0.14	0.12	0	0.2	0.15	0.13
X	0.66	0.63	1	0.78	0.67	0.65
D_i	0.20	0.11	0.23	0.21	0.20	0.14
$\ln S$	0.50	0.26	0.65	0.51	0.51	0.32
W_i	0.92	0.36	0.23	0.37	0.93	0.39

3.3.2. Conservation: Full cooperation versus Nash

The conservation game for a two player game is similar to the fisheries game. However, since the response functions have a slope of -1 in the symmetric case, the reaction curves overlap completely implying an infinite number of Nash equilibria. If we require symmetry in outcomes, we are left with one Nash equilibrium. Applying formulas (3.25) and (3.28) and parameter values from Table 3.2, we get individual MPA sizes of 20% and 40% of the full sea for Nash equilibrium and full cooperation, respectively, implying total MPA sizes of 40% and 80%. The corresponding payoffs of full cooperation and Nash equilibrium are shown in Table 3.3.

From Table 3.3, it can also be seen that from a fisheries perspective the Nash equilibrium of the conservation game is preferred: under full cooperation fishing is not profitable but under the Nash equilibrium it is. This shows the advantage of considering the combined problem.

3.3.3. Combining both games

If we combine the games we get the objective functions as shown in section 3.2.3. In Table 3.3 the main results are shown to compare the full cooperation outcome with the Nash equilibrium. The results are intermediate to the values we find for the separate games except for effort. Because M^* of the combined game is slightly larger than in the fisheries game, more effort is needed due to lower catchability in the combined game.

In Figure 3.1 we show payoff of player as a function of MPA size, and we marked respective Nash and Full cooperation solutions for the separate fisheries and conservation games. Figure 3.1 also illustrates the sub-optimality of considering each issue in isolation. In this particular case the MPA that is optimal for the combined game is too large from a pure fisheries perspective and too small from a conservation perspective. It is therefore imperative to combine the two. Only in special cases, when optimal fisheries and conservation MPA sizes happen to coincide will the combined optimum be the same as in the separate games.

Another interesting finding demonstrated in Figure 3.1, is that, from society's point of view, cooperation on a single issue can be worse than playing Nash on that single issue (see also Table 3.3). Consider the payoff in the equilibrium where countries cooperate only on species conservation. We can see from Figure 3.1 that it is lower than the payoff of the Nash equilibrium of conservation. If the cooperative solution of the conservation game is applied, losses of fisheries are so large that they exceed all gains from conservation.

Whether this happens depends, of course, on parameter values, but the possibility is underlines the need of considering multiple uses in marine policy.

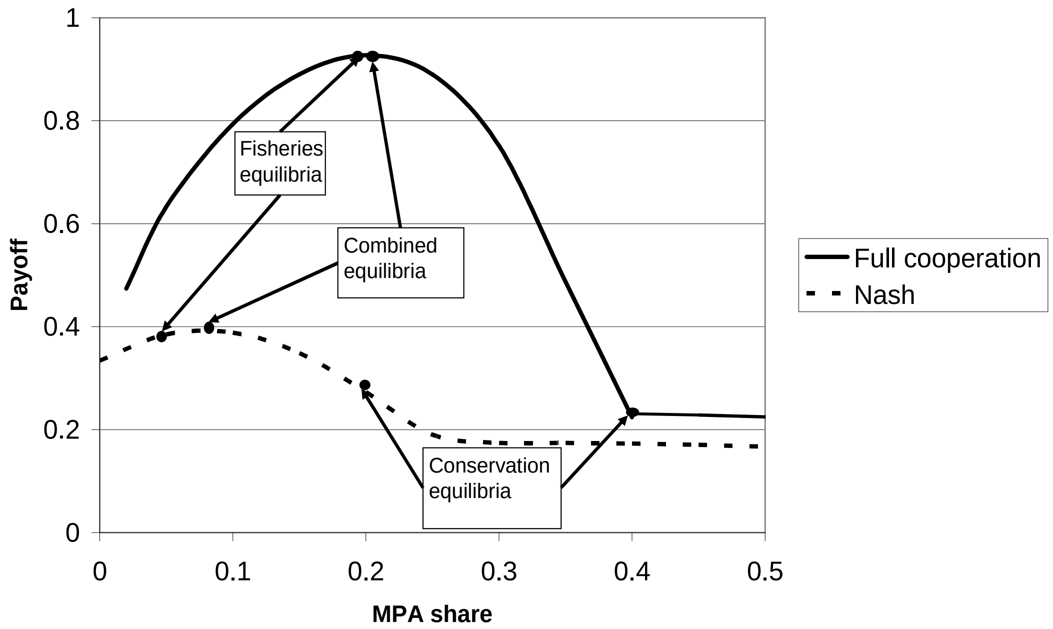


Figure 3.1: The total payoff of one country and respective equilibria of the isolated games, as a function of a) MPA size chosen by both countries (as a share of the full sea) under full cooperation, b) MPA size chosen by one country (as a share of the full sea) assuming that the other player plays Nash. For parameter values see Table 3.2.

As stated before, because externalities run in the same directions in both games, combining the games does not remove defection incentives. However, the combined Nash equilibrium gives a better total payoff than the total payoff in the separated games (compare Table 3.3).

3.3.4. Comparative statics

To illustrate the model further and to provide some more insights into incentives for cooperation we have carried out a sensitivity analysis on all parameters in the combined model. For each parameter we calculated the values of MPA sizes and the corresponding payoff for the change of a single parameter in steps of 10%, over a range of minus 50% to plus 50%, keeping other parameters fixed. We used the same procedure for the fisheries model to calculate the effect of parameters on the size of the difference between full cooperation and the Nash equilibrium because these differences could not be analyzed analytically.

Differences between full cooperation and Nash equilibrium in the fisheries game

In Figure 3.2 we show how payoff (Π^{FC} and Π^{N}) and MPA share (M^{FC} and M^{N}) change, with price of fish (p), keeping all other parameters fixed. From this

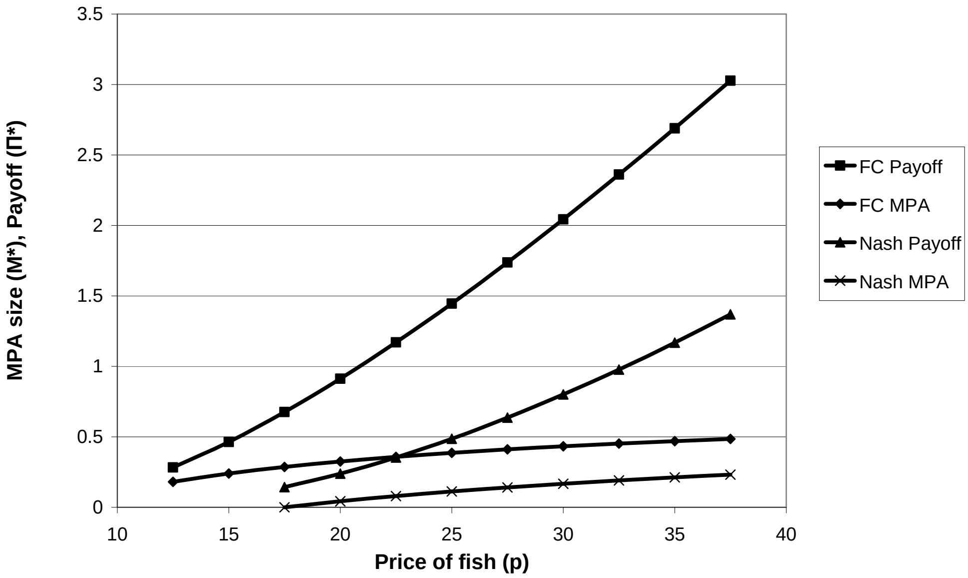


Figure 3.2: The difference between full cooperation and Nash equilibrium in payoff and MPA share in the fisheries game, as a function of changing the price of fish (p_f)

graph the differences can also be seen, and whether these differences are increasing or decreasing. How differences in payoff ($\Pi^{FC} - \Pi^{N}$) and MPA ($M^{FC} - M^{N}$) react upon changes of other parameters is shown in Table 3.4.

If the price of fish (p) increases, value of harvest increases and this drives up the payoff under full cooperation more than it drives up the payoff under Nash equilibrium, because harvests are larger under full cooperation. The incentive to set aside more area as an MPA, however, increases a bit more under Nash

Table 3.4: The relation between parameters and the difference between full cooperation and Nash equilibrium for payoff and MPA share in the separate fisheries game and the combined game

	Difference	p	c_E	r_b	r_M	q_o	q_M	b_P	z	c_P
Fisheries game	$M^{FC} - M^{N}$	-	+	+	-	-	+	0	0	0
	$\Pi^{FC} - \Pi^{N}$	+	-	+	+	-	+	0	0	0
Combined game	$M^{FC} - M^{N}$	Und.	Und.	-	Und.	Und.	Und.	-	-	+
	$W^{FC} - W^{N}$	+	-	+	+	-	+	+	+	-

Und. = Undetermined, 0 = no effect. A '+' indicates that the parameter and difference move in the same direction, a '-' indicates that the parameter and difference move in opposite directions. All results were derived from simulations.

equilibrium because all countries have a stronger incentive to set aside a larger MPA and thus the difference in total area set aside decreases.

The profitability of fisheries declines faster under Nash equilibrium than under full cooperation. At a price of 17.5 or less fishing is no longer profitable in Nash equilibrium whereas fishing still occurs under full cooperation. This is a consequence of reduced harvests in Nash equilibrium.

The effect of other parameters on the difference between full cooperation and Nash equilibrium differs. Parameter r_M has similar effects on the size of the differences as price of fish (p), and this effect is caused by the same mechanisms. A larger growth bonus (r_M) increases the incentive to set aside MPA, but larger benefits are reaped under full cooperation. As expected the cost side (c_E) has the exact opposite effect of the price of fish; the same mechanisms apply in opposite direction.

Assuming that cooperation becomes more likely when this difference is smaller, Table 3.4 shows that increasing parameter q_o has an unequivocal diminishing effect on the size of the differences. Similarly a decrease in r_b or q_M also decreases differences in payoff and equilibrium MPA size.

All these parameter changes make an MPA more attractive by reducing MPA costs in terms of forgone harvest.

Comparative statics of the combined game

The relation between parameters and payoffs and MPA sizes are shown in Table 3.5. As expected the effect of parameters on equilibrium payoff and equilibrium MPA is the same for full cooperation and Nash equilibrium, albeit sometimes larger or smaller. In Table 3.5 it is shown that increasing the benefits of an MPA (by increasing p, r_M, or b_P) or decreasing its costs (by increasing q_o or decreasing c_E or c_P) generally has the effect of increasing both equilibrium MPA and payoff. Two exceptions are basic growth rate (r_b) and curvature of the SPAR (z).

Increasing basic growth rate (r_b) decreases the incentive to set aside MPA, because the harvest forgone to acquire the same level of growth bonus (r_M) is

Table 3.5: The relation between parameters, payoffs and MPA share in the combined game both for full cooperation and Nash equilibrium

	p	c_E	r_b	r_M	q_o	q_M	b_P	z	c_P
M^*	+	-	-	+	+	-	+	+	-
W^*	+	-	+	+	+	-	+	-	-

A '+' indicates that the parameter and variable move in the same direction, a '-' indicates that the parameter and variable move in opposite directions. All results were derived from simulations.

larger, thus relatively less harvest is gained by setting aside MPA. The payoff however, does increase because overall harvest is larger.

Increasing the curvature of the species-area curve (z), on the one hand, causes MPA size to increase because the return in terms of species numbers gets higher. Payoffs on the other hand fall, because fisheries profits fall and because the log of species numbers increases less than (linear) cost increase.

In Table 3.4 we show the effect of parameters on the size of the difference in equilibrium payoff and MPA under full cooperation and Nash equilibrium. It is interesting to see that many of the fisheries parameters have an undetermined effect on the size of the difference between MPAs. What we often see here is a decreasing trend that eventually becomes negative, or the opposite: an increasing trend that starts negative but becomes positive. The difference in payoff however, has unambiguous signs: the direction of effects is the same as in the separate games.

In Figure 3.3 we show how payoff and MPA share change as a result of changing the reduction in catchability (q_M) given that all other parameters remain fixed. Increasing q_M causes both MPA and payoff to decrease, and this decrease is larger for the Nash equilibrium than for full cooperation (Figure 3.3), because a Nash player is punished double under an increase in q_M: it decreases his harvest because he can only harvest in his own EEZ and the MPA reduces harvests even further. If we once again assume that cooperation is easier when differences between payoffs are smaller, then cooperation becomes

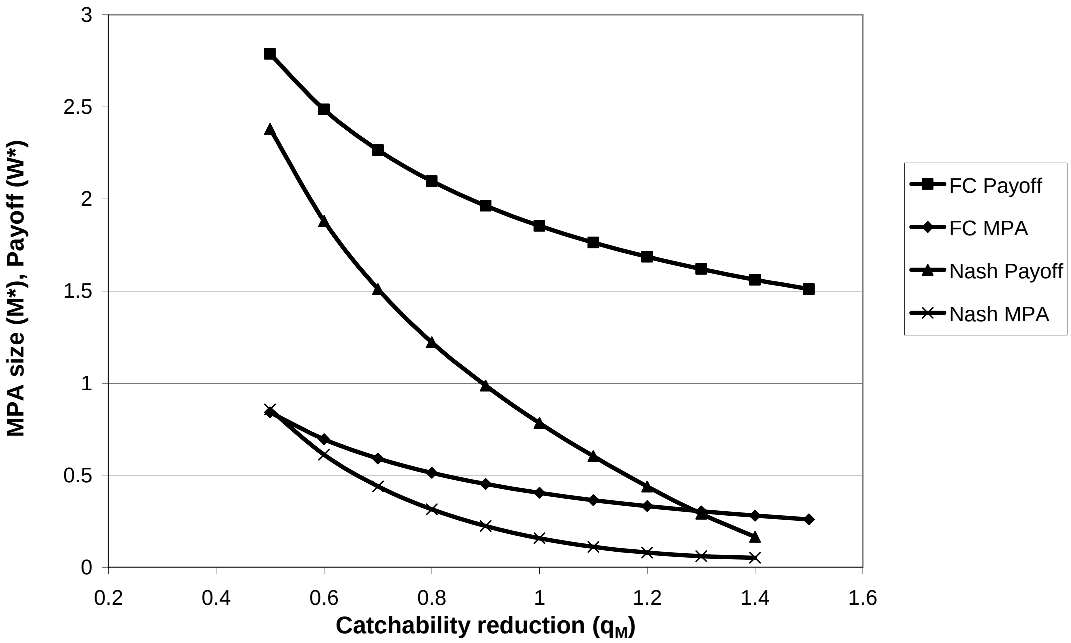


Figure 3.3: MPA and payoff under full cooperation (FC) and Nash equilibrium (Nash) in the combined game as a function of parameter q_M keeping other parameters fixed.

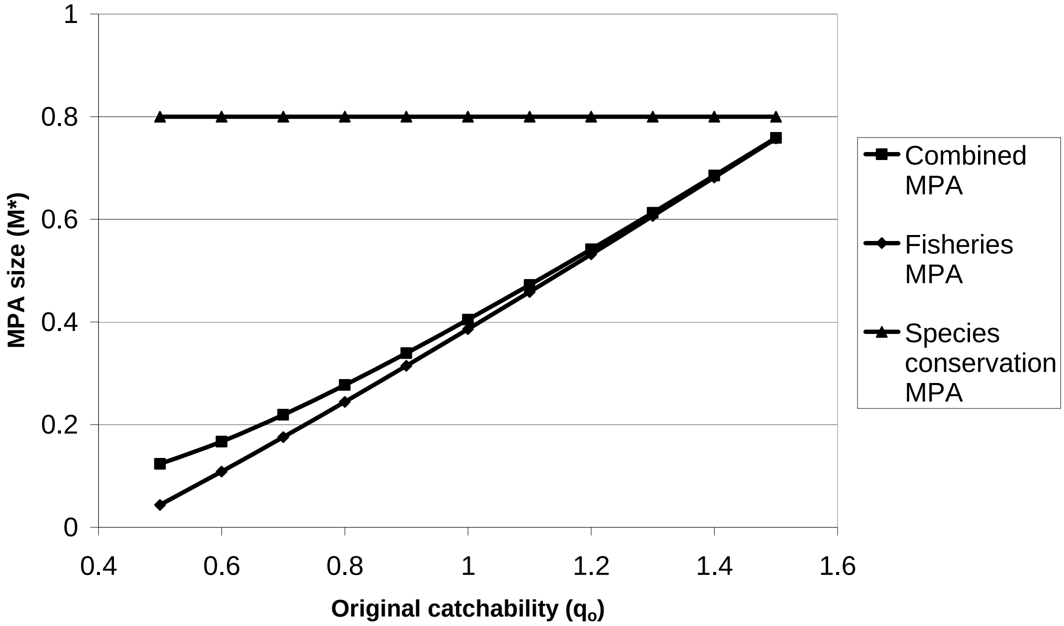


Figure 3.4: MPA size in the combined and from the separate games as a function of q_o, keeping other parameters fixed.

more likely with decreasing q_M. This can also be seen in Figure 3.3: if q_M becomes very small, payoff and MPA differences between full cooperation and Nash equilibrium decrease and, for MPA size, the difference eventually vanishes.

Figure 3.4 provides intuition on the mechanisms in the model: increasing original catchability reduces the costs of MPA from a fisheries perspective. Moreover it increases harvest and consequently payoff of fisheries. Therefore the optimum MPA size in the fisheries game goes up. The optimum of the separate conservation game is unaffected by changes in q_o. As a consequence the combined optimum goes up and approaches the conservation optimum as the fisheries optimum approaches the conservation optimum. Consequently increasing the original catchability has the expected effect of moving towards the conservation optimum.

3.4. Discussion & conclusions

3.4.1. Main findings

Here we summarize our main findings and then discuss some limitations of the study and offer suggestions for further research. In this chapter we consider effects of MPAs on fisheries and species conservation in a multi-country setting. We analyze resulting externalities and possible shifts in equilibria if two separate games of MPA assignment are combined. We focus on main issues and develop a model that contains some of the important aspects that arise in the multiple country, multiple use setting that MPA planners, and marine policy makers, in general, face.

The separate games provide novel ways to model impacts of MPAs both for conservation as well as for fisheries. Furthermore, the combination of the games provides a counterintuitive result: cooperation on a single issue may be worse than not cooperating if we take all aspects into account.

The setting of the fisheries model is such that if MPAs increase growth rates by more than the reduced harvest, it pays to set aside some area. Obviously cooperation is better than non-cooperation, because gains can be shared. Furthermore results show that the difference in MPA size assigned between cooperation and Nash equilibrium declines when either the growth bonus of the MPA increases, or when the price of fish increases. However, the payoff difference between full cooperation and Nash equilibrium also increases with these parameters which leads to stronger free-rider incentives. If the catchability reduction of the MPA increases, the difference in both assigned MPA and payoff decline. Essentially if MPAs become worth less, differences between full cooperation and Nash equilibrium decrease.

The conservation game offers a new approach to the conservation problem. Although game theory has been used to analyze transboundary parks in general (Busch, 2008), to our knowledge species-area curves were never used in a game. The results of the game are straightforward: for species conservation cooperation is better than Nash equilibrium, and due to defector incentives we have a social dilemma. In contrast with the fisheries game, the difference in MPA assigned as well as the difference in payoff between full cooperation and Nash equilibrium are increasing in both gains of the MPA and number of players. Therefore it seems that achieving cooperation will be even harder.

A core result of the chapter is that the combined game offers a new and counterintuitive perspective on these standard results of the separate games: if we take a combined view, cooperation on a single issue may be worse than non -cooperation. By ignoring multiple use of the MPA the damage done in one domain of use is so large that it destroys all gains from cooperation on the other issue. Therefore we conclude that accounting for multiple uses is a necessity when planning MPAs.

Furthermore the combined game has better possibilities from society's perspective than the single games do. In our numerical example the combined game increases MPA assigned compared to the fisheries case. Also it decreases the difference in MPA between full cooperation and Nash equilibrium compared to the conservation case. Still, combining both games gives a public goods game and free-rider incentives remain.

Summarizing, MPAs are a valuable tool for conservation and fisheries management in EEZs. Optimal use of this tool, however, requires consideration of its multiple use effects. Failing to consider multiple uses could, from society's point of view, be worse than not cooperating at all.

Although not a perfect solution, the MSD (and other European marine policies) encourages countries to cooperate and more importantly, to consider the full effects of their actions in their marine strategies. We conclude that MPAs may not be a panacea, but they will surely support safeguarding our marine environment.

3.4.2. Limitations of the study

Our analysis assumes symmetric players. In the issue linkage models of Folmer et al. (1993) and Cesar and de Zeeuw (1996) asymmetry is the key to break the prisoner's dilemma. Their models consist of two prisoner's dilemmas that have a reversed asymmetric payoff structure. The linkage of the two games removes the prisoner's dilemma payoff structure. In our model asymmetry does not have a decisive role since our model considers only one strategic decision variable, in contrast to the models of Folmer et al. (1993) and Cesar and de Zeeuw (1996) who consider two strategic decisions, one for each game. Even if one country would have a higher advantage in fisheries and the other in species conservation we expect that the combined game would still give a result intermediate to the equilibria of the two separate games. Where that equilibrium would be, depends on parameter values.

A further limitation of the assumed symmetry and uniform biological conditions in the model is that all spatial heterogeneity is ignored, whereas it is exactly this spatial heterogeneity both in fishermen behavior and in biological conditions that determine the effectiveness of MPAs. Regarding fishermen behavior, spatial heterogeneity in location choice and home ports decisions may largely negate any positive fisheries effects (Smith, 2004; Smith and Wilen, 2004). The biological heterogeneity may have the same effect (Holland, 2000), although positive effects have also been found when a patchy environment is considered (Sanchirico and Wilen, 2002; Sanchirico, 2004). Effects very much depend on assumptions about migration and growth rates (Sanchirico and Wilen, 2002; Sanchirico, 2004; Ruijs and Janmaat, 2007).

From the species conservation perspective biological heterogeneity translates into hotspots and species-poor areas, resulting in different SPARs. There is also the question whether species richness per se is desirable or that different species are valued differently, with different species located in different areas. Furthermore due to overlaps in protected species between MPAs and countries, questions arise whether species protected in one country should also be protected in the other.

Although the fisheries model suggests a positive role for MPAs in fisheries management, this result may no longer hold if open access is present, as demonstrated in Hannesson (1998) and Anderson (2002). Even though agents may play non-cooperatively in our model, resulting in a sub-optimal solution, we do not allow for open access, because the sea is partitioned into EEZs in our model.

3.4.3. Suggestions for further research

In our model the fisheries game does not include monitoring and compliance costs. A more realistic setting would include a set-aside cost in the fisheries case, equal to the one in the conservation case. The most likely outcome of such a setting would be a smaller MPA in the separate fisheries game, because of extra costs. The MPA for conservation and combined game would not change, as costs already occur once in both games. Such a double dividend scenario would further demonstrate the need to consider all possible uses in MPA policies.

As pointed out in the introduction, MPAs are a static tool in a dynamic environment. In further research it will be useful to explicitly include issues of time and space in addition to the steady state analysis of the current chapter. The importance of spatial dynamics has been demonstrated by e.g. Ruijs and Janmaat (2007), but just for the fisheries case and to our knowledge never for the combination of fisheries and conservation.

The evolution of species richness over time and space, is very complex and a constant source of debate among ecologists (see Gray, 2001 for a critical review for the marine environment). The current approach is therefore a pragmatic compromise, but further empirical and theoretical work on this issue would greatly increase our understanding of the functioning of MPAs and facilitate inclusion of conservation issues in the MPA and fisheries modeling debate.

3.A. Appendix: Derivatives with respect to parameters

We use the implicit function theorem and relevant first order conditions to evaluate the effect of parameters on the equilibrium MPA. The equilibrium value of other variables enters the relevant FOC, before the derivative is determined.

If we substitute E^* calculated from (3.10) in first order condition (3.12) and rearrange we get:

$$\frac{\left(p\left(q_M M - q_o\right) + c_E\right) \times \left(p q_M^2 r_M M^2 - \left(q_M r_M \left(2pq_o + c_E\right)\right) M + q_o r_M \left(pq_o - c_E\right) - 2c_E q_M r_b\right)}{4p\left(q_M M - q_o\right)^3} = 0. \tag{3A.1}$$

In the main text we pointed out that we are only interested in the solution stemming from the quadratic part, since the rest are corner solutions. For (3A.1) to be equal to zero, this is the part that equals zero, and only the effect of parameters on this part has an effect on the solution we are interested in. Therefore we can determine the effect of parameters on equilibrium MPA, by taking derivatives only on the part:

$$\left(p q_M^2 r_M M^2 - \left(q_M r_M \left(2pq_o + c_E\right)\right) M + q_o r_M \left(pq_o - c_E\right) - 2c_E q_M r_b\right) = 0. \tag{3A.2}$$

This results in the following derivatives:

$$\frac{\partial M}{\partial p} = \frac{-\left(q_M M - q_o\right)^2}{q_M \left(2p\left(q_M M - q_o\right) - c_E\right)} \tag{3A.3}$$

$$\frac{\partial M}{\partial c_E} = \frac{r_M \left(q_M M + q_o\right) + 2q_M r_b}{r_M q_M \left(2p\left(q_M M - q_o\right) - c_E\right)} \tag{3A.4}$$

$$\frac{\partial M}{\partial r_M} = \frac{c_E \left(q_M M + q_o\right) - p\left(q_M M - q_o\right)^2}{r_M q_M \left(2p\left(q_M M - q_o\right) - c\right)} \tag{3A.5}$$

$$\frac{\partial M}{\partial r_b} = \frac{2c_E}{r_M \left(2p\left(q_M M - q_o\right) - c_E\right)} \tag{3A.6}$$

$$\frac{\partial M}{\partial q_M} = \frac{c_E \left(r_M M + 2r_b\right) - 2p r_M M \left(q_M M - q_o\right)}{r_M q_M \left(2p\left(q_M M - q_o\right) - c_E\right)} \tag{3A.7}$$

$$\frac{\partial M}{\partial q_o}=\frac{2p\left(q_M M-q_o\right)+c_E}{q_M\left(2p\left(q_M M-q_o\right)-c_E\right)}. \tag{3A.8}$$

The procedure for the derivatives in Nash equilibrium is similar: calculate E_i^* from equation (3.17), insert it into equation (3.19), both for E_i^* and E_j^*, replace M_j^* with M_i^*, and take derivatives with respect to the quadratic part of the resulting cubic equation:

$$\frac{n\left(npq_o-pq_M\left(nM_i+n-1\right)-nc_E\right)\times\begin{bmatrix}n^2pq_M^2r_MM_i^2-nq_Mr_MM_i+pr_M\left(nq_o-nq_M+q_M\right)^2\\-c_En\left(\left(nq_o-(n-1)q_M\right)r_M+(n+1)q_Mr_b\right)\end{bmatrix}}{p(n+1)^2\left(nq_o-nq_MM_i-nq_M+q_M\right)^3}=0. \tag{3A.9}$$

Hence derivatives are taken with respect to the implicit function that defines the interior solution in the main text:

$$\begin{aligned}n^2pq_M^2r_MM_i^2-nq_Mr_MM_i+pr_M\left(nq_o-nq_M+q_M\right)^2\\-c_En\left(\left(nq_o-(n-1)q_M\right)r_M+(n+1)q_Mr_b\right)=0.\end{aligned} \tag{3A.10}$$

This results in the following derivatives:

$$\frac{\partial M_i}{\partial p}=\frac{-\left(nq_MM_i-q_M-nq_o+nq_M\right)^2}{nq_M\left(2p\left(nq_MM_i-q_M-nq_o+nq_M\right)-n^2c_E\right)} \tag{3A.11}$$

$$\frac{\partial M_i}{\partial c_E}=\frac{\left(r_M\left(n^2q_MM_i+n\left(q_o-q_M\right)+q_M\right)+(n+1)q_Mr_b\right)}{nq_Mr_M\left(2p\left(nq_MM_i-q_M-nq_o+nq_M\right)-n^2c_E\right)} \tag{3A.12}$$

$$\frac{\partial M_i}{\partial r_M}=\frac{c_En\left(n^2q_MM_i+nq_o-nq_M+q_M\right)-p\left(nq_MM_i-nq_o+nq_M-q_M\right)^2}{nq_Mr_M\left(2p\left(nq_MM_i-q_M-nq_o+nq_M\right)-n^2c_E\right)} \tag{3A.13}$$

$$\frac{\partial M_i}{\partial r_b}=\frac{(n+1)c_E}{nq_Mr_M\left(2p\left(nq_MM_i-q_M-nq_o+nq_M\right)-n^2c_E\right)} \tag{3A.14}$$

$$\frac{\partial M_i}{\partial q_M} = \frac{\begin{bmatrix} c_E n\left(nr_M\left(nM_i - 1\right) + r_M + \left(n+1\right) r_b\right) \\ -2pr_M\left(nM_i + n - 1\right)\left(nq_M M_i - q_M - nq_o + nq_M\right) \end{bmatrix}}{nq_M r_M\left(2p\left(nq_M M_i - q_M - nq_o + nq_M\right) - n^2 c_E\right)} \tag{3A.15}$$

$$\frac{\partial M_i}{\partial q_o} = \frac{2p\left(nq_M M_i - q_M - nq_o + nq_M\right) + nc_E}{nq_M\left(2p\left(nq_M M_i - q_M - nq_o + nq_M\right) - n^2 c_E\right)}. \tag{3A.16}$$

Chapter 4*: Marine Protected Areas for biodiversity conservation: cooperation, strategic behavior or conservation autarky?

4.1. *Introduction*

Marine Protected Areas (MPAs) have received growing attention as a tool for marine management. In fact some authors claim that Marine Protected Areas are a panacea (Bohnsack, 1993). Others, such as Hannesson (1998) and Anderson (2002) are more critical of the role of MPAs, especially under conditions of open access or when they are compared to first-best solutions. Alisson et al. (1998) point out that due to the open nature of the marine environment MPAs do not grant protection from mobile threats such as pollution, oil spills and invasive species.

Part of the debate is due to different meanings and purposes that people have in mind when using the term "MPA" (Jones, 2002). The term "protected" in "Marine Protected Area" begs the question: protected from what? In the case of fisheries management it would mean "protected from fishing", which would refer to a no-take zone. If "protected" is meant in a stricter sense, such as protection from all human activities, the term "marine reserve" is more appropriate. Lastly, "protected" sometimes refers to limited human use, e.g. banning fishing but allowing other activities such as offshore wind parks. In this case MPAs are a zoning tool.

We will interpret MPAs as fully protected zones, with biodiversity conservation as main goal, but with positive spill-over effects on direct uses of the ecosystem such as fisheries. The model developed thus reflects the multiple-use nature that exists in ocean space and that should be considered when making decisions about MPAs (Punt et al., 2010).

The Exclusive Economic Zones (EEZs) of countries are heterogeneous in terms of biodiversity, and therefore both location and size of an MPA are important. Location plays not only a role for biodiversity, but also has strategic implications: given that some biodiversity is protected elsewhere (e.g. in another country), can additional value be obtained by protecting biodiversity at a specific site? Essentially the question is whether biodiversity protected in other locations is a complement to, or substitute for biodiversity protected here. Moreover, different locations face different opportunity costs. Due to the

*This chapter is based on the paper: Maarten J. Punt, Hans-Peter Weikard, Ekko C. van Ierland, Jan H. Stel. Marine Protected Areas for biodiversity conservation: cooperation, free-riding or conservation autarky? Submitted.

combination of complement-substitute issues and differing costs and benefits for protecting different locations, plenty of opportunities arise for countries to cooperate or defect on decisions on size and location of MPAs.

MPAs have been modeled in several ways over the last decades; e.g. Sumaila (1998, 2002), Anderson (2002), Beattie et al. (2002), Boncoeur et al. (2002), Smith and Wilen (2003), Dalton (2004), Sanchirico (2004), Schnier (2005a,b), Armstrong (2007), Ruijs and Janmaat (2007), Kar and Matsuda (2008), Ngoc (2010) and Punt et al. (2010). Most models consider MPAs as a fisheries management tool and other uses are ignored (exceptions are Boncoeur et al., 2002, Dalton, 2004, Ngoc, 2010 and Punt et al., 2010). If multiple uses are accounted for, they focus on a single ecosystem and are generally non-spatial. Many countries however, have EEZs that comprise a number of ecosystems providing multiple direct and indirect services. Moreover, only a few papers, such as Sumaila (2002), Ruijs and Janmaat (2007) and Punt et al. (2010), consider strategic effects at country level.

In this chapter we will formulate a spatial game theoretic model to analyze combined problems of complement-substitute issues, location differences in costs and multiple use benefits of MPAs. The setup of our model is similar to economic models on spatial configuration of terrestrial reserves. These models build to a large extent on traditional industrial organization literature. Goeschl and Igliori (2004) investigate competition between an extractive reserve and forest plantation in a linear spatial setting. They formulate conditions under which the reserve can co-exist with the plantation. Ando and Shah (2010) study where a single reserve should be located on a line if the valuation of a reserve decreases with distance. Albers et al. (2008) have shown how preferences for agglomeration influence the spatial configuration pattern when two agents consider strategic site purchases.

Similar work on complement-substitute issues has been carried out by conservation biologists for terrestrial reserves (e.g. Rodrigues and Gaston, 2002, Bladt et al., 2009, Kark et al., 2009, Jantke and Schneider, 2010 and Bode et al., 2011) but these papers compare full cooperation on conservation among a number of countries with a situation where countries designate reserves without considering conservation efforts by others. Following the trade literature we call this setting “conservation autarky”.

What are plausible motives for conservation autarky? Several reasons may exist: countries may value protection in their own domain higher because they do not trust the protection in the other country, they may simply not know about species being protected in another country or they may feel it is their duty to protect species on moral grounds. Moreover, comparing full cooperation with conservation autarky is looking at the same problem at different scales, where full cooperation represents a regional scale, involving two or more countries. Conservation by independent countries however, may result in “location leakage”, i.e. countries do less because a species is protected by another country, even though from the global perspective they should do more.

Our chapter is the first to describe strategic and economic considerations for biodiversity conservation with MPAs in a spatial context, with the complement-substitute problem explicitly incorporated. Furthermore, we are the first to explicitly compare a setting of strategic interaction with a setting of conservation autarky in a multiple-use environment. Bode et al. (2011) analyze two trusts buying land parcels under conservation autarky, strategic behavior and cooperation, but their agents have differing conservation objectives.

We find that on the one hand conservation autarky implies an inefficiency from global perspective, because conservation efforts by others are ignored. On the other hand conservation autarky eliminates location leakage. Therefore for the global society conservation autarky is still preferred to the alternative scenario in which countries do not ignore others’ conservation efforts and exploit each other through leakage.

4.2. Model characteristics

4.2.1. Model background

We formulate a spatial game theoretic model of MPAs for biodiversity conservation, where two countries decide on locations and sizes of MPAs. An MPA generates ecological benefits, but it also has opportunity costs and costs of implementation and enforcement.

The value of biological benefits in an area can be disaggregated into direct use values and indirect use values that stem from different ecosystem services. These services for the marine environment have been classified by e.g. Beaumont et al. (2007). At ecosystem level direct values stem from production ecosystem services, such as provision of food and raw materials, local regulation services, such as bioremediation of waste, and over-arching support services, such as biologically mediated habitat. Indirect use values stem from cultural services, such as warm glow, and option-use services (Beaumont et al., 2007).

We would argue that there is another scaling issue here: direct use values are mainly benefiting local exploiters and are less dependent on biodiversity per se. Indirect use values, in contrast, are benefiting global society and are heavily dependent on biodiversity.

If we focus on ecosystems and their services, dissimilarity between ecosystems seems a natural starting point for measuring biodiversity. Weikard (2002) suggests to take the number of species *not* in common between two ecosystems as a measure of this dissimilarity. Maximizing biodiversity then boils down to selecting the largest number of unique species by protecting relevant ecosystems as reserves.[1] However, we can never be sure that a species can be fully save. Therefore we do not aim to maximize the total number of species, but the expected total number of species as in Polasky et al. (2000), Camm et al. (2002) and Arthur et al. (2004).

If this ecosystem dissimilarity approach is used in combination with biodiversity maximization subject to a budget constraint, the full problem is equal to the Maximum Species Coverage Problem (Church et al., 1996; Ando et al., 1998; Polasky et al., 2000, 2001a,b; Camm et al., 2002; Arthur et al., 2004). If it is used in combination with a cost-effectiveness approach for site selection subject to a biodiversity constraint the full problem is equal to the Minimum Set Cover problem (Williams and Araéjo, 2000; Sala et al., 2002; Stewart et al., 2003; Cabeza et al., 2004; Richardson et al., 2006). Both problems are the main building blocks of reserve site selection problems. In our framework however, we opt to use a benefit function approach, maximizing the net benefits of conservation, i.e. benefits from direct and indirect ecosystem services minus opportunity costs. Moreover, our model is generally applied to a larger scale. Reserve site selection problems mostly consider multiple sites within one ecosystem. In contrast, our "sites" are full ecosystems that can be partly protected.

An important issue when maximizing the expected number of species is that the survival probability of one species may depend on the survival probability of other species (Mainwaring, 2001; van der Heide et al., 2005). However, Weikard (2002) argues that, because ecosystems are by definition stand-alone entities, probabilities that ecosystems stay intact are independent.

To be applicable in our framework we have to develop the ecosystem dissimilarity approach (Weikard, 2002) further as it lacks three essential features:

- (i) An explicit specification of probabilities of ecosystems to stay intact (henceforth: persistence probabilities)
- (ii) An explicit specification of direct use values
- (iii) An explicit specification how biodiversity behaves in space

Therefore we introduce the following assumptions to amend it:

- (a1) Persistence probability of an ecosystem is directly related to the size of protected area in an ecosystem.
- (a2) Each additional unit of protected area in an ecosystem increases the direct use values of ecosystems services by improving their quality.

[1]Biologists generally measure biodiversity with indices based on relative abundance and species richness (e.g. Hill, 1973). Others have proposed to measure biodiversity based on dissimilarity between species (Faith, 1992; Weitzman 1992, 1993, 1998; Solow et al., 1993). Indices that combine dissimilarity, abundance and species richness also exist (Ricotta, 2004; Weikard et al., 2006b).

(a3) Dissimilarity between ecosystems increases with distance between those ecosystems.

The relation between persistence probability of an ecosystem and size of protected area (assumption (a1)) stems from the fact that with increasing protected area an increasing number of keystone species, the building blocks of the ecosystem, are protected. Without protection of these keystone species the essential features of the ecosystem would be lost, leading to a regime shift in the ecosystem. The system would then be transformed into another ecosystem, that would still support some species, but often such a new ecosystem has less value than the original one. Once transformed, it is very hard to get the ecosystem back to its original state (Folke et al., 2004). An example is the Waddensea area along the Dutch, German and Danish coastline. If it would be impacted too much by human activities such as fishing, dredging and oil and gas exploration, its keystone habitats, sandy bottoms and mud flats, would be destroyed and most species would be lost. If a large MPA would be installed the ecosystem would be protected from damaging activities in this area and the probability of ecosystem destruction becomes smaller. In fact the area is currently protected by a tri-lateral agreement between the countries involved.

The habitats sustained by keystone species are not just building blocks of the ecosystem, they are also the main contributors to direct use values and direct ecosystem services, either by providing a home to species that provide these services or by forming the basis for the service itself (assumption (a2)). An example of the former is fish habitat, whereas an example of the latter is sandy beaches providing tourism services. As the MPA gets bigger, more keystone species are protected and direct services have a higher quality. In the case of fishing, for example, protection of fish habitat could improve the growth rate and carrying capacity of the fish stocks. Of course, harvest would be restricted to a smaller area which is reflected in the cost function of MPAs.

The reasoning behind assumption (a3) is that many ecosystems are located along environmental gradients. Species that are very common occur in (nearly) all ecosystems. Other species occur only in specific habitats of ecosystems and, as we go on along the gradient, their specific habitats occur and disappear.

4.2.2. A biodiversity conservation model

In our MPA model we consider two countries that share a common sea. This common sea comprises the set of E ecosystems. Ecosystems are characterized by a number of keystone habitats and species, and their destruction would imply the destruction of the ecosystem. In all ecosystems together a set S of species exists, consisting of $|S|$ species, denoted i. Each ecosystem $e \in E$ is characterized by a subset $s_e \subseteq S$ of species, and consequently each of the i species occurs in a subset $N_i \subseteq E$ ecosystems. The intersection of two sets of species in ecosystems e and e', $s_e \cap s_{e'}$, decreases with distance. In this chapter we will use a simple exponential decay function to describe this relationship:

$$X = X_o e^{-r\delta} \tag{4.1}$$

with X describing the set intersection, X_o the set intersection between two neighboring areas, r the decay rate and δ the distance between the two ecosystems under consideration (Nekola and White, 1999). Implicitly this assumes a uniform decay gradient in all directions.

We can think of this setting as a long coastline shared by two countries where a natural gradient such as temperature, benthos conditions or salinity, defines the ecosystems occurring along this coastline, such as the coastline of the US and Canada. Alternatively, we can think of a transect between the coasts of two countries where an increasing depth gradient defines the occurring ecosystems.

As an example of a gradient consider the Dutch coast as starting point. The Waddensea area would then be the first ecosystem. The next ecosystem along the gradient would be the North Sea itself, and we would end up in the Arctic sea as the third ecosystem. The gradient is then defined by a combination of depth and temperature. This gradient is of course not as universal decaying as described in our stylized model, but it does capture the idea.

The boundaries between ecosystems are in reality not rigid, and interaction will occur at the edges of ecosystems. Moreover, there may be highly migratory species that travel between ecosystems, such as tuna and whales. These species, however, although of great importance to certain ecosystems, are exceptions. We will assume throughout the chapter that ecosystems are independent units, large enough to contain the home ranges of most species occurring in those ecosystems, and ignore those species that traverse multiple ecosystems. Species can occur transboundary, but are assumed to be independent populations.

The gradient defines the distribution of species over ecosystems and as such is the most important factor determining the spatial configuration of MPAs. Along a line, this uniformly decaying gradient will result in two ecosystems on the edges with a relatively large number of unique species. The ecosystems between those two edges all have a lower number of unique species, as a number of their species are shared with their neighbors.

We consider two countries that wish to designate one or several protected areas, so as to maximize net benefits of conservation. The benefits stem from direct ecosystem services that have non-increasing marginal returns in area and only accrue to the assigning country, and from indirect ecosystem services that stem from biodiversity and accrue to both countries. Hence the game is in part a public goods game.[2]

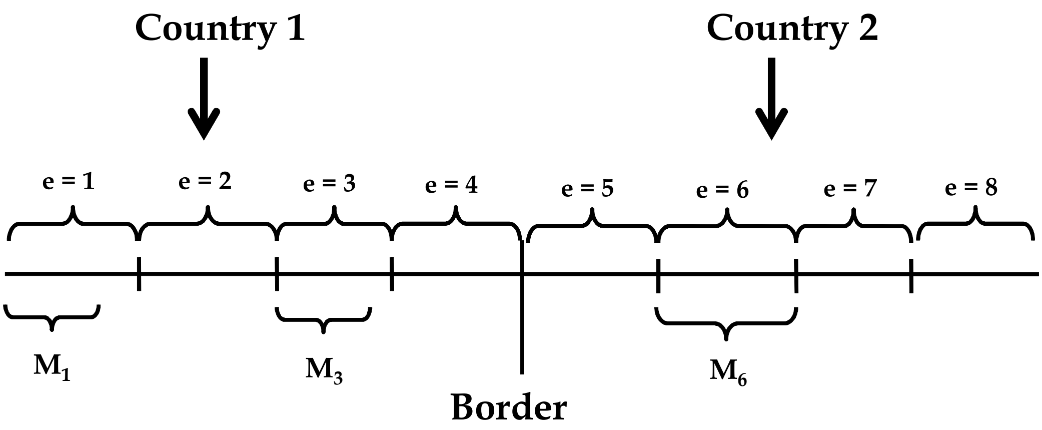


Figure 4.1: A schematic presentation of the nature protection model. Each of the two countries 1 and 2 own four ecosystems. In every ecosystem a discrete MPA of size $0 \leq M_e \leq 1$ can be designated. Ecosystem 1 and 3 in country 1 are partially protected, ecosystem 6 in country 2 is fully protected.

Both countries own half of the sea. Each country k = 1,2 can decide to protect a share M_e $(0 \leq M_e \leq 1)$ of ecosystem e, i.e. M_e is set apart as an MPA. See Figure 4.1 for a schematic representation of this model.

Political boundaries do not always coincide with ecosystem boundaries and consequently many countries share ecosystems as well as species. This gives rise to additional transboundary problems such as shared stock problems and MPA location and size problems. These problems have been considered in detail by e.g. Hannesson (1998), Ruijs and Janmaat (2007), Punt et al. (2010) and Pintasilgo et al. (2010). The focus of this chapter is more on shared species in different ecosystems and therefore we simplify the model by assuming ecosystem boundaries coincide with political boundaries.

The MPA model with its ecosystems is in fact rather similar to Hotelling's well-known firm location model, and the terrestrial conservation literature that builds on that specific model (e.g. Goeschl and Igliori, 2004, Albers et al., 2008 and Ando and Shah, 2010). The important differences with Hotelling's model are the number of firms (in our model the number of MPAs). In our model this number is endogenous, whereas in the Hotelling model and its extensions it is given. Similarly in our model the preferences for agglomeration follow from

[2]A public goods game is a game in which all players can contribute to a public good. In such a game it is in each player's individual interest to contribute less than the social optimum to a public good. Examples include greenhouse gas mitigation (e.g. Finus, 2003 and Weikard et al., 2006a) and catch restrictions on transboundary fish stocks (e.g. Pintassilgo 2003, Pintassilgo and Lindroos, 2008 and Pintassilgo et al., 2010).

the (perceived) species distribution, whereas in the model of Albers et al. (2008) these are given.

4.2.3. Full cooperation among countries

Under full cooperation countries maximize the value V of their joint net benefits. Countries benefit directly from preserving ecosystems and indirectly through biodiversity. In effect they maximize:

$$V(M_e) = \sum_{e \in E} D_e(M_e) + B\left(\sum_{i \in S} \pi_i\right) - \sum_{e \in E} C_e(M_e) \tag{4.2}$$

with D_e the direct benefits from protecting area M_e in ecosystem $e \in E$, B are the indirect benefits from biodiversity, and $\sum_{e \in E} C_e(M_e)$ is cost of protection. $\sum_{i \in S} \pi_i$ is the total expected number of species. It is calculated in the following way: persistence probability of a single ecosystem e is a function of the area protected in that ecosystem, M_e. We assume this function follows a certain probability distribution that it is increasing in M_e:

$$\text{Prob}(e \text{ persists}) = f(M_e), \quad f'(M_e) \geq 0, \quad \forall e \in E \tag{4.3}$$

If species i occurs in ecosystem e, then the probability that species i goes extinct in ecosystem e, because ecosystem e is replaced by another ecosystem, is one minus the probability that ecosystem e persists: $1- f(M_e)$. Species i is found in the subset of ecosystems N_i, hence the probability that species i goes extinct in all N_i ecosystems is equal to: $\prod_{e \in N_i}(1 - f(M_e))$.

The survival probability of a species is then equal to the probability that it does *not* go extinct in all its ecosystems:[3]

$$\pi_i = 1 - \prod_{e \in N_i}(1 - f(M_e)) \quad \forall i \in S. \tag{4.4}$$

The sum of all survival probabilities of species is the expected number of species. In our model, in contrast to the model approaches by Polasky at al. (2000), Camm et al. (2002) and Arthur et al. (2004) the survival probability of a species in a specific ecosystem is dependent on the persistence probability of that ecosystem, which is in turn determined by the MPA size.

Substituting (4.4) in (4.2) the maximization problem is:

[3]The *survival* probability of a *species* is consequently a function of the *persistence* probabilities of the *ecosystems* it occurs in.

$$V(M_e) = \sum_{e \in E} D_e(M_e) + B\left(\sum_{i \in S}\left(1 - \prod_{e \in N_i}(1 - f(M_e))\right)\right) - \sum_{e \in E} C_e(M_e). \tag{4.5}$$

The first order condition (FOC) for an interior solution is:

$$\frac{\partial V}{\partial M_e} = \sum_{e \in E} \frac{\partial D_e(M_e)}{\partial M_e} + \frac{\partial B\left(\sum_{n \in S}\left(1 - \prod_{e \in N_e}(1 - f(M_e))\right)\right)}{\partial M_e} - \sum_{e \in E} \frac{\partial C(M_e)}{\partial M_e} = 0, \quad \forall e \in E. \tag{4.6}$$

This problem can be solved analytically for small problems with only a few ecosystems and species, and simple specifications of benefit, cost and probability functions. It gets complicated rapidly when large numbers of variables and parameters are involved.

As a simple illustrative example suppose we have three ecosystems, where each ecosystem has one species in common with its neighbor as in Table 4.1. As emphasized before, there is no migration between ecosystems and we are dealing with independent ecosystems. Furthermore assume linear benefits (d_e), quadratic costs with cost parameter c_e, and persistence probabilities linear in protected area. The net benefit function is then:

$$\begin{aligned} V(M_e) &= d_1 M_1 + d_2 M_2 + d_3 M_3 + \\ &\big((1-(1-M_1)) + (1-(1-M_1)(1-M_2)) + (1-(1-M_2)(1-M_3)) + (1-(1-M_3))\big) \\ &\quad -\tfrac{1}{2}c_1 M_1^2 - \tfrac{1}{2}c_2 M_2^2 - \tfrac{1}{2}c_3 M_3^2. \end{aligned} \tag{4.7}$$

The relevant FOC are:

Table 4.1: Example of ecosystem configuration

Ecosystem	Species
e = 1	1,2
e = 2	2,3
e = 3	3,4

$$
\begin{aligned}
&\frac{\partial V}{\partial M_1} = 0 \Leftrightarrow d_1 + \left(2 - M_2\right) = c_1 M_1 \\
&\frac{\partial V}{\partial M_2} = 0 \Leftrightarrow d_2 + \left(2 - M_1 - M_3\right) = c_2 M_2 \\
&\frac{\partial V}{\partial M_3} = 0 \Leftrightarrow d_3 + \left(2 - M_2\right) = c_3 M_3 .
\end{aligned} \tag{4.8}
$$

The first order conditions show the standard economic reasoning. On the left hand side of the equations we see marginal benefits of an additional unit of MPA: the marginal benefits of direct use (d_e) and the marginal change in the expected number of species. On the right hand side we see marginal costs of an additional unit of MPA, $c_e M_e$.

We also see the effect of the distribution of species on the optimal location and size of MPAs. Both species 2 and 3 can be protected in multiple ecosystems, hence the negative effect of M_2 in the FOC's for M_1 and M_3 and vice-versa.

In an interior solution countries will always assign at least a small MPA in all ecosystems. This can be seen from (4.8): an interior solution requires $0 < M_e < 1 \quad \forall e$, consequently the $2 - M_e$ part is always positive and we get a positive MPA for each ecosystem.

Solving for M_1, M_2 and M_3 simultaneously we get:

$$
\begin{aligned}
M_1 &= \frac{d_1\left(c_2 c_3 - 1\right) + c_3\left(2c_2 - d_2\right) + d_3 - 2c_3}{c_1 c_2 c_3 - c_1 - c_3} \\
M_2 &= \frac{c_1 c_3\left(d_2 + 2\right) - d_1 c_3 - d_3 c_1 - 2c_1 - 2c_3}{c_1 c_2 c_3 - c_1 - c_3} \\
M_3 &= \frac{d_3\left(c_1 c_2 - 1\right) + \left(2c_2 - d_2\right)c_1 + d_1 - 2c_1}{c_1 c_2 c_3 - c_1 - c_3} .
\end{aligned} \tag{4.9}
$$

As can be seen from the solutions the effect of parameters is ambiguous and depends on the value of other parameters, in particular cost parameters. If the denominator is positive and the product of cost parameters is larger than 1, direct benefits of an ecosystem (the *d*'s) increase the MPA in that ecosystem. Interestingly, there is also a positive effect of direct benefits in ecosystem 1 on the MPA size chosen in ecosystem 3 and vice-versa, whereas the direct benefit parameter of ecosystem 2 affects MPA sizes in the other two ecosystems in a negative way. This occurs because ecosystem 2 is partly a substitute for the other two. If M_1 increases, M_2 decreases causing a protection loss to some of the species in ecosystem 3 where consequently M_3 has to be raised to make up for this loss.

The cost parameters determine the sign of the denominator; if the denominator is positive they decrease the MPA in their own ecosystem. Furthermore it can be seen that c_2 has a positive effect on the numerators and denominators of M_1 and M_3. Whether an increase in c_2 thus increases these two MPAs depends on relative parameter values.

Because survival probabilities are linear in MPA size, the resulting survival probabilities of the different species are:

$$\begin{aligned}
\pi_1 &= M_1 \\
\pi_2 &= M_1 + M_2 - M_1 M_2 \\
\pi_3 &= M_2 + M_3 - M_2 M_3 \\
\pi_4 &= M_3,
\end{aligned} \tag{4.10}$$

with the respective M_e given in equation (4.9). Survival probabilities of species 1 and 4 are equal to the MPA size in ecosystem 1 and 3, because they only occur in those ecosystems. Species 2 and 3 have a weighted sum of MPA sizes as survival probabilities.

For comparison consider that each ecosystem contains all four species. This means in practice that we have three equal, independent ecosystems along a gradient. In this case the FOCs are:

$$\frac{\partial V}{\partial M_e} = 0 \Leftrightarrow d_e + |S| \prod_{f \neq e} \left(1 - M_f\right) = c_e M_e \quad \forall e \in E. \tag{4.11}$$

Because of symmetry the form of the solutions would be fully symmetric, as opposed to the gradient case as shown above. Moreover, if there was no overlap between the ecosystems, the FOC of each ecosystem would have been:

$$\frac{\partial V}{\partial M_e} = 0 \Leftrightarrow d_e + |s_e| = c_e M_e \quad \forall e \in E, \tag{4.12}$$

with $|s_e|$ the number of species occurring in that ecosystem. In that case there is no interdependence of the solutions and hence no strategic interaction in location choice. Consequently MPA size is determined solely by the biodiversity and direct benefits on the spot.

4.2.4. Strategic non-cooperation among countries

To describe the strategic non-cooperative or Nash equilibrium we first have to specify how the set of ecosystems E is distributed between countries $k \in \{1,2\}$. We divide the set of ecosystems into two subsets E_1 and E_2, one for each country.

Each country takes the decision of the other country as given when it maximizes its own net benefit function. Countries reap the direct benefits of their ecosystems and bear the costs of MPAs in their own ecosystems. Additionally the value of the biodiversity services accrues to both countries in equal shares. Consequently, each country maximizes:

$$V_k\left(M_e\right)=\sum_{e\in E_k} D_e\left(M_e\right)+\tfrac{1}{2}B\sum_{i\in S}\left(1-\prod_{e\in N_i}\left(1-f\left(M_e\right)\right)\right)-\sum_{e\in E_k} C_e\left(M_e\right),\ \forall k\in\{1,2\} \tag{4.13}$$

First order conditions for an interior solution are then:

$$\begin{aligned}\frac{\partial V_k}{\partial M_e}=\sum_{e\in E_k}\frac{\partial D_e\left(M_e\right)}{\partial M_e}+\tfrac{1}{2}\frac{\partial B\sum_{i\in S}\left(1-\prod_{e\in N_i}\left(1-f\left(M_e\right)\right)\right)}{\partial M_e}\\ -\sum_{e\in E_{1,2}}\frac{\partial C\left(M_e\right)}{\partial M_e}=0,\quad \forall k\in\{1,2\},\ \forall e\in E_k.\end{aligned} \tag{4.14}$$

These first order conditions also show the standard form. Private marginal benefits consist of the marginal benefits from direct services owned by the country and half of the total biodiversity services, private marginal costs are the marginal costs of MPAs that the country has to assign to get that additional benefits.

If we take the example of the previous section and assign ecosystems 1 and 2 to country 1 and ecosystem 3 to country 2, we get the following welfare functions:

$$\begin{aligned}V_1\left(M_e\right)&=\sum_{e\in E_1} d_e M_e+\tfrac{1}{2}\sum_{S}\left(1-\prod_{e\in N_i}\left(1-M_e\right)\right)-\sum_{e\in E_1}\tfrac{1}{2}c_e M_e^2\\ V_2\left(M_e\right)&=d_3M_3+\tfrac{1}{2}\sum_{S}\left(1-\prod_{e\in N_i}\left(1-M_e\right)\right)-\tfrac{1}{2}c_3M_3^2.\end{aligned} \tag{4.15}$$

These two welfare functions result in the following first order conditions for an interior solution:

$$\begin{aligned}\frac{\partial V_1}{\partial M_1}&=d_1+\tfrac{1}{2}\left(2-M_2\right)-c_1M_1=0\\ \frac{\partial V_1}{\partial M_2}&=d_2+\tfrac{1}{2}\left(2-M_1-M_3\right)-c_2M_2=0\\ \frac{\partial V_2}{\partial M_3}&=d_3+\tfrac{1}{2}\left(2-M_2\right)-c_3M_3=0.\end{aligned} \tag{4.16}$$

It is informative before solving these three equations to look at the reaction curves for the three MPAs. These curves are described by the following equations:

$$\begin{aligned}
M_1 &= \frac{2(d_1+1)c_2 - d_2 - 1 + \tfrac{1}{2}M_3}{2c_1c_2 - \tfrac{1}{2}} \\
M_2 &= \frac{2(d_2+1)c_1 - d_1 - 1 - c_1M_3}{2c_1c_2 - \tfrac{1}{2}} \\
M_3 &= \frac{d_3 + 1 - \tfrac{1}{2}M_2}{c_3}.
\end{aligned} \tag{4.17}$$

The sizes defined by (4.16) and (4.17) are affected in two ways that will induce smaller MPA choices: free-riding and location leakage. Free-riding occurs because countries do not account for generated benefits in other countries by biodiversity. This can be seen from the ½ factor in the FOC in (4.16).

Location leakage can be seen from (4.17): if country 1 increases its MPA in ecosystem 2, country 2 will reduce its MPA size in ecosystem 3, and vice-versa. Another interesting effect of location leakage is that even though M_3 did not play a role in the original FOC (4.16) with respect to M_1, it has a positive influence on the equilibrium outcome of M_1. The intuition is that if country 2 invests in the species of ecosystem 3, location leakage applies in ecosystem 2. This in turn also decreases the protection of species shared between ecosystem 1 and 2 and therefore the MPA in ecosystem 1 is increased. The effect also runs the other way, even though that cannot be seen directly from the reaction curves.

Furthermore, we can see from the equations in (4.17) that the size of M_1 and M_2 is influenced positively by the direct benefits from the ecosystem itself and the cost in the other ecosystem, and negatively by the benefits of the other ecosystem.

Solving the equations in (4.17) simultaneously we get:

$$\begin{aligned}
M_1 &= \frac{d_1(4c_2c_3 - 1) + 2c_3(2c_2 - d_2) + d_3 - 2c_3}{4c_1c_2c_3 - c_1 - c_3} \\
M_2 &= \frac{4c_1c_3(d_2+1) - 2d_1c_3 - 2c_1 - 2d_3c_1 - 2c_3}{4c_1c_2c_3 - c_1 - c_3} \\
M_3 &= \frac{d_3(4c_1c_2 - 1) + 2c_1(2c_2 - d_2) + d_1 - 2c_1}{4c_1c_2c_3 - c_1 - c_3}
\end{aligned} \tag{4.18}$$

which have a similar structure as in the social optimum (cf. (4.9)) but differ in several products with factors of two and four. If benefits would only consist of the public good, and ecosystems and species numbers were evenly distributed among countries, the strategic non-cooperative equilibrium would exactly be half of the full cooperation case. However, private benefits are also involved and are fully accounted for. Furthermore the distribution of ecosystems and species is not fully symmetric. Therefore strategic non-cooperative equilibrium payoffs and MPA sizes are not necessarily a factor two smaller than full cooperation payoffs and MPA sizes.

From the equations in (4.18) we can also see the positive influence of the direct benefits in ecosystem 1 on the MPA size in both ecosystem 1 and 3 and vice versa: an increase in d_1 will increase M_1 if $(4c_2c_3-1)>0$ and the denominator is positive. Similarly if the denominator is positive d_1 will increase M_3. This effect also holds in reverse.

To show that MPAs are generally smaller than under full cooperation we further simplify the above sizes in equation (4.9) and (4.18) by assuming that no direct benefits exist (i.e. d_1=d_2=d_3=0) and symmetric costs for all ecosystems (i.e. c_1=c_2=c_3=c). This results in:

$$\begin{aligned} M_1^{FC}-M_1^{N} &= M_3^{FC}-M_3^{N}=\frac{c\left(2c^2-3c+2\right)}{\left(c^2\text{ - }2\right)\left(2c^2-1\right)\right)} \\ M_2^{FC}-M_2^{N} &= \frac{2c\left(c^2-3c+1\right)}{\left(c^2\text{ - }2\right)\left(2c^2-1\right)\right)}. \end{aligned} \tag{4.19}$$

Where M_e^{FC} is MPA size under full cooperation in ecosystem e and M_e^{N} is MPA size under non-cooperative strategic equilibrium in ecosystem e. An interior solution for all M_e requires that $c>2$. In that case full cooperation MPAs in ecosystem 1 and 3 are always larger than those under the strategic equilibrium. The MPA in ecosystem 2 under full cooperation is smaller for $2<c<\frac{\sqrt{5+3}}{2}\approx 2.6,$ and larger for $c>2.6.$

The associated equilibrium survival probabilities of species are again as in equation (4.10), but now the associated M_e are those defined in equation (4.18). Because MPAs sizes are smaller under strategic non-cooperation, the survival probabilities are smaller as well.

For comparison, consider the case with three independent but equal ecosystems as before. The FOCs of the countries are then:

$$\frac{\partial V}{\partial M_e}=0\Leftrightarrow d_e+\frac{1}{2}|S|\prod_{f\neq e}\left(1-M_f\right)=c_eM_e \quad \forall e\in E. \tag{4.20}$$

The solution would still be fully symmetric, as under full cooperation.

Similarly, if there would have been no overlap the FOC would have been:

$$\frac{\partial V}{\partial M_e} = 0 \Leftrightarrow d_e + \frac{1}{2}|S_e| = c_e M_e \quad \forall e \in E. \tag{4.21}$$

Qualitatively both the cases of no-overlap and three equal ecosystems give the same result: free-riding affects only size, but there is no location leakage. With three equal ecosystems as in (4.20) all species are affected in the same way, and hence there is no location choice. In the no-overlap case each species occurs in exactly one ecosystem and consequently it is impossible to exploit each other through location leakage. In both cases however, countries only account for the benefits of protection in their own country and consequently choose smaller MPAs than they would have done under full cooperation, so free-riding is not eliminated.

4.2.5. Conservation autarky

In the previous section we have implicitly assumed that from an individual country's perspective it does not matter for the indirect services where a species is protected. This gives rise to location leakage, causing countries to free-ride on the protection of species by others. In a situation of conservation autarky it is assumed that protection of species in one country is not a substitute for protection in another country (as in Kark et al., 2009 and Jantke and Schneider, 2010).

Conservation autarky eliminates a part of the public goods issue because location leakage no longer occurs. If the global society, however, considers protection of the same species in another country as a substitute, the most effective protection plan would not be implemented, because countries ignore protection in other countries. Moreover countries still account for biodiversity protected in their own country only, and therefore they still free-ride.

Under conservation autarky each country maximizes its perceived benefits:

$$V_k(M_e) = \sum_{e \in E_k} D_e(M_e) + \tfrac{1}{2} B \sum_{i \in S_k} \left(1 - \prod_{e \in N_{ik}} \left(1 - f(M_e)\right) \right) - \sum_{e \in E_k} C(M_e), \quad \forall k \in \{1,2\} \tag{4.22}$$

where S_k denotes the set of species in country k, and N_{ik} denotes the set of ecosystems where species i occurs in country k. The factor ½ applies because we are only considering a fraction of all countries. The associated first order conditions for a maximum are:

$$\frac{\partial V_k}{\partial M_e} =$$

$$\sum_{e \in E_k} \frac{\partial D_e\left(M_e\right)}{\partial M_e} + \tfrac{1}{2} \frac{\partial B \sum_{i \in S_k}\left(1 - \prod_{e \in N_{ik}} 1 - f\left(M_e\right)\right)}{\partial M_e} - \sum_{e \in E_k} \frac{\partial C_e\left(M_e\right)}{\partial M_e} = 0, \tag{4.23}$$

$$\forall k \in \{1,2\}, \forall e \in E.$$

These first order conditions differ from the conditions derived for strategic non-cooperation in the second term. Marginal benefits of species protection are restricted to species protected domestically.

Turning to our earlier example the private welfare functions of the countries are:

$$\begin{aligned} V_1\left(M_e\right) &= \sum_{e \in E_1} d_e M_e + \tfrac{1}{2} \sum_{i \in S_1}\left(1 - \prod_{e \in N_{i1}}\left(1 - M_e\right)\right) - \sum_{e \in E_1} \tfrac{1}{2} c_e M_e^2 \\ V_2\left(M_e\right) &= d_3 M_3 + \sum_{i \in S_2}\left(1 - \prod_{e \in N_{i2}}\left(1 - M_e\right)\right) - \tfrac{1}{2} c_3 M_3^2 \end{aligned} \tag{4.24}$$

with resulting first order conditions:

$$\begin{aligned} \frac{\partial V_1\left(M_e\right)}{\partial M_1} &= d_1 + \tfrac{1}{2}\left(2 - M_2\right) - c_1 M_1 = 0 \\ \frac{\partial V_1\left(M_e\right)}{\partial M_2} &= d_2 + \tfrac{1}{2}\left(2 - M_1\right) - c_2 M_2 = 0 \\ \frac{\partial V_2\left(M_e\right)}{\partial M_3} &= d_3 + 1 - c_3 M_3 = 0. \end{aligned} \tag{4.25}$$

As can be seen from (4.25) the substitution effect between countries and, hence, location leakage is gone. There is no longer a negative effect of M_2 on M_3 and vice versa. The substitution effect within one country remains. Solving the system in (4.25) gives:

$$\begin{aligned} M_1 &= \frac{4c_2\left(1 + d_1\right) - 2\left(1 + d_2\right)}{4c_1 c_2 - 1} \\ M_2 &= \frac{4c_1\left(1 + d_2\right) - 2\left(1 + d_1\right)}{4c_1 c_2 - 1} \\ M_3 &= \frac{1 + d_3}{c_3} \end{aligned} \tag{4.26}$$

where the substitution effect within country 1 clearly remains, as the direct benefits of ecosystem 2 (d_2) affect the size of MPA in ecosystem 1 (M_1) negatively and vice versa. However, the effect between countries has disappeared: the negative effect of d_3 on M_2 and the positive effect on M_1, and vice versa are both gone.

Using the same simplification as in section 4.2.4. (d_1=d_2=d_3=0 and c_1=c_2=c_3=c) to compare full cooperation with conservation autarky, we find:

$$\begin{aligned} M_1^{FC} - M_1^{CA} &= \frac{2\left(c^2 - c + 1\right)}{(2c+1)\left(c^2 - 2\right)} \\ M_2^{FC} - M_2^{CA} &= \frac{2c\left(c - 3\right)}{(2c+1)\left(c^2 - 2\right)} \\ M_3^{FC} - M_3^{CA} &= \frac{c^2 - 2c + 2}{c\left(c^2 - 2\right)}, \end{aligned} \tag{4.27}$$

where M_e^{CA} denotes the MPA size in ecosystem e under conservation autarky. Again, $c > 2$ is required for an interior solution. These differences are generally positive, implying that full cooperation assigns larger MPAs than is done under conservation autarky, except in ecosystem 2 for $2 \le c \le 3$. Here free-riding is outweighed by the fact that a country considers certain species as unique (whereas they are not) and therefore overprotects these species. Survival probabilities are again as in (4.10) with the relevant M_e given by (4.26). Because M_e is generally larger under full cooperation than under conservation autarky, the survival probabilities are generally larger under full cooperation as well.

Compared to the free-riding solutions, we find:

$$\begin{aligned} M_1^{N} - M_1^{CA} &= \frac{1}{(2c+1)\left(2c^2 - 1\right)} \\ M_2^{N} - M_2^{CA} &= \frac{-2c}{(2c+1)\left(2c^2 - 1\right)} \\ M_3^{N} - M_3^{CA} &= \frac{1 - c}{c\left(2c^2 - 1\right)}. \end{aligned} \tag{4.28}$$

Here the area in the first ecosystem is larger under strategic non-cooperation, but the others are larger under conservation autarky. In general, because countries only consider protection in their own ecosystems, species that are transboundary receive more protection than under strategic non-cooperation but not as much as under full cooperation. Hence from the global perspective inefficiencies still occur, even though the situation is an improvement from strategic non-cooperation. Because M_1 is larger under strategic non-cooperation than under conservation autarky and M_2 and M_3 are smaller, the survival probability of species 1 is larger under strategic non-cooperation and that of species 3 is smaller. The effect on species 2 and 3 is ambiguous.

For comparison consider again the three equal independent ecosystems case. Then the following FOC apply:

$$\frac{\partial V}{\partial M_e} = 0 \Leftrightarrow d_e + \frac{1}{2}|S| \prod_{f \neq e \in E_{1,2}} \left(1 - M_f\right) = c_e M_e \quad \forall e \in E_1 \wedge \forall e \in E_2. \tag{4.29}$$

Generally, since countries ignore contributions by others and have all species in their domain, they will assign larger MPAs than under the strategic non-cooperative equilibrium. These MPAs are not as large as under full cooperation because countries only account for the services in their own country.

Similarly in the no-overlap case the FOCs are:

$$\frac{\partial V}{\partial M_e} = 0 \Leftrightarrow d_e + \frac{1}{2}|S_e| = c_e M_e. \tag{4.30}$$

Consequently in the no-overlap case conservation autarky is equal to strategic non-cooperation.

4.3. Simulation model

4.3.1. The ecosystem model

We will now explore the effects of a more realistic probability function, as well as larger numbers of ecosystems and species. For the simulation we will consider a coastline of ecosystems with a universal decay rate and an equal maximum number of species in each ecosystem. A non-universal decay rate would result in more abrupt changes in species composition between ecosystems. An example would be that ecosystems that are not neighbors have no species in common.[4]

From a matrix of distances between ecosystems and the maximum number of species, a distribution of species over ecosystems can be calculated. We will consider this distribution exogenous; its calculation is explained in Appendix 4.A.

[4]A more detailed specification in two dimensions and non-universal decay rate would be a more realistic setting, but is more difficult to solve and adds little to the results found here except from a richer set of possible solutions and configurations.

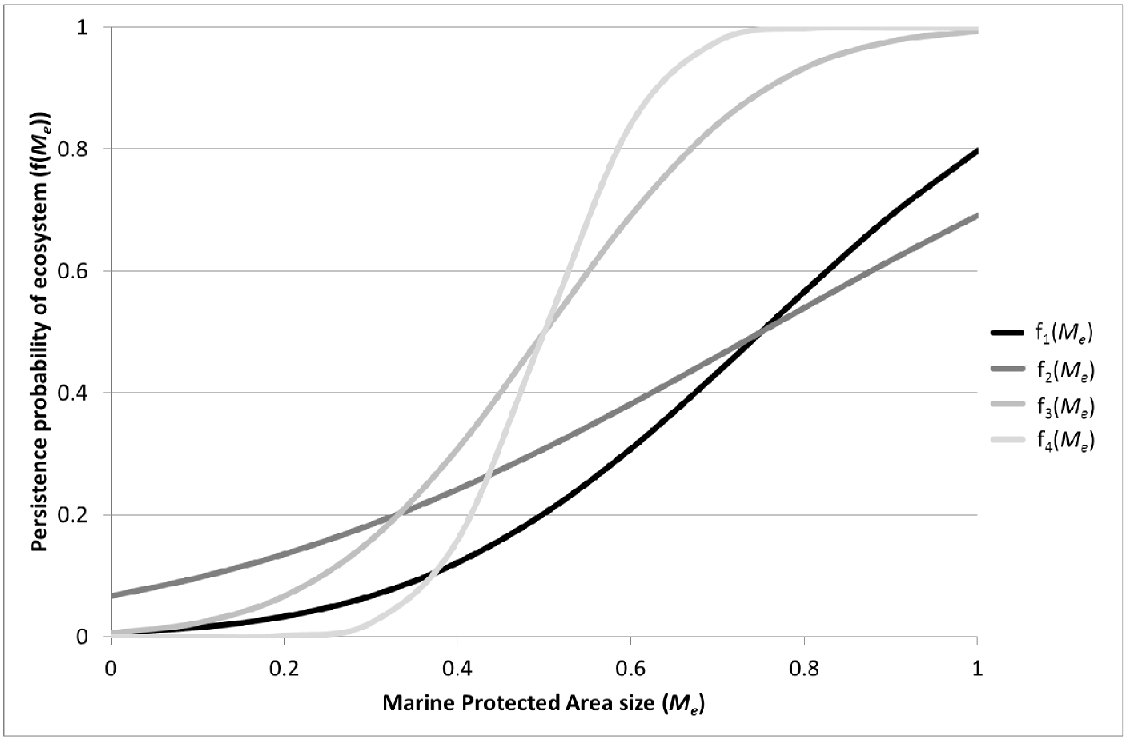


Figure 4.2: Examples of possible persistence probability functions of the ecosystem ($f(M_e)$) as a function of MPA size. The used parameters are for $f_1(M_e)$: mean (μ) = 0.75, standard deviation (σ) = 0.3, for $f_2(M_e)$: μ = 0.75, σ = 0.5, for $f_3(M_e)$: μ= 0.5, σ= 0.2 and for $f_4(M_e)$: μ = 0.5, σ = 0.1.

The persistence probability function is assumed to be a cumulative normal distribution with a mean (μ) between zero and one and a small standard deviation (σ) to keep the relation between the minimum and maximum survival probabilities and MPA size between zero and one. Furthermore it is assumed that the persistence probability function is equal across ecosystems. Examples of persistence probability functions for several parameter combinations are shown in Figure 4.2. The fact that such functions do not always cross the origin or are equal to one even if an ecosystem is fully protected reflect that ecosystems may have some persistence probability even without protection, and that ecosystems may not be guaranteed to persist even if they are fully protected.

4.3.2. The economic model

For the economic side of the model we have to specify a benefit function and a cost function. The benefit function consists of direct benefits from ecosystem services of the separate ecosystems and indirect benefits from biodiversity. The direct benefits under full cooperation are specified as:

$$D_e(M_e) = \sum_{e \in E} d_e M_e \tag{4.31}$$

The equivalent formulation under the strategic non-cooperation and conservation autarky is:

$$D_e(M_e) = \sum_{e \in E_k} d_e M_e, \quad \forall k \in \{1,2\}. \tag{4.32}$$

Similarly indirect benefits under full cooperation are specified as the benefits of biodiversity *b*, multiplied with the total expected biodiversity, measured as expected number of species:

$$B(M_e) = b \sum_{i \in S} \left(1 - \prod_{e \in N_i} \left(1 - \left(\frac{1}{2} \left(1 + \operatorname{erf}\left(\frac{M_e - \mu}{\sqrt{2\sigma^2}} \right) \right) \right) \right) \right) \tag{4.33}$$

where erf is the error function used for calculations of the cumulative normal distribution used for the persistence probability, μ is the mean of the distribution and σ is its standard deviation. Its equivalent under the strategic non-cooperative specification is:

$$B(M_e) = \frac{1}{2} b \sum_{i \in S} \left(1 - \prod_{e \in N_i} \left(1 - \left(\frac{1}{2} \left(1 + \operatorname{erf}\left(\frac{M_e - \mu}{\sqrt{2\sigma^2}} \right) \right) \right) \right) \right). \tag{4.34}$$

In (4.34) some of the M_e are exogenous to the decision maker as they are controlled by the other country. For conservation autarky the specification is:

$$B(M_e) = \frac{1}{2} b \sum_{i \in S_k} \left(1 - \prod_{e \in N_{ik}} \left(1 - \left(\frac{1}{2} \left(1 + \operatorname{erf}\left(\frac{M_e - \mu}{\sqrt{2\sigma^2}} \right) \right) \right) \right) \right) \quad k \in \{1,2\}. \tag{4.35}$$

In the simulations we will assume a cost function that is quadratic in MPA size:

$$C_e(M_e) = \tfrac{1}{2} \sum_{e \in E} c_e M_e^2. \tag{4.36}$$

4.3.3. Simulations

We will simulate a coastline with ten ecosystems and two countries; each country has the jurisdiction over five ecosystems. Parameter values for the simulations are given in Table 4.2. With these parameters we calculate a species distribution over the ten ecosystems that matches the patterns in

Table 4.2: Arbitrary parameter and set values of the simulation in the base case

Parameters	Values
S_e^{MAX}	50
X_o	1
r	0.9
δ	1 (per ecosystem)
μ	0.75
σ	0.3
d_e	2 (k€/ share protected)
b	0.2 (k€/expected species)
c_e	10 (k€/share protected)

Sets	Range on elements
E	(1-10)
E_1	(1-5)
E_2	(6-10)
S	(1-255)
S_1	(1-149)
S_2	(33), (71), (78), (86), (97), (100-104), (118-255)

Table 4.3: Initial distribution of species over ecosystems

Ecosystem	Species
1	(1-50)
2	(1-20), (51-80)
3	(21-28), (51-70), (81-102)
4	(29-31), (51-54), (71-74), (81-96), (103-125),
5	(32), (75-77), (81-85), (97-99), (103-117), (126-148)
6	(33), (78), (97), (100-101), (103), (118-124), (126-143), (149-167), (91-110)
7	(71), (86), (104), (126-131), (144), (149-162), (168-193)
8	(102), (125), (145-147), (149-154), (163-164), (168-181), (194-216)
9	(125), (145), (163), (165-166), (168), (182-188), (194-209), (217-237)
10	(148), (167), (189-191), (194), (210-235), (238-255)

Table 4.4: Overlap in number of species between ecosystems

	1	2	3	4	5	6	7	8	9	10
1	50									
2	20	50								
3	8	20	50							
4	3	8	20	50						
5	1	3	8	20	50					
6	1	1	3	8	20	50				
7	1	1	1	3	8	20	50			
8	0	1	1	1	3	8	20	50		
9	0	0	1	1	1	3	8	20	50	
10	0	0	0	1	1	1	3	8	20	50

The matrix is symmetric and therefore only the lower halve is shown

exponential decay as described in Appendix 4.A. The resulting pattern is shown in Tables 4.3 and 4.4.

Next we solve the economic model given this species distribution and other parameters both for full cooperation, strategic non-cooperation and conservation autarky. To overcome the non-convexities of this problem we used a hybrid evolutionary algorithm to calculate the global maximum for full cooperation and conservation autarky and a hybrid coevolutionary algorithm for the strategic non-cooperative equilibrium as described in Son and Baldick (2004). We used an adapted version of the continuous genetic algorithm described by Haupt and Haupt (2004).

The results for full cooperation, strategic non-cooperation and conservation autarky are shown in Figure 4.3 and Table 4.5. In fact two non-cooperative equilibria exist: in the first equilibrium country 1 is slightly better off, in the second equilibrium country 2 is slightly better off (cf. Table 4.5). The two different equilibria differ in chosen main MPAs. The ecosystems with the highest number of unique species (ecosystems 1 and 10) get the highest priority in protection. However, under strategic non-cooperation a smaller area is protected than under full cooperation. The differences between the two are not too large because of the S-shape of the persistence probability.

There are two main reasons for assigning smaller MPAs under the strategic equilibria: free-riding and location leakage. Free-riding can be generally observed in Figure 4.3, i.e. all MPAs are smaller than under full cooperation because countries do not account for the benefits generated in the other country.

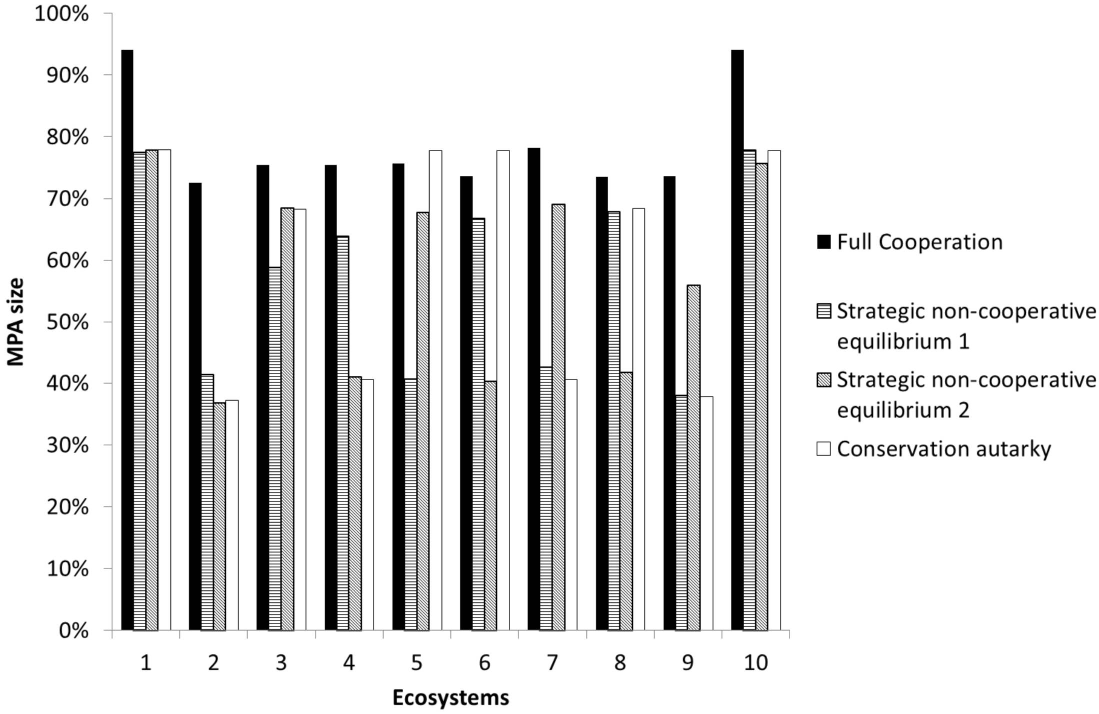


Figure 4.3: MPA sizes in ecosystems under full cooperation, strategic non-cooperative equilibrium (twice) and conservation autarky

Table 4.5: Net benefits to the global society and to separate countries under full cooperation, strategic non-cooperative equilibria and conservation autarky (perceived and actual)

	Full cooperation	Strategic non-cooperative equilibrium 1	Strategic non-cooperative equilibrium 2	Conservation autarky from a global perspective
Total net benefits	24.58	20.32	20.39	21.45
Net benefits country 1	12.29	10.42	9.94	-
Net benefits country 2	12.29	9.90	10.45	-
Expected number of species	200.38	132.36	132.97	145.97

Location leakage is also visible in Figure 4.3, but it mainly occurs near the border. Moreover it runs in different directions depending on the non-cooperative equilibrium. In equilibrium 1 country 1 is the country that exploits the leakage, in equilibrium 2 country 2 exploits the leakage, resulting in a much smaller MPA in the exploiting country and a larger MPA in the exploited country.

In conservation autarky free-riding still exists, but location leakage is absent. In that case ecosystems at the borders are stronger protected. Because countries do not account for protection of species in other countries, some species are considered to be unique, whereas they actually also occur in other countries. From the global perspective some ecosystems are protected stronger than necessary, because protection in one country can be substituted by protection in another country.

From Figure 4.3 we can clearly see these results: most MPA sizes under conservation autarky are comparable to those under strategic equilibria, except for border cases. The similarity to the strategic equilibria is caused by the remaining free-riding problem. Ecosystems 5 and 6 in contrast are heavily protected, even more than under full cooperation. This occurs because species in these ecosystems are considered to be unique by the separate countries. This eliminates location leakage, and because substitution of protection across

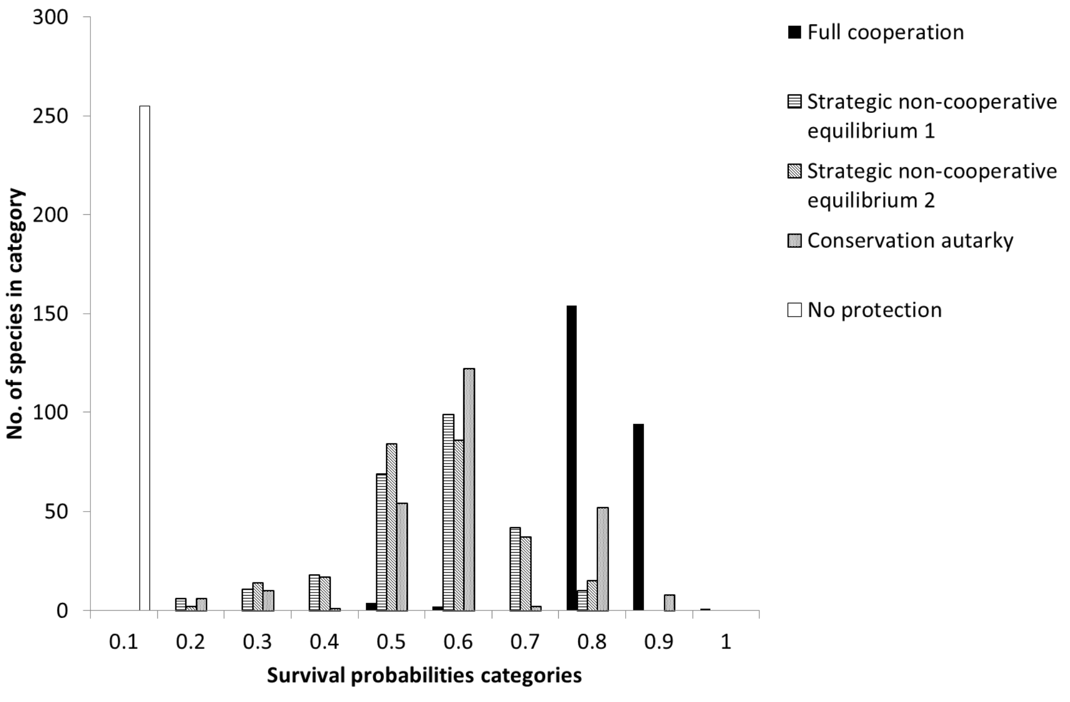


Figure 4.4: Histogram of survival probabilities of species under full cooperation, strategic non-cooperative equilibrium 1 & 2, conservation autarky and when no protection is applied (i.e. no MPAs are assigned). The interval size is 0.1.

countries is completely ignored, introduces an inefficiency from a global perspective.

Table 4.5 clearly illustrates the occurring inefficiencies with the associated equilibria. Total net benefits to the global society are highest under full cooperation, and lower under both the strategic equilibria and conservation autarky. The inefficiency under conservation autarky is smaller, but this depends partly on parameter values. In general from these two inefficiencies conservation autarky is probably preferred over the strategic non-cooperative equilibrium.

These distinct differences are also found in the survival probabilities of individual species. In Figure 4.4 we show histograms of survival probabilities of species under full cooperation, strategic non-cooperative equilibria, conservation autarky and when no protection is carried out.

Figure 4.4 shows that under full cooperation, the survival probability of most species is between 0.7 and 0.9. In both strategic equilibria in contrast the survival probability of the majority of species is between 0.4 and 0.7. Compared with full cooperation species receive less protection, than would be optimal from the global perspective. Under conservation autarky most species have survival probabilities between 0.5 and 0.9. This is an improvement compared to both strategic equilibria, but still not as good as full cooperation.

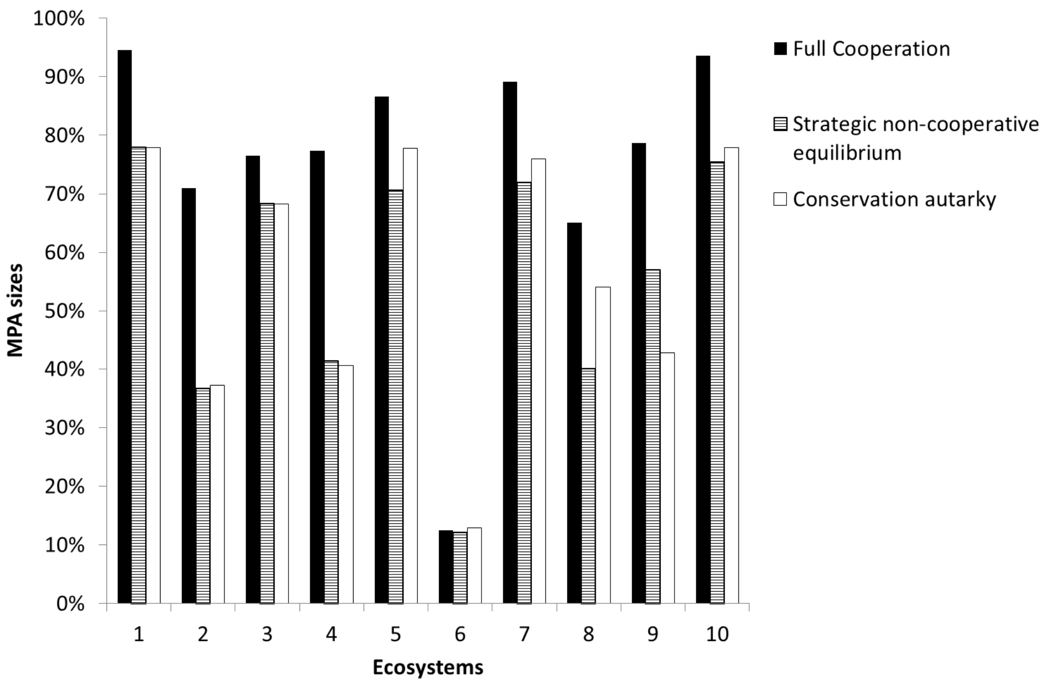


Figure 4.5: MPA sizes in ecosystems under full cooperation, strategic non-cooperative equilibrium and conservation autarky when the costs in ecosystem 6 are increased by 100%.

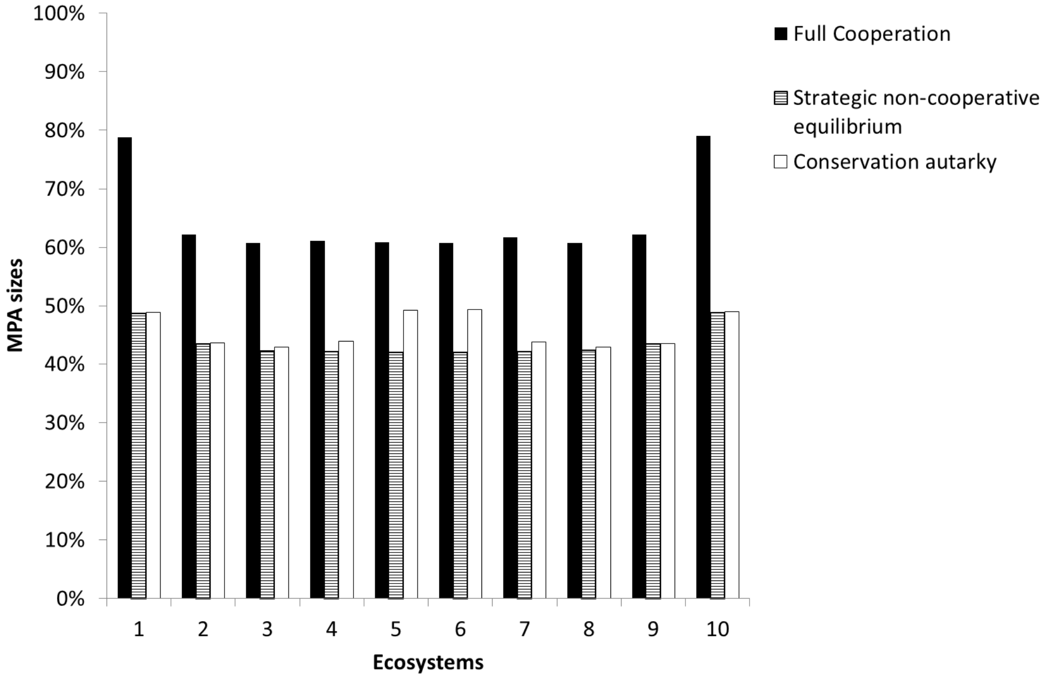


Figure 4.6: MPA size under full cooperation, strategic non-cooperative equilibrium and conservation autarky when μ=0.75 and σ=0.5

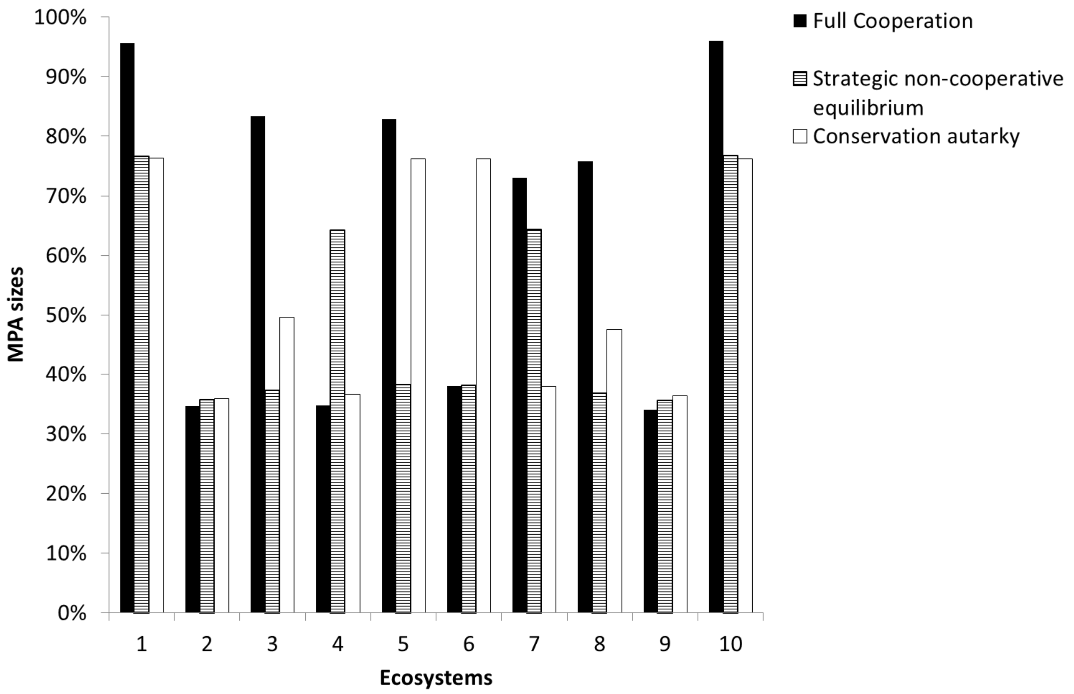


Figure 4.7: MPA size under full cooperation, strategic non-cooperative equilibrium and conservation autarky when r = 0.65 and number of species=200

If no MPAs are assigned, as in "no protection" in Figure 4.4, all species have a very low survival probability, all below 0.02.

4.3.4. Sensitivity analysis

The parameters describing locations in terms of costs and benefits have opposite effects. Raising direct benefits of MPAs in a certain ecosystem increases the MPA size in that ecosystem and decreases the MPA in adjoining ecosystems, for full cooperation, strategic non-cooperation and conservation autarky. In the case of conservation autarky however, the effect is limited to a country, as countries only consider their own species and ecosystems.

Raising the costs in one location has the opposite effect. This can be seen from Figure 4.5 where we have increased the costs of MPAs in ecosystem 6 with 100%. Compared with Figure 4.3 the MPA is lower in ecosystem 6 and MPAs in neighboring ecosystems are increased, except under conservation autarky where the effect stops at the border, and MPAs in country 1 are of the same size as in Figure 4.3. The effect of an increased price of conservation is most clear in the neighbor ecosystems and decreases with distance. Ecosystem 1 and 10 for example are hardly affected (cf. Figure 4.3).

Incidentally, the introduction of the sharp asymmetry in costs also removes one of the non-cooperative equilibria. Country 2 now always assigns a small MPA in ecosystem 6 because of the high costs. This decision in turn reduces the possibilities for location leakage.

It is also clear from Figure 4.5 that on the one hand conservation autarky overvalues species from the global perspective: under this scenario ecosystem 6 receives the highest level of protection compared to the other two scenarios. Strategic non-cooperation on the other hand undervalues species; it assigns the smallest MPAs and the compensation in other ecosystems is also the smallest.

The ecological parameters describing the persistence probability mainly affect the conservation pattern in the ecosystems that contain common species. A low σ, indicating a very steep probability curve (cf. Figure 4.2), induces a conservation pattern where conservation is concentrated in a few ecosystems with large MPAs and small MPAs in the neighboring ecosystems. A higher σ induces a more even spread pattern with overall smaller MPAs. The MPAs in ecosystem 1 and 10 are always higher than MPAs in other ecosystems and in the conservation autarky case ecosystem 5 and 6 also have high MPAs. This is clearly shown in Figure 4.6, where we show full cooperation, strategic non-cooperation and conservation autarky for $\mu = 0.75$ and $\sigma = 0.5$. Increasing or decreasing μ does not alter the pattern of MPA designation but mainly the level, increasing μ raises MPA levels and decreasing μ decreases them.

The decay rate r determines the overlap between ecosystems and hence the number of unique species in each ecosystem. An increase in r induces a smaller overlap between ecosystems and hence more unique species. Incidentally, it also increases the total number of species needed to generate a species distribution over ecosystems, because each ecosystem now requires a higher number of unique species.

A decrease in r lowers the MPA size for two reasons: fewer species exist in the first place and fewer unique species exist. Figure 4.7 shows the MPA sizes under the three scenarios with $r = 0.65$ and 200 species. The ecosystems with the (perceived) unique species are still almost fully protected, but other ecosystems have much smaller MPAs under these parameter values. Some ecosystems have become increasingly valuable, because the complete species distribution has changed and more unique species occur elsewhere, which results in a different conservation pattern as well.

4.5. Discussion & conclusions

In this chapter we have presented a modeling framework for the allocation of MPAs. We have investigated the full cooperation case and compared it with two cases of non-cooperation: strategic non-cooperation and conservation autarky. Strategic non-cooperation and conservation autarky both differ from full cooperation. In both cases ecosystems and species are under-protected due to free-riding. However, while under the strategic non-cooperation all ecosystems are generally under-protected, under conservation autarky species at the border receive a higher level of protection. The fundamental difference between these two scenarios is whether or not countries consider the same species protected in another country as a substitute to protection of that species in their own country.

Conservation autarky is inefficient, because biodiversity conservation in one country can be a substitute for conservation in another country. However, it is less inefficient than the free-riding case. The case of having no knowledge of species existing elsewhere may actually be a bliss in a global perspective, because location leakage is eliminated and species are no longer under-protected at the borders. On the contrary, species at the border are now over-protected from a global perspective. Free-riding, however, remains.

An important result from our simulations is that even when a country chooses to free-ride on the contribution of the other, most unique species are still decently protected, with the exception of those species occurring only in ecosystems next to heavily protected ecosystems. These species that become less protected, because neighboring ecosystems are heavily protected. They suffer from "local location leakage". It can be observed from Figures 4.3 and 4.5-4.7 that the largest differences in MPA sizes under strategic non-cooperation compared to full cooperation are found in ecosystems where common transboundary species dwell. The exceptions where strategic non-cooperation chooses a larger MPA than full cooperation can be explained by differences in conservation patterns.

This finding may seem counterintuitive at first, but makes sense from a valuation point of view: unique species are the most valuable and therefore most heavily protected. Location leakage cannot occur for these species precisely because they are unique. They cannot be protected elsewhere and consequently these species are better protected than more common transboundary species. Common species in contrast can be protected elsewhere and therefore countries choose to free-ride on the protection of these species and ecosystems.

A well-known question in ecology related to the substitution effect is whether we should select a Single Large Or Several Small reserves (SLOSS problem). In our model the answer to this question is shown to be very much dependent on the distribution of species over ecosystems and the persistence probability of ecosystems. Assuming that the persistence probability curve is the same in all ecosystems the following holds. If the persistence probability curve of ecosystems is very steep and species are wide spread a single large reserve is better, because a large reserve is needed to reach a decent level of protection and a lot of species can be protected in that single area. If many rare species exist and the curve is very flat, several small reserves are better, because only small reserves are needed for a decent level of protection and each added reserve adds extra protected species. This whole assertion, however hinges on the persistence probability curve being similar in all ecosystems, which is not necessarily true. A general answer to the SLOSS problem cannot be given.

In this chapter we have studied international cooperation on MPA allocation at the ecosystem level and assumed that all services and the distribution of the effects of the MPA were accounted for. As shown in Punt et al. (2010) accounting fully for all services is an important condition for the optimal allocation of MPAs because, if this is not the case, it may be better not to cooperate.

We have shown how the distribution of species over ecosystems affects the assignment of MPAs in neighboring ecosystems through location leakage. Location leakage induces preferences to spread MPAs. This is similar to the analysis for terrestrial conservation in e.g. Albers et al. (2008).

Although we have shown how the location of MPAs per ecosystem matters, the distribution of MPAs within an ecosystem is also very important, especially if an ecosystem crosses the border between countries. In that case the movement of species determines who bears the cost and reaps the benefits (Ruijs and Janmaat, 2007). We do not consider such movements in this chapter, but hypothesize that including movement would increase free-riding and location leakage.

In our simulations most of the parameters are symmetric with exception of the distribution of species. This asymmetry is the reason that two strategic equilibria exist, in which the gains are distributed differently. Asymmetry in other aspects then distribution of species would alter the outcomes of our model but not to a large extent. We have shown some of the effects of asymmetry in the sensitivity analysis and we found that the exact patterns of MPA allocation changes but the general conclusions remain.

This chapter is the first to investigate the effect of substitution of protection. We conclude that substitution is an important, but ignored effect in conservation planning and that the emphasis may have been too much on conservation autarky instead of on the dangers of free-riding and location leakage. As we have shown, conservation autarky may be not such a bad situation, if it is compared with strategic non-cooperation.

In the light of our analysis international cooperation efforts on the protection of species should focus on three areas:

- Transboundary species. Species that are known to occur in ecosystems on both sides of the borders, are the ones that will most likely be under-protected through location leakage
- Unique species. Some unique species may suffer from "local location leakage", when neighboring ecosystems are well protected because of unique species there.
- Species that occur in ecosystems that have low direct benefits of protection (or high costs). These are likely to be under-protected, if countries do not cooperate. Free-riding on indirect benefits will have a relatively large impact if direct benefits are small.

4.A. Appendix: Calculation of the distribution of species over ecosystems

Given a set of species and ecosystems, a universal decay rate, a matrix of distances between ecosystems and the maximum number of species in each ecosystem a distribution of species over ecosystem, $Distribution_{i,e}$, can be calculated with a relatively simple mathematical model. Starting from a dummy objective:

$$DUM = \sum_{e \in E} \sum_{i \in S} Distribution_{i,e} \tag{4A.1}$$

where DUM is a variable used for the maximization, and $Distribution$ is a binary matrix denoting species i's presence (1) or absence (0) in ecosystem e, and is the actual variable of interest.

Each ecosystem e has a number of species, $|s_e|$, and this number is exogenously given. In the distribution we want all ecosystems to contain that number of species, therefore:

$$|s_e| = \sum_{i \in S} Distribution_{i,e} \quad \forall e \in E \tag{4A.2}$$

is a restriction on the distribution that has to hold.

Consider the number of species common to two ecosystems. The similarity (in number of species) between two ecosystems $e \in E$ and $f \in E$, $Sim_{e,f}$, is calculated with the distance decay function as follows:

$$Sim_{e,f} = \left(X_o e^{-r\delta}\right) \min\left(|s_e|, |s_f|\right) \quad \forall e, f \in E, e \neq f \tag{4A.3}$$

with X_0 the maximum similarity (usually one), r the decay rate, and δ the distance between ecosystem e and f. The minimization term adjusts the similarity for the number of species present in each area.

Given a distribution of species over ecosystems, *Distribution*, we can check whether this distribution matches the required similarity, $Sim_{e,f}$, by calculating the similarity implied by this distribution. This similarity $Overlap_{e,f}$ is calculated as follows:

$$Overlap_{e,f} = Distribution^T \times Distribution \tag{4A.4}$$

with $Distribution^T$ denoting the transpose of the distribution matrix. Thus the full model becomes:

$$\max DUM = \sum_{e \in E} \sum_{i \in S} Distribution$$
$$\text{s.t.} \quad |s_e| = \sum_{i \in S} Distribution \quad \forall e \in E \tag{4A.5}$$
$$Sim_{e,f} = Overlap_{e,f} \quad \forall e, f \in E.$$

Although the model sketched above is strictly speaking a Mixed Integer Non-Linear Problem, it can be approximated with a normal Non-Linear Problem (NLP) by letting *Distribution* be continuous over the interval [0,1]. Through rounding of $Sim_{e,f}$ to the nearest integer, and equalizing it with $Overlap_{e,f}$ we have only constraints consisting of integers, thus the solution of the NLP will coincide with the mixed integer variant. The solution to is usually not unique as many configurations satisfy the constraints and the maximum value of the objective variable is the same for all those configurations.

Chapter 5*: Marine Protected Areas in the High Seas and their impacts on international fishing agreements

5.1. Introduction

Fisheries management is in a crisis. According to the FAO statistics in the last two decades 20 to 30 % of all fish stocks were either over-exploited or depleted (FAO, 2009). According to economic theory one of the main reasons for over-exploitation is the lack of property rights, offering no incentives for fishing nations to conserve stocks. The lack of property rights was partly resolved in 1982 with the introduction of Exclusive Economic Zones, with the UN Convention on the Law of the Sea (UN, 1982). In these zones of 200 miles, fishing is the exclusive right of a country, although this may still cause problems for transboundary stocks, especially if the adjoining countries cannot agree upon a cooperative management scheme.

However, beyond the exclusive economic zones exist the High Seas, where no country can claim exclusive fishing rights. These High Seas comprise a considerable part of the world oceans as shown in Figure 5.1. Although no country can claim property rights to fish, this does not mean that there are no rules governing fishing in the High Seas. The UN Fish stock agreement on straddling fish stocks states that countries should "adopt measures to ensure long-term sustainability of straddling fish stocks" (UN, 1995). Moreover international law rules that countries wishing to participate in fishing the High Seas should join a regional fishing management organization (RFMO) and that within an RFMO a management scheme has to be set up (UN, 1995).

In practice, however, the management through RFMOs has proven to be difficult. Large problems exist of unreported and unregulated fishing. Because enforcement is difficult or even impossible and the chance of detection is small, countries have incentives to free-ride on the agreed quota, by underreporting the amounts caught (unreported fishing). Furthermore countries that do fish in parts of High Seas, without joining the appropriate RFMO are said to be involved in unregulated fishing (FAO, 2001). Thus free-riding makes agreeing on quota within RFMOs very difficult, and persuading fishing nations to join even harder. Another problem associated with the formation of RFMOs is the new member problem. If a new country enters the fishery, this may decrease the payoffs of the current members, even to such an extent that cooperation is no longer feasible.

*This Chapter is based on the article: Maarten J. Punt, Hans-Peter Weikard and Ekko C. van Ierland. Marine Protected Areas in the High Seas and their impacts on international fishing agreements. Submitted.

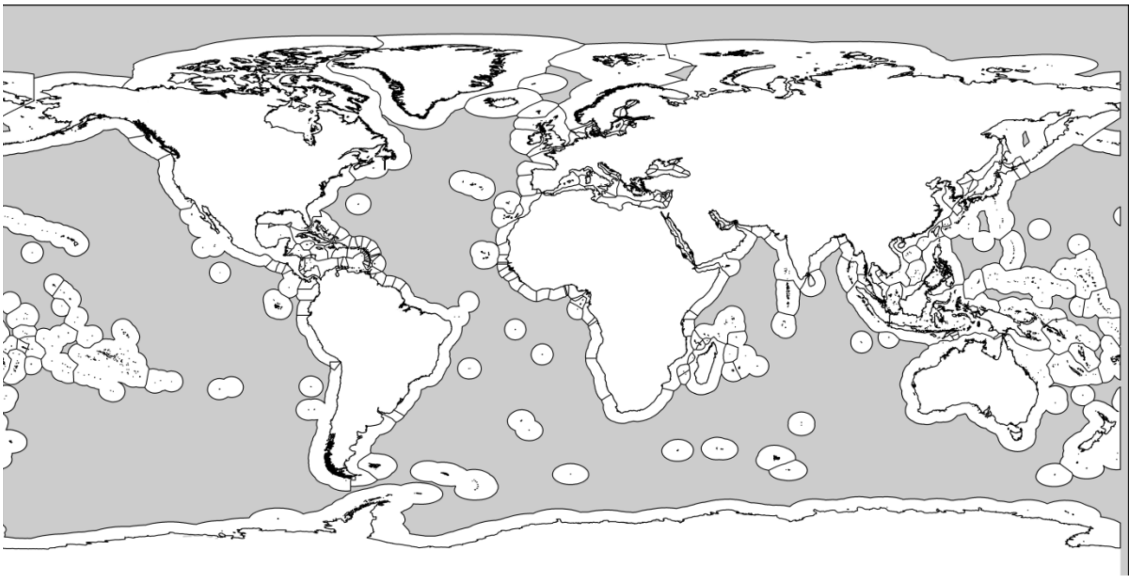


Figure 5.1: Countries of the world and their claimed EEZs (shown in white). The gray shaded areas comprise High Seas. Source: VLIZ 2011.

In part because of these failings some fisheries managers and scientists have now turned their attention to other management tools. One of these tools is the designation of Marine Protected Areas in the High Seas. Marine Protected Areas mean different things to different people and accordingly can have different goals, such as nature conservation or fisheries management (Punt et al., 2010). Several proposals and proponents for MPAs in the High Seas exist (Sumaila et al., 2007; Ardron et al., 2008; IUCN, 2010; WWF, 2010). Proposals for the size of MPAs range from 20%-30% of all seas (Sumaila et al., 2007). The current size of MPAs is 1.17% of all seas (Spalding et al., 2010), while the parties to the Convention on Biological Diversity have agreed in 2006 to conserve at least 10% of the seas. They have reaffirmed this decision in the 2010 convention (CBD, 2010).

In this chapter we will interpret MPAs as no-take zones for fisheries management. We will ignore other uses, although these are as important in considerations of MPAs if they are intended for multiple goals (Punt et al., 2010).

Several authors have looked into the strategic aspects of MPA designation for fisheries, but most have investigated the performance of MPAs in the presence of open access (e.g. Hannesson, 1998) or looked at MPAs within EEZs, i.e. shared or transboundary fish stocks (Sumaila, 2002; Ruijs and Janmaat, 2007). The possibilities for cooperation on management agreements on fish stocks has also been investigated, both with cooperative games (Kaitala and Lindroos, 1998; Li, 1998; Kronbak and Lindroos, 2007) and non-cooperative games (Pintassilgo, 2003; Kwon, 2006; Pham Do and Folmer, 2006; Pintassilgo and Lindroos, 2008; Pintassilgo et al., 2010). The possibilities and strategic aspects of MPAs in the High Seas, however, have not been examined so far.

In this chapter we will expand the modeling of RFMO fisheries agreements on highly migratory and straddling stocks with MPAs. In their RFMO model Pintassilgo et al. (2010) formulate a partition function based on the Gordon-Schaefer model and introduce asymmetric countries. Such a partition function approach is especially suited to analyze such problems. We will therefore use their model and adapt it, so that it facilitates the introduction of an MPA.

Our main contribution to the literature is to investigate the influence of MPAs on the formation and stability of RFMOs. We find that MPAs can improve fisheries in terms of profits and stock if MPAs increase growth rates, even in the absence of further cooperation in an RFMO. Furthermore we find that within our simulations, MPAs generally improve the stability of coalitions when countries differ in fishing costs, but not when fishing costs are the same.

5.2. The model

5.2.1. Background

Coalition models

Coalition formation models used in fisheries follow two approaches: the cooperative game theory approach or the non-cooperative approach. There are clear links with the literature on international climate agreements (e.g. Nagashima et al., 2009).

In the cooperative approach it is assumed that an agreement is formed, and the analysis mainly focuses on how to distribute the benefits between countries such that all countries perceive the agreement as beneficial. Important contributions within the fisheries literature are Kaitala and Lindroos (1998), Li (1998), Pintassilgo and Duarte (2000) and Kronbak and Lindroos (2007).

In the non-cooperative approach the analysis focuses on which agreements can be formed in the first place, mainly using the internal and external stability concept due to d'Aspremont et al. (1983). Pintassilgo (2003) introduced the coalition stability approach with externalities into the fisheries, and has analyzed free-riding and the associated problems to reach an agreement with an application to tuna fisheries. Pintassilgo and Lindroos (2008) apply the coalition approach using the standard Gordon-Schaefer model and assuming symmetric players. They found that under these circumstances free-riding always prevails, i.e. no coalition can be formed, except in the case of two players, where the grand coalition is stable. Pham Do and Folmer (2006) study partial cooperation and derive some general results such as an increased effort level by the coalition if marginal cost decrease. Lindroos (2008) and Pintassilgo et al. (2010) relax the assumption of symmetric players and find that generally small coalitions can be formed, but that larger coalitions of players are still unstable. Long (2009), following Barrett (2003) and Carraro et al. (2009), adds a minimum participation level to the agreement. Such a participation rate generally increases stability and allows for higher degrees of cooperation. Long (2009) also includes monitoring cost, in the manner of McEvoy and Stranlund (2009) and shows that this may raise the minimum participation rate and thus cooperation even further.

MPA models

MPAs as a fisheries management tool have been modeled in a number of ways. A number of choices and modeling assumptions have to be made a priori which may have major impacts on the model results. Models can be non-spatial (Hannesson, 1998; Anderson, 2002), continuous in space (White et al., 2008) or discrete (Sanchirico and Wilen, 2001; Sanchirico, 2004; Ruijs and Janmaat, 2007). The first two model types usually split the stock in an inside and outside MPA part. The last model type usually uses metapopulation models in discrete patches. Another important assumption is the regime outside the MPA: does open access prevail outside the MPA (Hannesson, 1998; Sanchirico and Wilen, 2001), or does some kind of limited entry system exist (Sanchirico, 2004; Costello and Kaffine, 2010). Finally the scale is important, although it is mainly defined by the problem at hand: some papers analyze MPAs at the country scale as a game between fishermen and regulator (Beattie et al., 2002; Sanchirico, 2004; Ngoc, 2010), others analyze international settings involving several countries' EEZs (Sumaila, 2002; Ruijs and Janmaat, 2007; Punt et al., 2010). MPAs at the High Sea scale, however, have not yet been considered.

The High Sea scale brings about several interesting aspects that need to be considered when modeling MPAs: first of all the number of players is potentially large and not necessarily constant over time. Two-player games are therefore inherently limited and, as we will be confronted with the new member problem, n-player games are more appropriate. Second because an MPA is not a real no-take zone if it is not acknowledged and respected by all (or a very large) number of countries, we need an MPA that everyone can agree upon, i.e. its size must be acceptable to all.

5.2.2. Model setup

Consider the High Seas where a number countries fish. In order to coordinate their fishing efforts and to prevent overfishing countries can form an RFMO. We will refer to this RFMO as the effort coalition. Given some RFMO all countries may wish to negotiate an MPA in the High Sea. We study the impact of the designation of an MPA on the incentives to join an RFMO.

We assume that the countries will target nursery areas such that the MPA does not only protect part of the fish stock, but it also increases the internal growth rate of the stock in the MPA. The reason for this increase is called habitat or hot-spot effect: limiting the fishing activities improves the habitat quality and this increases the growth rate. Establishing an MPA would generate this effect (Sanchirico, 2004; Schnier, 2005a,b; Armstrong, 2007; Armstrong and Falk-Petersen, 2008).

An interesting feature of MPAs is that they must be acknowledged and respected by (almost) everyone to be successful. If someone fishes in an MPA the habitat remediation effect would be lost. We will assume that if a single country does not support the MPA, the agreement breaks down. Hence single deviations result in a full breakdown of the agreement and the habitat effect. As a consequence the MPA must be acceptable to and respected by all players. The formation of an MPA agreement is, therefore, a weakest link game as described by Hirshleifer (1983) and Sandler (1998).

We model the full game as a three stage game: at the first stage countries decide whether or not they sign-up to the RFMO, at the second stage they decide on an MPA size that is agreed upon by all, and the final stage effort levels are determined based on RFMO membership and the given MPA size. If countries cannot agree upon an MPA, its size is zero. We normalize the size of the fishing grounds to one, and assume that the exact spatial distribution – whether several small or a single large reserve is agreed upon – does not matter.

The equilibrium effort, profit and coalition outcomes for such a game (without MPAs) have been described by Pintassilgo et al. (2010). We construct a modified model to accommodate MPAs. The full game is solved by backward induction, to find equilibrium efforts, the equilibrium MPA size, and the stable effort coalitions. We now proceed to the formal analysis of this game.

5.2.3. Formal description

A set of N countries are fishing in the High Seas. They can coordinate their fishing effort in a coalition or decide to fish alone. The coalition is used for effort decisions only, because in the decisions on MPAs only the grand coalition is effective. We will assume that only a single coalition can form and that other players act as singletons, i.e. only one RFMO exists. Effort coalition $S \subseteq N$ is then a subset of countries that form a coalition. A coalition structure can be described by a vector $(\sigma_i)_{i \in N}$ with $\sigma_i \in \{0,1\}$ where 0 denotes non-signatories and 1 denotes signatories to effort coalition S.

We use a static Gordon-Schaefer model to describe fishing activities. The model is a combination of the coalition model used by Pintassilgo et al. (2010) and the fisheries MPA model of Punt et al. (2010) but in contrast to the last model we now explore a High Seas setting and asymmetric players as opposed to Exclusive Economic Zones and symmetric players[1].

The growth of the fish stock is logistic, carrying capacity is normalized to one, and the stock X is divided proportionally over the MPA and the remaining seas into inside (X_M) and outside the MPA (X_o). The carrying capacity in each area is assumed to be proportional to area. The internal growth rate is r_o outside the MPA, and due to nursery and habitat effects r_o + r_M inside the MPA. Consequently, if an MPA of size M is present, the stock growth is:

$$\begin{aligned}&\frac{dX}{dt}=\frac{dX_M}{dt}+\frac{dX_o}{dt}=\\&(r_o+r_M)MX\left(1-\frac{MX}{M}\right)+r_o(1-M)X\left(1-\frac{(1-M)X}{(1-M)}\right)=\\&(r_o+r_MM)X(1-X).\end{aligned} \tag{5.1}$$

Thus the internal growth rate of the stock can be described by the function $R(M)=r_o+r_MM$. The growth function collapses to the standard logistic growth function if no MPA is assigned.

In the Gordon-Schaefer model catch is proportional to stock (X) and effort (E) and some catchability parameter q. In our model we go beyond the capturing the concept of catchability by a single parameter, but rather describe it as a function of fishable area. Therefore catchability Q is a decreasing function in the size of the MPA, M. This is to reflect that catching fish gets harder if just a smaller area can be fished. If no MPA is assigned $Q(M)$ collapses to the baseline catchability parameter (q), i.e. $Q(0)=q$. In the simplest specification $Q(M)=q(1-M)$ which is what we will analyze here. We then have as harvest of player i :

$$H_i(E_i)=Q(M)E_iX. \tag{5.2}$$

If the stock is uniformly distributed, all growth can be caught in equilibrium and the total harvest, i.e. the sum of harvests of all players, should equal the growth:

[1]In order to keep the model analytically tractable we use a very simple modification of the static Gordon-Schaefer model that allows for the introduction of MPAs. More elaborate specifications may offer additional insights but do not alter the basic line of reasoning.

$$R(M)X(1-X)=Q(M)\sum_i E_i X \Leftrightarrow X=1-\frac{Q(M)\sum_{i\in N} E_i}{R(M)}. \tag{5.3}$$

Non-signatory countries wish to maximize their individual profits π_j. Players differ in harvesting costs but are identical in all other aspects. They value their harvest at price p and incur a cost c_i for every unit of effort used. Payoff π_j of a non-signatory country j is:

$$\pi_j = pQ(M)E_jX - c_jE_j = pE_j\left(1-\frac{Q(M)\left(E_j+\sum_{k\in N\setminus\{j\}} E_k\right)}{R(M)}\right) - c_jE_j, \quad \forall j \notin S. \tag{5.4}$$

As stated earlier countries can form a coalition to coordinate their fishing effort. Because costs are linear in effort, the coalition will let the fishing be carried out by its most efficient member. Let $c_S^{\min}$ denote the cost parameter of the country with the lowest effort cost in coalition S. The coalition consequently maximizes:

$$\pi_S = pQ(M)E_SX - c_S^{\min}E_S = pQ(M)E_S\left(1-\frac{Q(M)\left(E_S+\sum_{j\in N\setminus S} E_j\right)}{R(M)}\right) - c_S^{\min}E_S. \tag{5.5}$$

5.2.4. Analysis

The game is solved by backward induction. We start the analysis at the last stage.

Third stage

At the third stage the RFMO has formed, and countries have decided on the MPA size, consequently S and M are given. An effort coalition S is a partial agreement Nash equilibrium (Chander et al., 2006) if coalition S plays a best reply against all other singletons, and all singletons play a best reply coalition S and all other singletons. Assuming that both the coalition and the other players optimize their profits taking the effort decisions by others as given we can derive equilibrium efforts and payoffs for every coalition structure.

Let n denote $|N|$ and s denote $|S|$. Taking first order conditions of (5.4) and (5.5) with respect to E_i and E_S, respectively, and solving gives the following equilibrium effort and payoffs (see also Pintassilgo et al., 2010):

$$
\begin{aligned}
E_S^*(S) &= \left(\frac{R(M)}{Q(M)}\right)\left(\frac{n-s+1}{n-s+2}\left(1-\frac{c_S^{\min}}{pQ(M)}\right)-\frac{1}{n-s+2}\left(\sum_{j\in(-S)}\left(1-\frac{c_j}{pQ(M)}\right)\right)\right)\\
E_j^* &= \left(\frac{R(M)}{Q(M)}\right)\left(\frac{n-s+1}{n-s+2}\left(1-\frac{c_j}{pQ(M)}\right)-\frac{1}{n-s+2}\left(\left(1-\frac{c_S^{\min}}{pQ(M)}\right)+\sum_{k\in(-j)}\left(1-\frac{c_k}{pQ(M)}\right)\right)\right)\\
\pi_S^*(S) &= p\frac{R(M)}{(n-s+2)^2}\left(1-(n-s+1)\frac{c_S^{\min}}{pQ(M)}+\sum_{j\in(-S)}\frac{c_j}{pQ(M)}\right)^2\\
\pi_j^*(S) &= p\frac{R(M)}{(n-s+2)^2}\left(1-(n-s+1)\frac{c_j}{pQ(M)}+\frac{c_S^{\min}}{pQ(M)}+\sum_{k\in(-j)}\frac{c_k}{pQ(M)}\right)^2.
\end{aligned}
\tag{5.6}
$$

Where asterisks denote equilibrium strategies, $-S$ denotes $j \in N \setminus S,$ and $-j$ denotes $k|k \in N \setminus S \wedge k \neq j$. An interior solution of course requires all effort levels to be strictly positive. This requirement translates into two equations:

$$pq(1-M) > (n-s+1)c_S^{\min} - \sum_{j\in(-S)} c_j \tag{5.7}$$

$$pq(1-M) > (n-s+1)c_j - c_S^{\min} - \sum_{k\in(-j)} c_k. \tag{5.8}$$

Without loss of generality assume that we order the countries such that $c_1 \leq c_2 \ldots \leq c_n$. It can then be shown that (5.7) and (5.8) are met if (Pintassilgo et al., 2010):

$$pq(1-M) > \left(nc_n - \sum_{i=1}^{n-1} c_i\right) \tag{5.9}$$

which gives us a restriction on the parameter space.

Second stage:

In the second stage countries either decide on the size of an MPA or, if they do not reach a unanimous agreement, no MPA is formed. This is a feature of the weakest link game. If the countries wish to design an MPA that is acceptable to all, such that full cooperation is reached, the MPA must increase profits for all countries relative to the no MPA case.

We will assume that the decision process on the size is as follows: an arbitrary country $i \in N$ is selected to make a proposal on the size M of the

MPA[2]. If the size is unanimously agreed upon, the MPA is installed. If not, negotiations fail and the MPA size is zero. The size of the MPA is now determined by a weakest link game (Hirshleifer, 1983; Sandler, 1998).

To see this, note that countries are asymmetric and consequently receive different net benefits due to the MPA. Consider the payoff of country *i*, selected to announce the MPA size *M*: if an MPA increases the payoff of everyone, clearly country *i* picks a size $M > 0$. However, consider country *j* that gains if instead of *M* a smaller size *M′* is selected. Then country *j* can create its desired *M′* by first signing up to the agreement, creating an MPA of size *M*, and then starting to fish in a part of the MPA such that *M* is reduced to *M′*. The other countries could then of course declare the MPA agreement null and void, but this threat is not credible as long as *M′* still offers net benefits to all other countries compared to the no MPA case. Consequently, in equilibrium player *i* will pick M^* such that it is equal to the smallest optimum size M_i of all countries $i \in N$.

Taking the first order conditions of π_S^* and π_j^* with respect to *M* yields:

$$\frac{\partial \pi_S}{\partial M} = \frac{2p\left(\frac{\sum_{j\in(-S)} c_j - (n-s+1)c_S^{\min}}{pq(1-M)^2}\right)\left(1-\frac{(n-s+1)c_S^{\min} - \sum_{j\in(-S)} c_j}{pq(1-M)}\right)(r_o + r_M M)}{(n-s+2)^2} + \frac{pr_M\left(1-\frac{(n-s+1)c_S^{\min} - \sum_{j\in(-S)} c_j}{pq(1-M)}\right)^2}{(n-s+2)^2} = 0 \tag{5.10}$$

$$\frac{\partial \pi_j}{\partial M} = \frac{2p\left(\frac{c_S^{\min} + \sum_{k\in(-j)} c_k - (n-s+1)c_j}{pq(1-M)^2}\right)\left(1-\frac{(n-s+1)c_j - c_S^{\min} - \sum_{k\in(-j)} c_k}{pq(1-M)}\right)(r_o + r_M M)}{(n-s+2)^2} + \frac{pr_M\left(1-\frac{(n-s+1)c_j - c_S^{\min} - \sum_{k\in(-j)} c_k}{pq(1-M)}\right)^2}{(n-s+2)^2} = 0. \tag{5.11}$$

[2]This procedure is similar to the formulation of a minimum participation rule for an international environmental agreement in Weikard et al. (2009).

The optimal M from the perspective of the effort coalition is then[3]:

$$M_S^* = 1 + \frac{\left((n-s+1)c_S^{\min} - \sum_{j\in(-S)} c_j\right)}{2pq} - \sqrt{\frac{\left(\sum_{j\in(-S)} c_j - (n-s+1)c_S^{\min}\right)^2}{4p^2q^2} + \frac{2\left((n-s+1)c_S^{\min} - \sum_{j\in(-S)} c_j\right)}{pq}\left(\frac{r_o}{r_M}+1\right)} \tag{5.12}$$

and from the perspective of an individual that is not in the coalition:

$$M_j^* = 1 + \frac{\left((n-s+1)c_j - c_S^{\min} - \sum_{k\in(-j)} c_k\right)}{2pq} - \sqrt{\left[\frac{\left(c_S^{\min} + \sum_{k\in(-j)} c_k - (n-s+1)c_j\right)^2}{4p^2q^2} + \frac{2\left((n-s+1)c_j - c_S^{\min} - \sum_{k\in(-j)} c_k\right)}{pq}\left(\frac{r_o}{r_M}+1\right)\right]}. \tag{5.13}$$

The equilibrium proposal M^* is:

$$M^* = \min\left(M_S^*, \left(M_j^*\right)_{j\in(-S)}\right). \tag{5.14}$$

Which M^* is the smallest hinges upon the distribution of the cost parameters and the effort coalition formed. This effort coalition itself is also determined by the distribution of the cost, price and the baseline catchability (Pintassilgo et al., 2010). It can be shown that an interior solution for M requires (see Appendix 5.A1.):

$$\frac{pqr_M}{2r_o + r_M} - \left(nc_n - \sum_{i=1}^{n-1} c_i\right) > 0 \tag{5.15}$$

[3]Both (5.11) and (5.12) produce in fact three solutions, but two of those are corner solutions if we assume that all parameters are > 0 and effort > 0. The proof is in Appendix 5.A1.

$$nc_1 - \sum_{i=2}^{n} c_i > 0 \tag{5.16}$$

Turning to the effect of parameters: A necessary but not sufficient condition for (5.12) and (5.13) is $(n-s+1)c_S^{\min} - \sum_{j\in(-S)} c_j > 0$ and $(n-s+1)c_j - c_S^{\min} - \sum_{k\in(-j)} c_k > 0$ (Appendix 5.A1.). The effect of r_M and r_o can then readily be seen from the solutions: r_M increases M and r_o decreases M. This is in line with expectation, because r_M makes an MPA more worthwhile, whereas an increase in r_o makes it relatively less attractive.

The effect of other parameters can be determined with the help of the implicit function theorem as explained in Appendix 5.A2. The full list of effects is shown in Table 5.1. The signs are as expected: an increase in the price of fish (p) increases the desired MPA size because the extra growth from the MPA is valued higher. Similarly an increase in own effort cost decreases the desired MPA size, because MPAs make fishing more expensive in the first place. An increase in costs of other countries increases the desired MPA size because opponents will fish less, and this means that the country (or coalition) under consideration can catch a larger share of the growth bonus. Consequently a larger MPA with more growth is then desired.

Under full symmetry $c_j = c_S^{\min} = c$. The optimal M^* is the same for everyone and the game is a degenerated weakest link game. The optimal M^* in (5.12) and (5.13) simplifies in both cases to:

$$M^* = 1 + \frac{c}{2pq} - \sqrt{\frac{c^2}{4p^2q^2} + \frac{2c}{pq}\left(\frac{r_o}{r_M} + 1\right)}. \tag{5.17}$$

Since everyone is symmetric and (5.17) is the result of an optimization, as long as M^* exists it is better than the no-cooperation case, where $M^* = 0$. Hence in the symmetric case an MPA will always be designated. The full benefits of

Table 5.1: Comparative statics of parameters with respect to optimal MPA size

	r_M	r_o	p	c_S^{min}	c_j	c_k
Effect on M_S	+	-	+	-	+	n.a.
Effect on M_j	+	-	+	+	-	+

Assumptions: All parameters >0, $0 \le M \le 1$, $j \in (-S)$ and $k \in (-j)$. A + denotes that MPA size and the parameter move in the same direction, a – denotes the opposite direction, and n.a. denotes not applicable. Note that the sign of the derivative with respect to c_j and c_k only hold for marginal changes, i.e. as long as they do not affect the formed coalition.

this MPA may not be captured because there is a non-cooperative equilibrium in terms of effort.

First stage

In the first stage countries decide upon their membership of the RFMO. Their membership decision is driven by their payoff in the last stage. We use the concept of internal and external stability due to d'Aspremont (1983), combined with the optimal sharing rule as defined in Weikard (2009).

Using the equilibrium payoffs in (5.6) an effort coalition S is internally stable if and only if:

$$\pi_i^*(S) \geq \pi_i^*(S-\{i\}) \quad \forall i \in S. \tag{5.18}$$

That is coalition S is internally stable if and only if every country in S gets a payoff that is larger than or equal to the payoff it gets if it free-rides. Similarly a coalition S is said to be externally stable if and only if:

$$\pi_j^*(S) \geq \pi_j^*(S \cup \{j\}) \quad \forall j \notin S. \tag{5.19}$$

That is coalition S is externally stable if and only if the payoff of every country outside the coalition is larger than or equal to the payoff when joining the coalition.

Clearly, to check whether (5.18) holds, we need to specify how the profits within the coalition are shared. We introduce the Claim Rights Condition (Weikard, 2009) which states that every country in a coalition S should at least get its outside option payoff (its claim), provided the coalition payoff is large enough to satisfy all claims. The remaining surplus can then be shared among coalition members. Hence if:

$$\pi_S^*(S) \geq \sum_{i \in S} \pi_i^*(S-\{i\}) \tag{5.20}$$

is satisfied and the Claim Rights Condition is used for sharing, then (5.18) holds for every $i \in S$ and coalition S is internally stable. Consequently to check internal stability of coalition S it is enough to check whether (5.20) holds. Moreover, using the Claim Rights Condition it can be shown that every coalition $S \subset \boldsymbol{N}$ is externally unstable if there exists an enlargement $S \cup \{j\} \in N \setminus S$ such that $S \cup \{j\}$ is internally stable (Weikard, 2009). Hence to check whether a coalition is both internally and externally stable, it is sufficient to check for which coalitions (5.20) holds and within that subset find the coalitions that cannot be enlarged to another coalition in that subset.

Using the payoffs specified in (5.6) in condition (5.20) we find that internal stability can be satisfied if:

$$\frac{(n-s+3)^2}{(n-s+2)^2}\left(1-\frac{1}{pq\left(1-M^*(S)\right)}\left((n-s+1)c_S^{\min}-\sum_{j\in(-S)}c_j\right)\right)^2 \geq$$
$$\sum_{i\in S}\left(1-\frac{1}{pq\left(1-M^*(S-\{i\})\right)}\left((n-s+2)c_i-c_{(S-i)}^{\min}-\sum_{j\in(-S)}c_j\right)\right)^2 \tag{5.21}$$

where $M^*(S)$ denotes the equilibrium MPA size under coalition S and $M^*(S - \{i\})$ denotes the equilibrium MPA size when coalition $S - \{i\}$ is formed. Condition (5.21) shows that MPAs have a direct effect on stability but only through their effect on the catchability and not through the extra growth in the MPA. Indirectly, though the extra growth does play a role since it determines the equilibrium size of M. From (5.21) we see that an MPA does not necessarily improve stability in each particular case, as can be seen for full symmetry below.

If we assume full symmetry (5.21) simplifies to:

$$\left(\frac{n-s+3}{n-s+2}\right)^2\left(1-\frac{c}{pq(1-M)}\right)^2 \geq s\left(1-\frac{c}{pq(1-M)}\right)^2 \Leftrightarrow$$
$$\left(\frac{n-s+3}{n-s+2}\right)^2 \geq s \tag{5.22}$$

which shows that MPAs do not have an effect on stability under full symmetry. Equation (5.22) is consistent with the result of Pintassilgo and Lindroos (2008) that under full symmetry no coalition is stable except in the case of $n = 2$ when full cooperation is stable. This result reflects the full leakage in this game, i.e. coalition members' reductions in effort are fully offset by non-signatories. Under asymmetry coalitions have two effects: they reduce over-fishing and they reduce costs of fishing. In symmetry the latter does not apply and because of is full leakage no coalition is stable. The exception is the two-player game where leakage cannot occur, once a coalition is established.

As a special case let us consider what happens if all countries are identical, and there is no RFMO, when a new entrant arrives with a different cost structure as in Lindroos (2008). Hence we assume all countries but one are identical, and that the non-identical country has higher costs, i.e. $c_1 = c_2 = c_3 ... = c_{n-1} = c$ and $c_n > c$. Then (5.21) simplifies to:

$$\frac{9}{4}\left(1-\frac{c}{pq(1-M_N)}\right)^2 \geq (n-1)\left(1-\frac{c}{pq(1-M_N)}\right)^2+\left(1-\frac{2c_n-c}{pq(1-M_n)}\right)^2. \tag{5.23}$$

In (5.23) M_N refers to the MPA size under full cooperation, and M_n is the MPA size demanded by the n^{th} player, the high cost country. If one of the low cost countries defects, inspection of (5.12) and (5.13) shows that the MPA size demanded is not affected, but if the high cost country defects it demands a smaller MPA, as higher costs induce smaller MPAs (see Table 5.1). If MPAs would not exists (5.23) simplifies further to:

$$\frac{9}{4}\left(1-\frac{c}{pq}\right)^2 \geq (n-1)\left(1-\frac{c}{pq}\right)^2 + \left(1-\frac{2c_n-c}{pq}\right)^2. \tag{5.24}$$

If we subtract (5.23) from (5.24) we get the reduction on both sides that is due to the MPA. The reduction on the left hand side is:

$$\frac{9c}{4pq}\left(\frac{c}{pq}\left(1-\frac{1}{(1-M_N)^2}\right)-2\left(1-\frac{1}{(1-M_N)}\right)\right) \tag{5.25}$$

and on the right hand side:

$$\begin{aligned}&\frac{(n-1)c}{pq}\left(\frac{c}{pq}\left(1-\frac{1}{(1-M_N)^2}\right)-2\left(1-\frac{1}{1-M_N}\right)\right)+\\&\frac{2c_n-c}{pq}\left(\frac{2c_n-c}{pq}\left(1-\frac{1}{(1-M_n)^2}\right)-2\left(1-\frac{1}{1-M_n}\right)\right).\end{aligned} \tag{5.26}$$

For large values of n, the reduction on the left hand side (5.25) is less than the reduction on the right hand side (5.26). This shows the potential stabilizing effect of MPAs. However, the effect in general, for arbitrary S the stability cannot be seen from these equations. Therefore we investigate the effects with simulations.

5.2.5. Simulations

Here we illustrate the model with a number of simulations. The (arbitrary) parameter values are shown in Table 5.2. They have been selected according to the restrictions for an interior solution in (5.9), (5.15) and (5.16). To get more general conclusions their effects are later tested in a sensitivity analysis. We calculate the profits of each effort coalition using the solutions in (5.6), (5.12) and (5.13). We assume that the MPA assigned is the smallest MPA among the solutions to (5.12) and (5.13). To investigate the influence of MPAs on stability, we do the calculations twice: once with the MPA sizes as determined by (5.12) and (5.13) and once assuming that no MPA is formed. In a four player game there are 12 possible coalition structures. The results are shown in Table 5.3 and 5.4.

Table 5.2: Arbitrary parameter values for the example

Parameter	Value
Number of countries (n)	4
Number of coalition structures	12
Price of fish (p)	20
Cost per unit of effort (c_i)	[5, 5.5, 6, 7]
Growth rate outside MPA (r_o)	0.2
Extra growth rate inside MPA (r_M)	0.8
Original catchability (q_o)	1

From these tables it can be seen that the presence of an MPA increases the benefits to all players, and has a positive influence on stock size. However as can be seen in Table 5.4 the more free-riders are present the smaller the MPAs formed.

Using the Claim Rights Condition we can determine which coalitions are internally and externally stable. In principle coalitions are internally stable if the sum of the outside option payoffs is smaller than or equal to the payoff of the coalition, and a coalition is externally stable if no enlargement of that

Table 5.3: Stock size, profits of the coalition and free-riders, and global profits for each coalition structure in absence of an MPA

Coalition structure	Stock size	Profit coalition	Profit country 1	Profit country 2	Profit country 3	Profit country 4	Global profits
(1),(2),(3),(4)	0.44	-	0.14	0.10	0.07	0.03	0.34
(1,2),(3),(4)	0.48	0.20	-	-	0.12	0.06	0.39
(1,3),(2),(4)	0.47	0.19	-	0.15	-	0.06	0.40
(1,4),(2),(3)*	0.46	0.17	-	0.13	0.10	-	0.40
(1),(2,3),(4)	0.47	0.15	0.19	-	-	0.06	0.40
(2,4),(1),(3)*	0.46	0.13	0.17	-	0.10	-	0.40
(3,4),(1),(2)	0.46	0.10	0.17	0.13	-	-	0.40
(2,3,4),(1)	0.51	0.22	0.27	-	-	-	0.48
(1,3,4),(2)	0.51	0.27	-	0.22	-	-	0.48
(1,2,4),(3)	0.52	0.28	-	-	0.19	-	0.47
(1,2,3),(4)	0.53	0.32	-	-	-	0.13	0.46
(1,2,3,4)	0.63	0.56	-	-	-	-	0.56

Coalition members share the coalition profit. Because we do not specify a sharing rule, we do not provide profit for individual coalition members. Stable coalitions have been marked with an asterisk.

Table 5.4: Stock size, profits of the coalition and free-riders, global profits and equilibrium MPA for each coalition structure

Coalition structure	Stock size	Profit coalition	Profit countr y 1	Profit country 2	Profit countr y 3	Profit countr y 4	Global profits	MPA size
(1),(2),(3),(4)	0.45	-	0.17	0.12	0.08	0.03	0.40	0.05
(1,2),(3),(4)	0.50	0.28	-	-	0.16	0.07	0.51	0.10
(1,3),(2),(4)*	0.49	0.25	-	0.19	-	0.06	0.50	0.09
(1,4),(2),(3)*	0.51	0.28	-	0.19	0.13	-	0.60	0.20
(1),(2,3),(4)	0.49	0.19	0.25	-	-	0.06	0.50	0.09
(2,4),(1),(3)*	0.51	0.19	0.28	-	0.13	-	0.60	0.20
(3,4),(1),(2)*	0.51	0.13	0.28	0.19	-	-	0.60	0.20
(2,3,4),(1)	0.57	0.32	0.44	-	-	-	0.76	0.27
(1,3,4),(2)	0.57	0.44	-	0.32	-	-	0.76	0.27
(1,2,4),(3)	0.57	0.46	-	-	0.25	-	0.72	0.22
(1,2,3),(4)	0.57	0.47	-	-	-	0.16	0.63	0.14
(1,2,3,4)	0.69	0.91	-	-	-	-	0.91	0.32

Coalition members share the coalition profit. Because we do not specify a sharing rule we do not provide profit for individual coalition members. Stable coalitions are marked with an asterisk.

Table 5.5: Desired MPA sizes from the coalition and individual countries

Coalition structure	M_s	M_1	M_2	M_3	M_4
(1),(2),(3),(4)	-	0.6029	0.3859	0.2466	0.0546
(1,2),(3),(4)	0.5475	-	-	0.2711	0.1044
(1,3),(2),(4)	0.5000	-	0.3541	-	0.0872
(1,4),(2),(3)	0.4203	-	0.2970	0.2013	-
(1),(2,3),(4)	0.3541	0.5000	-	-	0.0872
(2,4),(1),(3)	0.2970	0.4203	-	0.2013	-
(3,4),(1),(2)	0.2013	0.4203	0.2970	-	-
(2,3,4),(1)	0.2711	0.3541	-	-	-
(1,3,4),(2)	0.3541	-	0.2711	-	-
(1,2,4),(3)	0.3859	-	-	0.2234	-
(1,2,3),(4)	0.4581	-	-	-	0.1407
(1,2,3,4)	0.3246	-	-	-	-

Countries is in the effort coalition desire the coalition MPA size, therefore we do not provide an individual MPA size for countries in the coalition.

coalition exist that is internally stable. Stable coalitions in Table 5.3 and 5.4 have been marked with an asterisk.

Looking at stability we see that the introduction of MPAs increases stability, such that two additional coalitions are stable, although the two extra coalitions do not perform better than the coalitions that were also stable in absence of the MPA. Full cooperation is still not reached though.

The desired MPA sizes are shown in Table 5.5. It can be seen from Table 5.5 that countries with higher costs generally desire lower MPAs, but that the size is also influenced by the number of active fishing countries, i.e. the size of the coalition. Consider countries 3 and 4: they have the highest effort costs, and therefore when they free-ride they effectively determine the MPA size. The MPA size they desire increases with coalition size, but by how much depends on the composition of the coalition. The opposite holds for the countries with the lowest effort costs, countries 1 and 2.

To investigate the effect of MPAs on stability in general we have run simulations for the above four player game while varying some parameters, and keeping other parameters constant, such that interior solutions are still possible in all coalitions. We have also investigated the effect of larger MPAs

Table 5.6: Stability of coalitions under differing parameter values

$p; c_1; c_2; c_3; c_4; r_0; r_M$	With MPA		Without MPA	
	No. stable coalitions	Maximum size	No. stable coalitions	Maximum size
20; 5; 5.5; 6; 7; 0.2; 0.8	4	2	2	2
25; 5; 5.5; 6; 7; 0.2; 0.8	3	2	1	1
40; 5; 5.5; 6; 7; 0.2; 0.8	2	2	1	1
75; 5; 5.5; 6; 7; 0.2; 0.8	1	1	1	1
200; 50; 55; 60; 70; 0.2; 0.8	4	2	2	2
200; 50; 52; 53; 80; 0.2; 0.8	3	2	3	2
200; 50; 52; 70; 75; 0.2; 0.8	3	3	4	2
200; 50; 60; 70; 75; 0.2; 0.8	4	2	1	2
200; 50; 52; 53; 54; 0.2; 0.8	1	1	1	1
200; 50; 55; 60; 70; 0.1; 0.8	5	2	2	2
200; 50; 55; 60; 70; 0.05; 0.8	3	3	2	2
200; 50; 55; 60; 70; 0.2; 1.5	5	2	2	2

by decreasing r_0, keeping other parameters fixed. In Table 5.6 we list the parameter values and the size and number of stable coalitions with and without MPAs.

From Table 5.6 we see that MPAs increase stability, either by increasing the number of stable coalitions or by increasing the maximum size of the coalition. MPAs do not increase stability if the parameter setting is almost full symmetry, i.e. when the cost parameters are all almost equal but one (line 6 of the results in Table 5.6) or when the cost parameters are all almost equal (line 9 of the results in Table 5.6)

The latter occurs if countries are (nearly) identical or when the price of fish (p) is large compared to the costs of effort. In all other cases MPAs improve stability suggesting a positive role for MPAs in coalition formation. We have not analyzed the effect of baseline catchability (q). This parameter, however, always appears in combination with the price of fish, both in the determination of the size of the MPA, and in the stability condition. Therefore its effects are the same as those of p.

From Table 5.6 it can also be inferred that the potential gains of an MPA are reaped better by larger coalitions. We know from Table 5.1 that a decrease in the growth rate of the fish stock outside the MPA (r_o) or an increase inside the MPA (r_M) increases the MPA size. The stability of effort coalitions in Table 5.6 also increases when r_o goes down when an MPA is present, i.e. larger coalitions are better equipped to capture the additional benefits of an MPA and consequently are stabilized by the presence of an MPA. The growth bonus in the MPA (r_M) has a similar but opposite effect: an increase in r_M increases stability in the MPA case.

5.3. Discussion & conclusions

In this chapter we investigate the influence of MPAs on the formation of Regional Fisheries Management Organizations (RFMOs). We have extend the classic Gordon-Schaefer model to accommodate MPAs, and we link this adjusted model to a coalition formation model, generally used in the literature of international environmental agreements. Our results are in agreement with the results of Pintassilgo et al. (2010): under full symmetry no coalition is stable, but the introduction of asymmetry in cost parameters does stabilize a number of coalitions. The inclusion of MPAs, however, offers additional insights.

Given that MPAs increase the internal growth rate of the stock, they tend to have a positive influence on both stock and profits. MPAs basically make the stock more resilient against large harvest. This increased resilience comes at a price: reduced catchability; but in an interior solution this is a price that countries are willing to pay.

Another observation is that MPAs in the High Seas constitute a weakest link public good. The observation that MPAs are a weakest link public good in the absence of enforcement carries over to terrestrial parks when enforcement is lacking and hard to establish. In the model we have taken this observation to extremes, i.e. an MPA loses its extra growth bonus if any country fishes in the MPA. Nevertheless even if the weakest link property would hold in a less strict form, the property still implies that MPA agreements form as if under a minimum participation constraint, albeit not requiring full cooperation. Barrett (2003) has shown that such participation constraints stabilize larger agreements. In this chapter we show that such an agreement may make it easier to agree on other issues.

The results of the simulations in this chapter show that MPAs tend to increase stability of RFMOs, for most of the analyzed parameter values, and that larger MPAs do a better job at this than smaller MPAs. The reason is that the benefits of an MPA can be reaped better by a coalition than by free-riders, because coalitions generally have a cost advantage. As we have seen, free-riding remains a problem, but it is reduced by the presence of MPAs. This result hinges on three important assumptions: MPAs increase growth, the stock is uniformly distributed and decisions on MPA size are made in the setting of a weakest link public good. If these conditions do not *exactly* hold in reality, all seem to be reasonable to approximations. MPAs have been found to increase growth rates inside the MPA (Lester et al., 2009; Stewart et al., 2009) and the fact that a number of High Sea MPAs have been installed, albeit only for trawling (North-East Atlantic Fisheries Commission, 2009), suggests that decisions on MPAs can be taken even in the absence of enforcement. Even if the stock is not uniformly distributing, it is likely that some spill-over will take place. In practice our model can be adjusted for this by decreasing the growth bonus parameter.

In this chapter we take an important first step in the analysis of how MPAs influence the formation of coalitions for fishing effort. We have shown, for the first time, that the introduction of MPAs in the High Seas have the potential to increase the stability of RFMOs, although not in all cases. In future research it will be useful to address how MPAs function in a dynamic setting and whether this modifies the result. The actual effects of MPAs and whether or not they increase growth, as well as the migration patterns are other important features that need to be addressed.

5.A1. Appendix I: Necessary and sufficient conditions for an optimal M

First we will show that only one solution of equation 5.11 is admissible, if we assume positive parameters and a positive effort level. Then we will show the necessary and sufficient conditions for an optimal M_j^* for this specific solution. The conditions for M_S^* follow the same line of reasoning.

The three solutions to M_j^* are:

$$M_j^* = \frac{pq-(n-s+1)c_j + c_S^{\min} + \sum_{k\in(-j)} c_k}{pq}$$

$$M_j^* = 1 + \frac{\left((n-s+1)c_j - c_S^{\min} - \sum_{k\in(-j)} c_k\right)}{2pq} + \sqrt{\left[\frac{\left(c_S^{\min} + \sum_{k\in(-j)} c_k - (n-s+1)c_j\right)^2}{4p^2q^2} + \frac{2\left((n-s+1)c_j - c_S^{\min} - \sum_{k\in(-j)} c_k\right)}{pq}\left(\frac{r_o}{r_M}+1\right)\right]} \tag{5.A1}$$

$$M_j^* = 1 + \frac{\left((n-s+1)c_j - c_S^{\min} - \sum_{k\in(-j)} c_k\right)}{2pq} - \sqrt{\left[\frac{\left(c_S^{\min} + \sum_{k\in(-j)} c_k - (n-s+1)c_j\right)^2}{4p^2q^2} + \frac{2\left((n-s+1)c_j - c_S^{\min} - \sum_{k\in(-j)} c_k\right)}{pq}\left(\frac{r_o}{r_M}+1\right)\right]}.$$

Substituting the first solution for M_j^* in equation (5.A1) into the solution for effort level E_j^* in equation (5.6) gives an effort level of 0. Therefore this solution is a corner solution and not admissible.

A necessary condition for an interior solution for M_j^* requires that the discriminant of (5.A1) ≥ 0 , i.e.:

$$\left((n-s+1)c_i - c_S^{\min} - \sum_{k\in(-j)} c_k\right)^2 + 8pq\left(\frac{r_o}{r_M}+1\right)\left((n-s+1)c_i - c_S^{\min} - \sum_{k\in(-j)} c_k\right) > 0, \quad (5A.2)$$

which can be satisfied under two cases. Either:

$$\left((n-s+1)c_j - c_S^{\min} - \sum_{k\in(-j)} c_k\right) > 0 \text{ and } \left((n-s+1)c_j - c_S^{\min} - \sum_{k\in(-j)} c_k\right) > -8pq\left(\frac{r_o}{r_M}+1\right) \quad (5A.3)$$

or

$$\left((n-s+1)c_j - c_S^{\min} - \sum_{k\in(-j)} c_k\right) < 0 \text{ and } \left((n-s+1)c_j - c_S^{\min} - \sum_{k\in(-j)} c_k\right) < -8pq\left(\frac{r_o}{r_M}+1\right). \quad (5A.4)$$

In (5A.3), if the first inequality holds then, by the assumption of positive parameters the second condition of the pair also holds.

The second pair can never occur in an interior solution. The proof is by contradiction:

If $\left((n-s+1)c_j - c_S^{\min} - \sum_{k\in(-j)} c_k\right) < 0$ then the root in the solutions in equation (5A.1) is smaller than $\frac{\left((n-s+1)c_j - c_S^{\min} - \sum_{k\in(-j)} c_k\right)}{2pq}$ and therefore an interior solution for M_i^* also requires $1+\frac{(n-s+1)c_j - c_S^{\min} - \sum_{k\in(-j)} c_k}{2pq} > 0$ to hold. This implies that $2pq > \sum_{k\in(-j)} c_k + c_S^{\min} - (n-s+1)c_j$. Inserting this as a minimum value for $2pq$ in (5A.4) we get:

$$\left((n-s+1)c_j - c_S^{\min} - \sum_{k\in(-j)} c_k\right) + 4\left(\sum_{k\in(-j)} c_k + c_S^{\min} - (n-s+1)c_i\right) < 0 \Leftrightarrow$$

$$\underbrace{\left((n-s+1)c_j - c_S^{\min} - \sum_{k\in(-j)} c_k\right)}_{I}\underbrace{\left(1-4\left(\frac{r_o}{r_M}+1\right)\right)}_{II} < 0 \tag{5A.5}$$

From the first inequality in (5A.4) we know that part *I* in equation (5A.5) is negative, but because part *II* in (5A.5) is also negative, the final result is positive, hence (5A.5) cannot be true, hence (5A.5) can never hold in an interior solution. We have established that in an interior solution $(n-s+1)c_j - c_S^{\min} - \sum_{k\in(-j)} c_k > 0$, which in turn rules out the second solution for M_j^* in (5.A1), because, by the assumption of positive parameters, solution 2 in (5A.1) can only produce interior solutions for $(n-s+1)c_j - c_S^{\min} - \sum_{k\in(-j)} c_k < 0.$ Furthermore this fact is used in the determination of the effects of r_o and r_M, and the sign of the derivatives with respect to parameters in appendix 5.A2.

The sufficient conditions for an interior solution for M_j^* require:

$$0 < \left[1 + \frac{(n-s+1)c_j - c_S^{\min} - \sum_{k\in(-j)} c_k}{2pq} - \sqrt{\frac{\left((n-s+1)c_j - c_S^{\min} - \sum_{k\in(-j)} c_k\right)^2}{4p^2q^2} + 2\frac{(n-s+1)c_j - c_S^{\min} - \sum_{k\in(-j)} c_k}{pq}\left(\frac{r_o}{r_M}+1\right)}\right] < 1 \tag{5A.6}$$

Let $a = (n-s+1)c_j - c_S^{\min} - \sum_{k\in(-j)} c_k$. Taking the left part of (5A.6) we get:

$$0 < 1 + \frac{a}{2pq} - \sqrt{\frac{a^2}{4p^2q^2} + 2\frac{a}{pq}\left(\frac{r_o}{r_M} + 1\right)} \Leftrightarrow$$

$$\left(1 + \frac{a}{2pq}\right)^2 > \frac{a^2}{4p^2q^2} + 2\frac{a}{pq}\left(\frac{r_o}{r_M} + 1\right) \Leftrightarrow$$

$$\frac{pq + a\left(1 - 2\left(\frac{r_o}{r_M} + 1\right)\right)}{pq} > 0 \Leftrightarrow \tag{5A.7}$$

$$\frac{pqr_M}{2r_o + r_M} - a > 0.$$

The maximum value of a is $nc_n - \sum_{i=1}^{n-1} c_i$. Thus we arrive at (5.15). Similarly taking the right part of (5A.6) we get:

$$1 + \frac{a}{2pq} - \sqrt{\frac{a^2}{4p^2q^2} + 2\frac{a}{pq}\left(\frac{r_o}{r_M} + 1\right)} < 1 \Leftrightarrow$$

$$\frac{a^2}{4p^2q^2} + 2\frac{a}{pq}\left(\frac{r_o}{r_M} + 1\right) > \left(\frac{a}{2pq}\right)^2 \Leftrightarrow \tag{5A.8}$$

$$\frac{2a\left(\frac{r_o}{r_M} + 1\right)}{pq} > 0$$

For (5A.7) to hold $a > 0$. The minimum value of a is $nc_1 - \sum_{i=2}^{n} c_i$. Thus we arrive at (5.16).

5.A2. *Appendix II: Derivatives with respect to parameters*

The solution to (5.10) and (5.11) consist of two polynomials of power three that can both be factorized in quadratic and linear parts. This function is for (5.10):

$$\frac{\left((n-s+1)c_S^{\min}-\sum_{j\in(-S)}c_j+pq(M-1)\right)\times\left(\begin{array}{l}pqr_M M_S^{\;2}-r_M\left((n-s+1)c_S^{\min}-\sum_{j\in(-S)}c_j+2pq\right)M_S\\-\left((n-s+1)c_S^{\min}-\sum_{j\in(-S)}c_j\right)(2r_o+r_M)+pqr_M\end{array}\right)}{pq(s-n-2)^2(M_S-1)^3}=0 \tag{5A.9}$$

and for (5.11):

$$\frac{\left((n-s+1)c_j-c_S^{\min}\sum_{k\in(-j)}c_k+pq\left(M_j-1\right)\right)\times\left(\begin{array}{l}pqr_M M_i^{\;2}-r_M\left(\left((n-s+1)c_j-c_S^{\min}-\sum_{k\in(-j)}c_k\right)+2pq\right)M_j-\\\left(\left((n-s+1)c_j-c_S^{\min}-\sum_{k\in(-j)}c_k\right)(2r_o+r_M)+pqr_M\right)\end{array}\right)}{pq(s-n-2)^2\left(M_j-1\right)^3}=0. \tag{5A.10}$$

The interior solutions considered in the chapter are the solutions in which the quadratic function in the numerator is equal to zero. Consequently the implicit functions determining the optimal M for a coalition player and a free-rider are respectively:

$$\left(\begin{array}{l}pqr_M M_S^{\;2}-r_M\left((n-s+1)c_S^{\min}-\sum_{j\in(-S)}c_j+2pq\right)M_S-\\\left(\left((n-s+1)c_S^{\min}-\sum_{j\in(-S)}c_j\right)(2r_o+r_M)+pqr_M\right)\end{array}\right)=0 \tag{5A.11}$$

$$\left(\begin{array}{l} pqr_M M_j^{\,2} - r_M\left(\left((n-s+1)c_j - c_S^{\min} - \sum_{k\in(-j)} c_k\right) + 2pq\right)M_j \\ -\left((n-s+1)c_j - c_S^{\min} - \sum_{k\in(-j)} c_k\right)(2r_o + r_M) + pqr_M \end{array}\right) = 0. \tag{5A.12}$$

Function (5A.10) has the following derivatives with respect to individual parameters:

$$\frac{\partial M_S^*}{\partial p} = \frac{-q(M_S-1)^2}{-\left(2pq(1-M_S)+\left((n-s+1)c_S^{\min} - \sum_{j\in(-S)} c_j\right)\right)} \tag{5A.13}$$

$$\frac{\partial M_S^*}{\partial c_S^{\min}} = \frac{(n-s+1)\left(2r_o + r_M(1+M_S)\right)}{-r_M\left(2pq(1-M_S)+\left((n-s+1)c_S^{\min} - \sum_{j\in(-S)} c_j\right)\right)} \tag{5A.14}$$

$$\frac{\partial M_S^*}{c_j} = \frac{-\left(2r_o + r_M(1+M_S)\right)}{-r_M\left(2pq(1-M_S)+\left((n-s+1)c_S^{\min} - \sum_{j\in(-S)} c_j\right)\right)}. \tag{5A.15}$$

Function (5A.12) has the following derivatives with respect to individual parameters:

$$\frac{\partial M_j^*}{\partial p} = \frac{-q\left(M_j-1\right)^2}{-\left(2pq\left(1-M_j\right)+\left((n-s+1)c_j - c_S^{\min} - \sum_{k\in(-j)} c_k\right)\right)} \tag{5A.16}$$

$$\frac{\partial M_j^*}{\partial c_j} = \frac{(n-s+1)\left(2r_o + r_M\left(1+M_j\right)\right)}{-r_M\left(2pq\left(1-M_j\right)+\left((n-s+1)c_j - c_S^{\min} - \sum_{k\in(-j)} c_k\right)\right)} \tag{5A.17}$$

$$\frac{\partial M_j^*}{\partial c_S^{\min}} = \frac{-\left(2r_o + r_M\left(1+M_j\right)\right)}{-r_M\left(2pq\left(1-M_j\right)+\left(\left(n-s+1\right)c_j - c_S^{\min} - \sum_{k\in(-j)} c_k\right)\right)} \tag{5A.18}$$

$$\frac{\partial M_j^*}{\partial c_k} = \frac{-\left(2r_o + r_M\left(1+M_j\right)\right)}{-r_M\left(2pq\left(1-M_j\right)-\left(\left(n-s+1\right)c_j - c_S^{\min} - \sum_{k\in(-j)} c_k\right)\right)}. \tag{5A.19}$$

Chapter 6: Discussion & conclusions

6.1. Summary of results

This thesis focused on several problems of resource allocation in ocean space. I analyzed these problems on a variety of policy scales going from the local scale, involving a single country, to the regional scale, involving a few countries to the international scale, involving a large number of countries.

Chapter 2 dealt with the spatial planning of offshore wind farms. I presented an optimization model that allocates offshore wind farms in order to maximize profits under ecological constraints. The model results show that spatial planning is an essential instrument to derive an optimal management plan of the Exclusive Economic Zone (EEZ), because the allocation of offshore wind farms is highly dependent on both spatial economic factors such as location costs and ecological restrictions. Only by considering the full ecological effects, and specifying clear policy targets for both offshore wind energy and nature conservation can an optimal spatial plan of the EEZ be formulated.

In Chapter 3 I studied the multiple use nature of assigning Marine Protected Areas (MPAs) in EEZs and the associated incentives. I developed a game theoretic model in which MPAs are considered both as an instrument for fisheries management and as an instrument for nature conservation. The model shows that the multiple use nature of MPAs does not affect the general incentive of countries not to cooperate. It also shows that when the multiple use nature of MPAs is disregarded, i.e. regarding only a single use of MPAs, cooperating on such a single use may produce worse outcomes than not cooperating at all when all uses are accounted for.

Chapter 4 dealt with the question how the distribution of species over ecosystems in ocean space and how different ways of accounting for the contributions of others affects the MPA assignment in ecosystems. I developed a game theoretic model that incorporates the distribution of species over ecosystems into the decisions on MPA size in different ecosystems. Three different regimes were considered: full cooperation, strategic non-cooperation and ignoring the contribution of others. The model shows that unique species occurring in a single ecosystem are better protected than species that occur in multiple ecosystems, even when countries are behaving strategically. Furthermore ignoring contributions by others generally gives higher protection levels than strategic behavior, although MPA size is not as high and the full plan is not as efficient as the fully cooperative solution.

Chapter 5 dealt with the effect of MPAs in the High Seas on the formation of

Regional Fisheries Management Organisations (RFMO). Note that "regional" in this case refers to a region of the High Seas, and that it is not a membership restriction. I adapted the fisheries model from Chapter 3 to a High Sea setting and combined it with a game theoretic model to investigate the stability of RFMO coalitions. I found that in the symmetric case MPAs are implemented but have no effect on stability. In the asymmetric case MPAs stabilize a number of extra coalitions such that more and larger coalitions are stable when an MPA is present compared to the no MPA case. Full cooperation, however, is not reached.

6.2. Brief answers to the research questions

In this section I will briefly discuss the results in light of the research questions.

Question 1: *How can spatial planning of new uses of ocean space improve the ecosystem management in the marine environment?*

Chapter 2 develops and applies a constrained spatial optimization model as a basic first step in spatial planning. In the model profits from the economic activity under consideration (i.e. offshore wind farms) are maximized, subject to spatial and non-spatial constraints. The spatial constraints are other activities, such as shipping or nature conservation. The non-spatial constraints are minimum or maximum targets, such as a total capacity target for offshore wind farms or a minimum number of birds that survive.

The model results show that spatial considerations are very important factors when considering multiple activities, because one activity does not necessarily exclude another activity. In our model for example it is shown that, given the assumptions on the effects of offshore wind farms on bird and fish populations, it is possible to plan offshore wind farms such that bird populations are not harmed seriously and fish stocks increase. This suggest that synergy between marine nature conservation and offshore wind farms is possible. This has also been pointed out by Petersen and Malm (2006) who argue that careful siting of offshore wind farms may benefit the marine environment. Other possible synergies that have been suggested are offshore wind and aquaculture (Buck et al., 2004; Michler-Cieluch et al., 2009), nature conservation through MPAs and tourism (Brown et al., 2000; de Groot and Bush, 2010; Thur, 2010) nature conservation through MPAs and fisheries management (Tundi Agardy, 1994; Gell and Roberts, 2003; Meester et al., 2004) and nature conservation and scientific research (Lindeboom, 1995). These possible synergies can only be realized however, through marine spatial planning.

Spatial optimization models such as the one described in Chapter 2 are useful tools to identify the areas where synergy possibilities exist and where trade-offs between activities have to be made. Furthermore the maps generated by such a model can then form the basis for discussions with and consultation of stakeholders. The views and opinions expressed in such meetings as well as new insights from ecological and economic science can then be incorporated in the model to generate new maps which can then be used as a basis for further discussions, resulting in an iterative process to derive a spatial master plan for an EEZ. Such an approach was used in e.g. California, resulting in a master plan for MPAs (Scholz et al., 2004; Gleason et al., 2010). Marine Spatial Planning does not stop there however. The ocean as well as the economy are dynamic systems, changing constantly. Therefore marine spatial planning has to be adaptive and flexible (Douvere et al., 2007; Douvere, 2008).

In conclusion: Marine Spatial Planning is an instrument that, with the help of spatial optimization models and stakeholder involvement, can identify locations to maximize the effect of synergies and can show locations where trade-offs between activities must be considered. Within Marine Spatial Planning, spatial optimization models can play a positive role in identifying these locations. Consequently Marine Spatial Planning is an instrument that can be used to improve the marine environment, while planning new activities on the seas. For its implementation, however, there is an urgent need for spatial models that combine ecological knowledge with economic information, such as this one. These models require further research into the ecological spatial effects of activities in ocean space, as well as detailed spatial information on costs and benefits.

Question 2: *How does the multiple use nature of Marine Protected Areas (MPAs) affect the incentives of countries to assign such areas?*

As demonstrated in the answer to the previous question there is a large scope for synergies between nature conservation and other economic activities in the seas through the use of MPAs. Consequently when an MPA is assigned it is expected to fulfil multiple goals. In Chapter 3 we look at one of the proposed synergies: fisheries and nature conservation through MPAs.

On the local scale it is relatively easy to account for all benefits that an MPA creates, but at a regional sea scale this is much harder, because multiple countries are involved that both derive benefits from an MPA designated in a single country's EEZ. When dealing with a single use countries may choose to free-ride: a single country bears the costs of the MPA, whereas all countries benefit. In equilibrium countries will only assign an MPA of the size that is beneficial to each of them separately.

Chapter 3 investigates how these incentives change once the multiple use nature of MPAs is accounted for. The full game theoretic model is a combination of two sub-models: a fisheries model where symmetric countries decide simultaneously on their MPA size in their EEZ and their level of fishing effort and a nature conservation model where countries decide simultaneously on their MPA size in their EEZ. In the fisheries model the MPA generates benefits for all countries by increasing the internal growth rate of the shared fish stock, but reducing the catchability of the stock in a single country. In the conservation model the number of protected species within all MPAs generate benefits to all countries but the cost of an MPA in a country's EEZ are borne by that country alone. Both sub-models offer scope for free-riding.

If a single use is considered for an MPA the equilibrium of both sub-models is a non-cooperative outcome, i.e. the MPA size assigned is smaller than under full cooperation. When countries consider both uses of the MPA the final outcome is qualitatively the same, i.e. non-cooperative and smaller than under full cooperation. MPA sizes assigned are a compromise between the non-cooperative outcomes of the different single uses. Because accounting for all uses does not change the game structure per se, it does not induce cooperation.

This outcome does not imply that nothing is gained from accounting for all uses, or that we should focus on a single use and try to reach cooperation on that single use. On the contrary, the outcome of the game shows that accounting for a single activity and cooperating accounting only for that activity may be worse than not cooperating at all, if the benefits from both uses are accounted for. This particular outcome occurs because cooperating on a single issue may induce an MPA size that is so large that its negative effects on one use cancel out all gains from the other use. In our example this holds for the conservation case, but depending on parameter values it could also occur for fisheries, although this seems less probable. Consequently, cooperating on a single use can be worse than not cooperating at all.

In conclusion: the multiple use nature of MPAs does not change the structure of the incentives but does change the outcomes. Accounting for all possible uses when assigning an MPA is important because ignoring one use can undo all positive gains from another. Therefore it is very important that activities in the marine domain such as fisheries and nature conservation are considered simultaneously. This implies that the sectoral approach in policy making should be replaced with a more holistic and integral approach, such as Marine Spatial Planning.

Question 3: *How does perceived uniqueness of species and their distribution over ecosystems owned by different countries affect the MPA assignment of countries?*

The answer to the previous question addressed how incentives to assign MPAs change as a result of their multiple use nature. In Chapter 4 I assumed that all uses have been accounted for and I studied how the distribution of species over ecosystems and the attitudes towards conservation by other countries affect the MPA assignment in a country.

The model devised in this chapter represents a two player game. The game comprises a number of ecosystems, and each country has authority over a number of these ecosystems. Each ecosystem is a stand-alone unit but there is overlap in species between ecosystems. In each ecosystem an MPA can be assigned, protecting the ecosystem and consequently protecting the direct and indirect services the ecosystem provides. The direct services accrue only to the country that owns the ecosystem and are a function of the level of protection of that ecosystem, the indirect services depend on the total number of species protected in all ecosystems and accrue to countries in equal shares. The level of protection, i.e. the size of the MPA, in each ecosystem is investigated under full cooperation, strategic non-cooperation and conservation autarky. Under strategic non-cooperation countries take contributions by others as given, under conservation autarky countries ignore contributions by others.

In general ecosystems that contain many unique species are better protected than ecosystems that contain species occurring in many ecosystems. This holds for all scenarios, although the level of protection is lower under strategic non-cooperation and conservation autarky than under the fully cooperative solution. An interesting feature of conservation autarky is that individual countries consider some ecosystems to contain unique species, and consequently assign larger MPAs there than would have been strictly necessary, because there is overlap with the other country.

How the species distribution over ecosystems affects MPA assignment depends critically on the perceived uniqueness of the species by countries. Taking full cooperation as a benchmark, countries assign too small MPAs if they account for protection of species in other countries, but only value the protection in their own country. In contrast if countries ignore the protection of species by other countries some ecosystems are better protected than under strategic non-cooperation.

In conclusion: Unique species are generally better protected than general species, especially if these species occur transboundary. Although "conservation autarky" is not as good as a fully cooperative solution when trying to maximize global welfare, it is generally better than strategic non-cooperative behavior, especially for species that occur only near the boundary. Also our cooperative protection effort should focus on species that occur transboundary in a broad range as these are the ones that are under the most severe threat.

Question 4: *How does the assignment of MPAs in the High Seas influence the formation of Regional Fisheries Management Organizations (RFMOs)?*

The previous questions all focused on local or regional scale where property rights have been clearly delineated. This question in contrast is focused on the High Sea level, where property rights have not been explicitly defined and that are governed by international law, treaties and custom.

In Chapter 3 we have seen how MPAs can change incentives. The setting of the High Sea is different from the one in this chapter however, because there are no EEZs, and effort restrictions cannot be enforced, i.e. they have to be voluntary.

In Chapter 5 I investigated how MPAs can be assigned on the High Seas, and how they make a difference in the formation of international fishing agreements on the High Seas, i.e. RFMOs. Using the MPA fisheries model from Chapter 3 and combining it with a coalition formation model for RFMOs designed by Pintassilgo et al. (2010) I investigated the coalitions formed with and without MPAs, in otherwise similar circumstances.

The first important finding is that for MPAs to work in the High Seas they have to be acknowledged and respected by all. They are in effect a public good of the weakest-link type. Therefore the size of the MPA will be equal to the smallest optimal size proposed by a country or a coalition of countries. If a larger MPA would be declared, the country that wishes a smaller size can simply create an MPA of the desired size by fishing in parts of the MPA until it reaches the desired size.

Another important finding is the large difference in outcome between fully identical countries and countries that have differing fishing costs. In the model of Pintassilgo et al. (2010) no coalition is stable in the case of identical countries and only small coalitions are stable when countries differ in fishing costs. I found that with MPAs in the symmetric case still no coalition is stable, and the MPA size is also always the same; the globally optimal size. In the asymmetric case with MPAs, more and larger coalitions are stable, although full cooperation is not necessarily reached. Furthermore the MPA size is smaller than the globally optimal, unless full cooperation is reached.

In asymmetric settings coalitions have a cost advantage over free riders and this advantage is increased by MPAs. This cost advantage is the driving force behind the stability, and hence MPAs generally increase stability, if countries are not too symmetric. In the symmetric case no cost advantage exists and consequently MPAs have no influence, but because everyone gains and suffers from the MPA in equal shares the optimal MPA size is still reached, even though the potential full benefits are not reaped.

In conclusion: the assignment of MPAs in the High Seas generally improves the stability of RFMOs, if countries are asymmetric. They have no effect on the incentives if countries are symmetric but they do increase profits and stock in that case.

Three important conclusions can be drawn from this thesis as a whole. First

Marine Spatial Planning and Marine Protected Areas can contribute in a positive way to the management of human activities in ocean space. Second, neither of them is a silver bullet. Both need careful implementation, where all uses are accounted for, and especially the public good aspects of MPAs needs to be addressed. Third the success of MPAs (and as such of Marine Spatial Planning) is not only highly dependent on the incentives and social norms but also on the implementation scale. As shown in the introductory chapter there are a number of governance levels in ocean space and I have shown in the subsequent chapters how these scale levels result in different incentives for MPA assignment.

6.3. Conclusions on methodology

In this section I will address the appropriateness of the methodology in the different chapters and outline their strengths and weaknesses.

In Chapter 2 a spatial optimization model was formulated. The strength of the model lies in the fact that through the spatial explicitness possible synergies and trade-offs can be examined that could not have been identified otherwise. The spatial specification of revenues and costs of offshore wind farms, as well as the combination with some of the potential spatial ecological effects allows explorations of effects and possibilities that are otherwise ignored. Even though some of the ecological effects are severely simplified, the potential of the model is clearly illustrated. The weaker points of the model are the linear parameterization of the ecological model and the fact that the model is not dynamic, whereas the ocean system is inherently dynamic. In order to be of good use in practical policy making and to form a basis for ecosystem based management, it would be useful to be able to quantify the effect of offshore wind farms on the environment more precisely. An ecological model is needed, that takes into account the major effects of offshore wind farms, and includes a time dependent specification.

Another important point, as has been pointed out under the research questions, is that such a model can only be one of the first steps in a Marine Spatial Planning process. Although a number of other economic uses have been included in the model some of the possible uses may only be identified after discussions with stakeholders. Furthermore some of the effects of offshore wind farms that are not taken into account in the model may only become clear after discussions, such as effort displacement of fishermen. The maps resulting from different scenarios should therefore not be taken as an endpoint but as the starting point for a discussion.

In the other chapters I analyzed strategic aspects of the assignment of MPAs, sometimes spatial, sometimes non-spatial. The economist’s tool of choice to analyze strategic situations is game theory. Game theory is a mathematical method to analyze strategic interactions among agents. Agents choose a strategy from their possible set of strategies, such that they maximize their payoffs given the behavior of other agents. Because MPAs constitute at least partly public goods and fish stocks constitute a common pool resource strategic interaction between countries is likely to occur. Therefore game theory is an appropriate method to analyze such situations.

In Chapter 3 I formulated a model where MPAs are an instrument for fisheries management as well as a model where MPAs are an instrument for nature conservation. Although both the strategic effects of MPAs have been studied (e.g. Sumaila, 2002 and Ruijs and Janmaat, 2007) as well as the economic consequences of their multiple uses (e.g. Brown et al., 2000 and Boncoeur et al., 2002) their combination has not been considered so far.

The fisheries MPA model of Chapter 3 (and Chapter 5) is highly stylized, and relies on simplifying assumptions such as a single stock that is uniformly distributed, and no interaction effects between the fish stock and other stocks. The assumption of uniform distribution is unusual in MPA models, because in the absence of other effects an MPA would just make fishing more inefficient. However, as we have argued in Chapters 3 and 5, following Armstrong (2007), Armstrong and Falk-Petersen (2008), Sanchirico (2004) and Schnier (2005a,b), MPAs may have a habitat effect, increasing the growth rate of the fish stock, and this can be captured by increasing the growth rate in the MPA. The spill-over between MPA and fishable area depends on movement parameters. In our model movement is not explicitly modeled, but low movement out of the MPA could be captured by adjusting the growth rate bonus downward. The exact ecological and spill-over effects of MPAs urgently need further research, especially in the temperate regions, because the justification for MPAs for fisheries management lies in these effects.

The MPA as a nature conservation tool in Chapter 3 shows the importance of considering the transboundary aspects of the species in MPAs. Busch (2008) has shown that the gains of transboundary parks depend upon the exact goals and how they are measured. In the model in Chapter 3 there are diminishing returns in number of protected species per extra unit of area, and protection in one country is a substitute for protection in another country. Therefore countries have an incentive to free-ride on the contribution of others, but this is not necessarily true in absence of these assumptions. I explored this subject further in Chapter 4.

Symmetry is an important assumption in Chapter 3 as well. Chapters 4 and 5 clearly show the important differences that occur through the introduction of asymmetry. In general the introduction of asymmetry mitigates the effects of non-cooperation to some extent, because it introduces greater cooperation and better protection, although not up to the fully cooperative level. We can therefore think of symmetry as a worst case scenario.

The multiple use nature of MPAs applies equally to transboundary parks and ecosystems in a terrestrial setting; they provide multiple services (e.g. Munthali, 2007 and López-Hoffman et al., 2010). Accounting for all these services is important and transboundary cooperation on just one service may be detrimental to another.

Chapter 4 can be thought of as an extension of the conservation model in Chapter 3. In this chapter I explored the subject of substitution and transboundary species further and on a higher ecological scale. The fact that species do not occur in all places necessitates a spatial model, and this is exactly what is formulated in Chapter 4. To reduce the complexity the model in Chapter 4 is one dimensional; it can be interpreted as a coast line or transect. The extension to a two dimensional model is straightforward, but adds little value to the basic analysis.

The analysis of transboundary protection and especially the issue of substitution between protected areas or, more generally, nature conservation by different countries is still in its infancy. A number of authors explore the current efficiency of transboundary protection in a terrestrial context (e.g. Kark et al., 2009 and Jantke and Schneider, 2010) but to my knowledge no such attempt has been made in the marine domain. Moreover these authors consider efficiency, i.e. they compare a situation of full cooperation with what I have termed "conservation autarky", but strategic non-cooperation is not explored. This is where Chapter 4 contributes: it formulates a framework in which both strategic non-cooperation and conservation autarky can be explored, in a spatially explicit model.

Game theory requires an approach where benefits and costs are explicitly accounted for, because of the difference between the shared benefits and the individual costs. The usual approach for these kind of protected area selection problems is a cost minimization approach, or a maximization under a budget constraint (Williams et al., 2005). This is the very first paper applying game theory to this subject, and therefore our benefits approach is necessarily very stylized. In later papers the description of the benefits and costs as well as the exact calibration of the ecological model can be more precise, but for now the model gives at least some intuition of the problems at hand in transboundary conservation.

In Chapter 5 I used game theoretic concepts from the literature on international environmental agreements and the fisheries MPA model from Chapter 3 to provide insights in how MPAs affect the formation and stability of RFMOs. The decision on the size of the MPA is modeled as a weakest link game as described by Hirshleifer (1983) and Sandler (1998). The assumption that a country can destroy the habitat effect in a part of the MPA by starting to fish in that area, may seem a rather extreme, but is justified if the fisheries activity comprises e.g. frequent trawling in vulnerable benthos.

The game is in essence a broadening of the model formulated by Pintassilgo et al. (2010) and looks at the formation of RFMOs in a broader context, accounting for more management tools, i.e. MPAs, in High Sea fisheries. These tools generally broaden the scope for cooperation.

6.4. Policy recommendations

Looking at the previous conclusions we get the following recommendations for policy makers in order to manage our marine resources optimally:
Marine Spatial Planning is a step forward, that needs to be implemented. To be successful, however, it needs to be holistic, and run across sectors and stakeholders. Modeling tools such as the one described in Chapter 2 form an important basis, but are just the beginning, not the end point. Also, we need to be clear on objectives and (expected) effects, and involve stakeholders from the beginning of the process. Only in this way can we achieve an ecosystem-based management plan for our EEZ.

As part of the Marine Spatial Planning process, and in the European context within the Marine Strategies that countries are in the process of formulating, MPAs should be included. MPAs offer a broad range of possible advantages, but their potential can only be fully realized if all uses of the MPA are accounted for. At the local level again, the involvement of stakeholders is crucial to identify the possible costs and benefits of MPAs. As shown in Chapter 3, at the more regional level the economic incentives for under-provision of MPAs are strong because most of their beneficial effects are public goods. Although the Marine Strategy Directive encourages countries to cooperate it cannot enforce cooperation. Therefore in the EU the Maritime Policy should be strengthened to enforce these MPA assignments. Alternatively, it can be structured such that the incentives associated with the assignment of MPAs are realigned. Possible examples include increasing the fishing quota of countries that assign shared MPAs for fisheries, or the possibility to account for shared MPAs for countries that have to meet conservation objectives.

When MPAs have conservation as a main goal, the location and size of MPAs is strongly influenced by the distribution of species or other ecological features, as shown in Chapter 4. In the absence of enforcement or a realignment of incentives countries will generally free-ride on the transboundary species. Therefore, exactly these kind of species that should receive the most attention when bargaining about conservation efforts among countries. Species that are unique to a country will generally receive more protection, and are therefore less prone to international bargaining efforts on conservation, given of course that countries have enough funds to protect the species in the first place.

Chapter 5 shows the importance of MPA presence in the High Seas. Their presence may make bargaining over effort and reaching stable effort coalitions easier. Because MPAs have to be agreed upon unanimously they are a weakest link good. The scope for reaching agreements on large MPAs in the High Seas is therefore small, and international efforts should focus on protecting the most vulnerable areas first, especially the areas that are nursery areas. By protecting these, their habitat function is restored and this may in turn improve the fish stock, which in turn may give scope for larger agreements.

6.5. Recommendations for further research

Marine Spatial Planning can make a positive contribution to protection of the marine environment. Our model in Chapter 2 shows the potential of models as a tool for scenario analysis and as a basis for discussions with stakeholders. To really use such a model in practice however, the ecological relationships between offshore wind farms and the environment need to be further clarified.

A lot of the research done here has implications for terrestrial research, especially the research on transboundary parks and ecosystem services. Including ecosystem services and environmental effects of economic activities in terrestrial and marine spatial planning is a step forward that has been implemented by some but not by all.

The use of game theory in (terrestrial) allocation of reserves is a relative new and unexplored area of research. Most of the research done on transboundary parks is terrestrial and uses reserve site selection methods. Incorporation of game theory directly in these methods is difficult because these methods generally use cost effectiveness or optimization under a budget constraint. For game theory to be used in these areas we need to reformulate these problems to benefits approaches or devise a way of incorporating game theory in these algorithms.

The conservation models in both Chapter 3 and Chapter 4 are symmetric in their benefits function, i.e. in both cases the benefits of conservation accrue to all countries in equal shares. An interesting case to further explore is to relax this assumption, as well as a more unequal endowment of biodiversity. Similar work but on a higher scale level has been done by Barrett (1994) who explores the incentives for biodiversity conservation in the poor regions of the world paid for by the rich part and the associated incentives in a supergame. He finds that the developed countries of the world have an incentive to contribute too little to conservation of this kind, and that a fully cooperative solution that is renegotiation proof offers little extra benefits.

The notion that MPAs are a weakest link good in absence of enforcement is also interesting from a terrestrial perspective, especially in developing countries where enforcement of conservation areas is often difficult. Applying a similar framework on a lower level, i.e. that of stakeholders may be an interesting extension. In that case it would also be important to account for all the costs and benefits from protected areas, especially to the stakeholders. By accounting for all possible opportunity costs and benefits of the stakeholders one can try to find the protected area size that is acceptable to all. If opportunity costs are very high or benefits do not accrue to stakeholders this minimum size may be absent.

Finally the ecological effects of human activities on ocean space and their interactions are an area of urgent research need. With offshore wind farms we generally know their direct impacts, but their long term effects and spatial spill-over effects are generally unknown. The same holds for MPAs; we generally know what happens inside, but their effects on the surrounding areas and the human response are less well known, whereas my work and that of others has shown the importance of human behaviour on (spatial) spill-over effects, especially since the spill-over effects are generally public goods.

6.6. (Sea)Food for thought

Ocean space is still as limitless as in the days of Hugo de Groot, considering how little we know about the ocean floor and the full ocean system. Even with our limited knowledge however, it is clear that we are rapidly approaching some of its boundaries. Rockström et al. (2009) have identified the "Planetary Boundaries", nine interrelated boundaries on important geo-physical processes and current trends, that keep our planet stable and habitable. Ocean space is an integral and important part of these boundaries. For some of these boundaries it functions as a supplier (e.g. the fresh water cycle and biodiversity), for some it functions as a sink (e.g. ocean acidification and chemical pollution) and for some it functions as a stabilizing element (e.g. climatic change). As we are finding out more about these boundaries, we are also discovering the vital role the ocean system plays in these processes.

To make sure that we do not cross these boundaries and to make sure that both the current and future generation can continue to enjoy the goods and services provided by the ocean we have to manage our activities on land as well as on ocean space. Management is needed on all scale levels and needs to be integrated across sectors and space, incorporating both ecology and socio-economic aspects. This thesis has shown how Marine Spatial Planning and its tools can play a positive role in achieving such management.

At the European level the first steps towards such an integrated approach have been taken with the formulation of the Maritime Policy, and its subsidiaries the Marine Strategy Directive and the Common Fisheries Policy. If we can manage to implement this policy, whilst avoiding the pitfalls of free-riding, as well as involve the stakeholders to design Marine Strategies at the country level, the foundations for good management for now and the future have been laid.

In this way, with well-coordinated management of our human activities in ocean space, we may be able to turn the negative trends into positive ones such that we and future generations can continue to benefit from our planet that we inappropriately called Earth, when it is quite clearly “Ocean” (Arthur C. Clarke).

Summary

Ocean space is in a crisis. Globally, many fish stocks are over-exploited or in decline, pollution increases, and there is an increasing competition among the growing number of users and activities in ocean space.

Yet, ocean space forms the basis for a large number of ecosystem goods such as food and provides essential ecosystem services such as climate regulation, that form the foundation for our human economy. If we want to be able to sustainably harvest the goods from the system and keep the foundations of our economy stable we will have to manage our activities in ocean space in a radically different way.

A new way of management that offers a new holistic perspective for ecosystem based management of our ocean space is Marine Spatial Planning (MSP). Although far from a panacea, Marine Spatial Planning, with Marine Protected Areas as one of its tools is a dynamic approach that can identify possible synergies and plan economic activities in ocean space in such a way that they do not cause irreversible damage to the system.

Many open questions remain in the area of optimal management of marine resources through Marine Spatial Planning and Marine Protected Areas (MPAs). First of all the suitability of MSP and MPAs depends on the activity being managed and the state of the ocean system. Second the suitability of the instruments is scale dependent, because different sets of rules and regulations apply on different scale levels of ocean space management.

This thesis focuses on three scale levels, based on the regulatory framework. These levels are: a single country's Exclusive Economic Zone (EEZ), a territorial sea fully claimed by Exclusive Economic Zones and the High Seas. At the first level activities are governed by the laws of a single country. At the second level the activities within the EEZ are governed by the local laws, but countries cannot force each other to cooperate or exhibit certain activities. At the third level activities of countries are governed by international law only.

In this thesis I investigate the use of Marine Spatial Planning of offshore wind farms on the level of a single country's Exclusive Economic Zone, and I investigate the suitability of MPAs as instruments for both fisheries management and conservation on the scale level involving several Exclusive Economic Zones as well as the High Sea level.

At the first level I study the spatial planning of offshore wind farms with an optimization model that allocates offshore wind farms under ecological constraints. The model results show that space is an essential element to derive an optimal management plan of the EEZ, because the allocation of offshore wind farms is highly dependent on both spatial economic factors such

as location costs and ecological restrictions. The results show that Marine Spatial Planning is necessary, because only in this way can possible synergies between e.g. offshore wind farms and environmental protection be identified and eventually realized. The model can assist with the first steps in Marine Spatial Planning of offshore wind farms; its results can be used as a basis for conversation and consultation with stakeholders.

I then move on to the second level to study MPAs as a tool for conservation and fisheries management. In a game theoretic model I investigate the multiple use nature of assigning MPAs in EEZs. MPAs are considered as an instrument for both fisheries management and nature conservation. The model results shows that the multiple use nature of MPAs does not alter the general free-riding incentives that are also present when it is used as solely as a fisheries management or nature conservation tool. The magnitude of the incentives and the general equilibrium MPA size do change. The free-riding MPA size becomes a compromise between the fisheries free-riding solution and the conservation free-riding solution. An interesting side effect of the multiple use nature of MPAs is, that if only a single use of MPAs is considered, cooperating on that single use may produce worse outcomes than not cooperating when accounting for all uses. Thus the importance of considering all possible uses and stakeholders of an MPA, and the need to realign incentives in MPA assignment on the regional scale is shown.

At the same scale level I study how species distributions and different ways of accounting for the contributions of others affects MPA assignment as a tool for biodiversity conservation. With a spatial game theoretic model I investigate three different conservation regimes: full cooperation, strategic non-cooperation, and conservation autarky. Under strategic non-cooperation countries anticipate protection by the other, under conservation autarky they ignore these contributions. The main results show that unique species occurring in a single ecosystem are relatively well protected, even when countries are free-riding. Species that occur in multiple ecosystems on both sides of the border in contrast are under non-cooperation under-protected, compared to full cooperation. This is in part caused by location leakage, i.e. protecting a number of species less because they are protected by others. On the one hand conservation autarky eliminates location leakage and generates larger MPAs at the border. On the other hand these MPA sizes are often too high from a global perspective. From this we can conclude that international conservation efforts should mainly focus on transboundary occurring species. Also, although conservation autarky is not a first-best solution, if it occurs, e.g. through social norms, it is certainly better than strategic non-cooperation.

I then move on to the third level: international waters or the High Seas. I study the effect of the assignment of internationally recognized MPAs in the High Seas on the formation of Regional Fisheries Management Organisations

(RFMO) with a game theoretic model.

MPAs are assigned through a weakest-link game: because everyone has to agree on an MPA before it actually can be protected, it can only be as large as the strongest opposing player wants it to be. The full coalition formation of the RFMO follows a three stage game: in the first stage countries sign up to the RFMO, in the second stage both RFMO members and non-members agree on an MPA size through a weakest-link game and in the last stage RFMO members and non-members choose their fishing effort. The game is solved with backward induction.

I find that if countries have equal costs and benefits MPAs of optimal size are implemented but these have no effect on stability of RFMOs; the only stable coalition is the coalition where everyone acts alone. In the case where countries face different fishing costs, MPAs stabilize a number of extra coalitions such that more and larger coalitions are stable when an MPA is present compared to the no MPA case. Full cooperation, however, is not necessarily reached. A general conclusion is therefore that the assignment of MPAs in the High Seas can not only improve the fisheries through direct effects such as insurance and possible increases in catches, but also indirect by contributing in a positive way to the formation of RFMOs.

Three important conclusions can be drawn from this thesis as a whole. First Marine Spatial Planning and Marine Protected Areas can contribute in a positive way to the management of human activities in ocean space. Second, neither of them is a silver bullet. Both need careful implementation, where all uses are accounted for, and especially the public good aspects of MPAs needs to be addressed. Third the success of MPAs (and as such of Marine Spatial Planning) is not only highly dependent on the incentives and social norms but also on the implementation scale. As shown in the introductory chapter there are a number of governance levels in ocean space and I have shown in the subsequent chapters how these scale levels result in different incentives for both MPA assignment and Marine Spatial Planning.

Samenvatting

De oceanische ruimte verkeert in een crisis. Wereldwijd is sprake van overbevissing, vele visbestanden nemen af, vervuiling neemt toe, en er is steeds meer concurrentie tussen het groeiende aantal gebruikers en activiteiten in de oceanische ruimte.

Dat terwijl de oceanische ruimte de basis vormt voor een groot aantal ecosysteemgoederen en ecosysteemdiensten zoals voedsel en de regulering van het klimaat, die de basis vormen voor onze economie. Als we de goederen uit de oceanische ruimte duurzaam willen kunnen blijven oogsten en de fundering van onze economie stabiel willen houden zullen we onze activiteiten in de oceanische ruimte op een radicaal andere manier moeten beheren. Het roer moet om.

Ruimtelijke planning op zee is een nieuwe manier van management met een holistische kijk waarbij het ecosysteem de basis vormt voor het beheer. Hoewel zeker geen wondermiddel, is ruimtelijke planning op zee, met zeereservaten als een van de gereedschappen, een dynamische benadering waarmee synergieën gecreëerd kunnen worden en waarmee economische activiteiten zodanig gepland kunnen worden dat ze geen onomkeerbare schade aan het systeem toebrengen.

Er zijn nog veel openstaande vragen op het gebied van optimaal beheer van hulpbronnen op zee, door middel van ruimtelijke planning op zee en zeereservaten. Allereerst hangt de geschiktheid van ruimtelijke planning op zee en zeereservaten als beleidsinstrument af van de activiteit die we willen beheren en van de status van het oceanisch systeem. Ten tweede hangt de geschiktheid van deze instrumenten af van het schaalniveau, omdat verschillende sets van regels en wetten van toepassing zijn op verschillende schaalniveaus in de oceanische ruimte.

Dit proefschrift concentreert zich op drie schaalniveaus, die gebaseerd zijn op het kader van wetten en regels. Deze drie niveaus zijn: de Exclusieve Economische Zone (EEZ) van één land, een territoriale zee die volledig geclaimd wordt door meerdere Exclusieve Economische Zones en de internationale wateren. Op het eerste niveau worden activiteiten gestuurd door de wetten van één land. Op het tweede niveau worden de activiteiten binnen de EEZ gestuurd door lokale wetten van verschillende landen, maar landen kunnen elkaar niet dwingen om samen te werken of bepaalde activiteiten op zee uit te voeren. Op het derde niveau worden de activiteiten van landen uitsluitend gestuurd door het internationaal recht.

In dit proefschrift onderzoek ik het gebruik van ruimtelijke planning voor het plannen van windmolenparken op zee binnen de EEZ van één land (eerste

niveau). Verder onderzoek ik de geschiktheid van zeereservaten als beleidsinstrumenten voor zowel de visserij als natuurbehoud. Dit laatste onderzoek vindt plaats op het schaalniveau waarbij verschillende EEZs betrokken zijn (tweede niveau) en op het schaalniveau van de internationale wateren (derde niveau).

Op het eerste niveau onderzoek ik de ruimtelijke planning van windmolens op zee met een optimaliseringsmodel dat windparken op zee toewijst, rekening houdend met ecologische beperkingen. Uit de resultaten blijkt dat ruimte een essentieel onderdeel is voor een optimaal beheerplan van de EEZ, omdat de toewijzing van windparken erg afhankelijk is van ruimtelijke economische factoren zoals de kosten op locatie en ecologische beperkingen. De resultaten laten ook zien dat ruimtelijke planning op zee noodzakelijk is, omdat alleen op deze manier mogelijke win-win situaties tussen bijvoorbeeld windparken op zee en milieubescherming geïdentificeerd en gerealiseerd kunnen worden. Het model kan een hulpmiddel zijn voor de eerste stappen bij ruimtelijke planning op zee; de resultaten kunnen als basis dienen voor verdere consultatie en communicatie met belanghebbenden.

Op het volgende niveau onderzoek ik het gebruik van zeereservaten als beleidsinstrument voor natuurbehoud en visserijbeleid. Met een speltheoretisch model onderzoek ik het effect van het multifunctionele karakter van zeereservaten op het aanwijzen van deze reservaten in EEZs. Reservaten worden in het model beschouwd als een instrument voor zowel visserijbeleid als natuurbehoud.

De modelresultaten laten zien dat het multifunctionele karakter van zeereservaten geen invloed heeft op de algemene neiging van landen om mee te liften op elkaars bijdrage, die ook aanwezig is als zeereservaten worden ingezet als beleidsinstrument voor uitsluitend de visserij of natuurbehoud. In het evenwicht van ieder afzonderlijk spel, en van het gecombineerde spel wordt dus niet samengewerkt. De sterkte van deze neiging en de grootte van het zeereservaat in het evenwicht veranderen wel. De grootte van het zeereservaat in het evenwicht wordt een compromis tussen het meelift evenwicht van de visserij en het meelift evenwicht van natuurbehoud.

Een interessante bijwerking van het multifunctionele karakter van zeereservaten is dat, als landen samenwerken bij het aanleggen van het zeereservaat, maar daarbij slechts op één gebruiksvorm van het zeereservaat letten, deze samenwerking in totaal een slechtere uitkomst kan geven dan wanneer ze niet samenwerken en wel op alle gebruiksvormen letten bij hun individuele keuzes.

Hiermee is aangetoond hoe belangrijk het is om alle mogelijke gebruiksvormen van, en belanghebbenden bij, zeereservaten in acht te nemen. Verder heb ik laten zien hoe belangrijk het is dat de economische prikkels bij het aanwijzen van zeereservaten herschikt worden.

Op hetzelfde schaalniveau onderzoek ik hoe het aanwijzen van zeereservaten voor biodiversiteitsbescherming afhangt van de manieren waarop landen rekening houden met elkaars besluiten en van de verdeling van plant- en diersoorten. Met een ruimtelijk speltheoretisch model onderzoek ik drie verschillende beschermingsregimes: volledig samenwerken, strategisch gedrag en beschermingsautarkie. Bij strategisch gedrag anticiperen landen op bescherming door de ander, bij beschermingsautarkie negeren ze bescherming door de ander.

De resultaten laten zien dat unieke soorten die in één ecosysteem voorkomen relatief goed beschermd worden, zelfs als landen meeliften op de inspanningen van de anderen. Soorten die in meerdere ecosystemen aan beide kanten van de grens voorkomen worden, in vergelijking met volledige samenwerking, daarentegen te weinig beschermd als landen op elkaars inspanningen meeliften. Dit wordt gedeeltelijk veroorzaakt door locatielekkage: dat wil zeggen dat het ene land soorten minder beschermt, omdat andere landen ze al beschermen.

Aan de ene kant zorgt beschermingsautarkie ervoor dat locatielekkage vervalt en reservaten aan de grens groter zijn. Aan de andere kant zijn deze reservaten uit mondiaal perspectief vaak groter dan nodig.

Hieruit kunnen we concluderen dat de inspanningen voor internationale bescherming zich vooral zouden moeten richten op soorten die grensoverschrijdend voorkomen. Verder kunnen we concluderen dat, hoewel beschermingsautarkie niet de allerbeste oplossing is, indien het voorkomt, bijvoorbeeld als gevolg van morele verplichtingen, het in ieder geval beter is dan strategisch gedrag.

Vervolgens ga ik door met het derde schaalniveau namelijk dat van de internationale wateren. Ik onderzoek het effect van internationaal erkende zeereservaten in internationale wateren op de formering van regionale visserijbeheer organisaties (RVOs) met een speltheoretisch model.

Zeereservaten worden aangewezen door een spel met de zwakste schakel: omdat iedereen het eens moet zijn over een zeereservaat voordat het beschermd kan worden, kan het slechts zo groot zijn als de sterkste tegenstander van het reservaat wil dat het is. De volledige formering van een coalitie die een RVO vormt verloopt via een spel in drie stadia: in het eerste stadium worden landen lid van de RVO, in het tweede stadium onderhandelen de RVO leden en niet-leden over de grootte van het zeereservaat door middel van een spel met de zwakste schakel en in het derde en laatste stadium kiezen leden en niet-leden hoeveel ze willen vissen.

Ik ontdek dat als landen gelijke kosten en baten hebben voor wat betreft visserij, ze een zeereservaat van optimale grootte aanwijzen, maar dat dit zeereservaat geen invloed heeft op de stabiliteit van de RVO; de enige stabiele coalitie is die waar ieder voor zichzelf gaat, dat wil zeggen er is geen coalitie. Als landen verschillende kosten hebben voor visserij en we vergelijken een situatie waarin landen niet onderhandelen over een zeereservaat met een situatie waarin ze dat wel doen, dan zien we dat zeereservaten ervoor zorgen dat een aantal extra coalities stabiel zijn. Het is echter niet noodzakelijkerwijs zo, dat volledige samenwerking bereikt wordt. Een algemene conclusie is dus dat het aanwijzen van zeereservaten in internationale wateren niet alleen de visserij kunnen verbeteren door directe effecten zoals een verzekeringsfunctie en het verbeteren van de vangsten, maar ook indirect door bij te dragen aan de formering van RVOs.

Drie belangrijke conclusies kunnen getrokken worden als we dit hele proefschrift in ogenschouw nemen. Allereerst dat zowel ruimtelijke planning op zee als zeereservaten op een positieve manier kunnen bijdragen aan het sturen van de menselijke activiteit in de oceanische ruimte. Ten tweede dat geen van beide een wondermiddel is. Allebei moeten ze zorgvuldig toegepast worden, rekening houdend met alle gebruiksvormen. Hierbij moet vooral gelet worden op de aspecten van zeereservaten die publieke goederen vertegenwoordigen. Ten derde hangt het succes van zeereservaten (en dus ook van ruimtelijke planning op zee) niet alleen sterk af van de prikkels en sociale normen maar ook van de schaal waarop ze worden toegepast. Zoals ik laat zien in de inleiding zijn er een aantal sturingsniveaus in de oceanische ruimte en in de daaropvolgende hoofdstukken toon ik aan dat deze schaalniveaus resulteren in verschillende prikkels voor zowel zeereservaten als ruimtelijke planning op zee.

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Training and supervision plan

Sense PhD Courses

- Environmental research in context
- Research context activity: Report on participation in Economics of Marine Resources Course (24-28 January 2011 in Esbjerg, Denmark): The fisherman and the conservationist, two (not so) strange bed partners.
- Introduction to the marine sciences

Other PhD and MSc courses

- Theories and models in environmental economics
- Econometrics
- Advanced econometrics
- Advanced microeconomics
- Advanced macroeconomics
- Behavioural economics
- Environmental and resource economics
- Game theory
- Economics of marine resources
- Techniques for writing and presenting scientific papers

Oral presentations

- NAKE research day, 2008, Utrecht, The Netherlands
- Workshop on game theoretic applications in fisheries, 2009, Esbjerg, Denmark
- EAERE, 2009, Amsterdam, The Netherlands
- California and the world ocean, 2010, San Francisco, USA
- Bioecon, 2010, Venice, Italy

The research described in this thesis was financially supported by the MarBEF network of excellence 'Marine Biodiversity and Ecosystem Functioning' which is funded by the Sustainable Development, Global Change and Ecosystems Programme of the European Community's Sixth Framework Programme (contract no. GOCE-CT-2003-505446).