Evaluation of spatial management strategies in the German Bight: How to balance sustainable use and ecosystem health?

Dissertation

with the aim of achieving a doctoral degree at the
Faculty of Mathematics, Informatics and Natural Sciences

Department of Biology of University of Hamburg

submitted by

Antje Gimpel

2015 in Hamburg

The following evaluators recommend the admission of the dissertation:

Prof. Dr. Axel Temming

Dr. Vanessa Stelzenmüller

Day of the disputation: 02. October 2015

Table of contents

Summary

Facing the ongoing depletion of finite resources and the destruction of natural habitats, which is accompanied with a loss of biodiversity, the concept of Ecosystem-Based Management (EBM) attracts attention. EBM considers the whole ecosystem by taking into account humans. Its overarching goal is to maintain the ecosystem in a healthy, productive and resilient state, assuring the use of the ecosystem services it provides for current and future generations.

In the course of the Integrated Maritime Policy (IMP) and the Europe 2020 strategy tall orders are placed with the European countries. The member states are faced with multiple objectives such as Good Environmental Status (GES) or Blue Growth. Consequently, Marine Spatial Planning (MSP) was identified as the cross-cutting policy tool when applying an ecosystem-based approach to the management of human activities. Given the lack of indicators and targets that could describe the achievements towards multi-objective planning, the effectiveness of a management approach cannot be assessed. Consequently, the most efficient management strategy needs to be identified to provide guidance towards Ecosystem-Based Marine Spatial Planning (EB-MSP).

The overall aim of this thesis was to develop and test concrete, place-based tools which allow a transparent evaluation of spatial management options and their consequences for the German Bight. Hereby the attention is directed to the fishery sector, demersal fish populations and the benthic ecosystem in particular.

In **Manuscript 1** of this dissertation the ecological and economic consequences of current and future management strategies were assessed. The current management strategies involved the daily risks and conflicts occurring in the German Exclusive Economic Zone (EEZ) of the North Sea. The future management strategies integrated the realisation of the offshore wind farm development. As a first step, a spatially explicit conflict analysis was applied. The

corresponding levels of conflict have been measured based on the distribution of important individual human activities. The application gave a first overview about the conflicting uses and enabled the user to carry out a multi-sector approach. Further, an increased conflict potential at the expense of the German fishery sector could already be predicted.

As a second step, the risks for important ecosystem components such as the nursery grounds of plaice *Pleuronectes platessa*, a fish species of high commercial value, were qualitatively assessed. The approach facilitated the visualisation of cause-effect pathways of human impacts on the nursery grounds. Hereby, tendentious effects were identified: the German Bight is facing increased pressures as a result of offshore wind farms, while the pressures excerted through demersal fisheries will decrease.

In **Manuscript 2** a synergy analysis was performed to index suitable co-location sites for the coupling of offshore aquaculture farms and wind farms in the German EEZ of the North Sea. The aim was to compute ecological and, as a consequence thereof, economic benefits of future management strategies. The main advantage hereby was the exclusion of high-risk areas not being suitable for aquaculture species and the integration of a suitability-scale, which facilitates the choice of co-location sites. Consequently, multiple sites suitable for co-locations in the German Bight were identified.

In **Manuscript 3** of this dissertation a quantitative risk analysis was conducted. The aim was to calculate the ecological risks of current and future management strategies. The current management strategies involved the 'business as usual' pressure-state relationships that are occurring in the German EEZ of the North Sea. The future management strategies integrated the realisation of the offshore wind farms which causes a spatial shift of 15 % of the total fishing frequency of large beam trawlers and 3 % of the small beam trawlers. The risks have been measured using the 'current and future state of benthic communities'. The applied method allowed assessing the likelihood of the occurrence of a benthic disturbance. Accordingly, the German Bight is facing the risk of an increased disturbance in 8 % of the remaining areas open for fishing, and consequently an increasing impact on benthic communities.

In **Manuscript 4** a spatially explicit trade-off assessment of ecosystem services (ES) was applied using the German EEZ of the North Sea as a case study. The aim was to calculate the future environmental and economic consequences obtained from alternative management strategies and their effects on the ES values. The methods applied facilitated the integration of multiple data sets originating from different uses. Therefore, trade-offs were depicted in a transparent manner and could be directly linked to the decision-making process related to the German Bight. The scenarios resulted in environmental risks on the one hand, such as a decrease in supporting services (e.g. habitats) and in returns such as increasing provisioning services (e.g. food from fisheries) on the other hand.

In **Manuscript 5** of this dissertation a stakeholder preference analysis was conducted to empirically record priorities for socio-cultural future management strategies on the basis of six case studies (including the German part of the North Sea). The strategies were derived from different management objectives at different management levels. This approach allowed an empirical analysis of the priorities ascribed to management objectives, while facilitating interactions between the different stakeholders. The results included e.g. consensus about the future objectives to "reduce benthic disturbance" and "enhance friendly energy". However, weak consensual preferences were recorded for the objectives "competitiveness of aquaculture", "competitiveness of fisheries", "preservation of target stocks/GES" or "ensure high resource rent".

Ultimately, the risks and benefits generated from alternative management objectives derived for the German Bight were assessed. Information about the (spatial) extent of management effects is a fundamental requirement for decision makers. Further, the tools that are useful to estimate the most efficient management strategies with regard to EB-MSP were identified.

Through the implementation of such methods synergistic or conflicting effects can be shown in advance, future risks can be identified, trade-offs can be eased and thus the communication between stakeholders, planners and decision makers can be facilitated. Nevertheless, the majority of the performed analyses require a coherent knowledge of marine systems and underlying ecosystem processes. The analytical methods applied were highly multidisciplinary, and data processing for the tools identified had resulted in a high degree of complexity. Accordingly, the scientific underpinning is still inevitable in order to evaluate management strategies and consequently to be able to offer guidance to EB-MSP towards a sustainable use of the German Bight as a healthy ecosystem.

Zusammenfassung

Angesichts einer fortlaufenden Erschöpfung endlicher Ressourcen und der Zerstörung natürlicher Lebensräume, welche einhergeht mit einem Verlust an Biodiversität, gewinnt das Konzept des Ökosystembasierten Managements (EBM) zunehmend an Bedeutung. EBM betrachtet das Ökosystem unter Einbezug des Menschen im Gesamten. Oberstes Ziel ist die Erhaltung des Ökosystems in einem gesunden, produktiven und robusten Zustand, um die biologische Vielfalt zum Nutzen heutiger und künftiger Generationen zu erhalten. Im Zuge der Integrierten Maritimen Richtlinien (IMP) und der europäischen 2020-Strategie werden den europäischen Ländern umfangreiche Aufgaben zugeteilt. Managementziele sollen u.a. das Erreichen eines guten Umweltzustandes (GES) sowie ein „Blaues Wachstum" verfolgen, welches die Förderung einer nachhaltigen Entwicklung durch die Organisation menschlicher Aktivitäten in Raum und Zeit erfordert. Infolgedessen wurde die sog. Marine Raumplanung (MSP) als übergreifende Methode identifiziert, welche einen derart ökosystembasierten Managementansatz menschlicher Aktivitäten leisten kann. In Anbetracht der fehlenden Indikatoren und klaren Zielvorgaben, welche den Grad des Erreichens multipler Managementzielsetzungen beschreiben könnten, kann keine Bewertung der Effektivität eines Managementansatzes durchgeführt werden. Infolgedessen muss die effizienteste Managementstrategie in Richtung eines Ökosystembasierten Marinen Räumlichen Managements (EB-MSP) identifiziert werden.

Ziel dieser Arbeit war die Entwicklung und Prüfung konkreter, raumbezogener Werkzeuge, welche eine transparente Evaluierung räumlicher Managementoptionen sowie derer Konsequenzen für die deutsche Küste ermöglichen. Ein Hauptaugenmerk lag hierbei auf dem Fischereisektor, auf demersalen Fischpopulationen sowie benthischen Ökosystemen.

In **Manuskript 1** dieser Dissertation wurden die ökologischen und ökonomischen Konsequenzen derzeitiger und zukünftiger Managementstrategien abgeschätzt. Derzeitige Managementstrategien beinhalteten Risiken und Konflikte, welche in der deutschen außerordentlichen Wirtschaftszone (AWZ) der Nordsee alltäglich sind. Zukünftige Managementstrategien beinhalteten den Ausbau von Offshore-Windparks. Im ersten Schritt wurde eine räumlich aufgelöste Konfliktanalyse durchgeführt. Die individuellen Konfliktlevels wurden anhand der Verteilung einzelner menschlicher Aktivitäten gemessen. Die Analyse ergab einen ersten Überblick über die Konflikte menschlicher Nutzungen und lieferte somit die Grundlage für ein fundiertes, multisektorales Management. Darüber hinaus konnte eine Zunahme des Konfliktpotenzials auf Kosten des deutschen Fischereisektors vorausgesagt werden.

In einem zweiten Schritt wurden die Risiken für die Ökosystemkomponente „Aufwuchsgebiete der Scholle *Pleuronectes platessa*" qualitativ bewertet. Schollen sind eine Fischart von hohem wirtschaftlichem Interesse. Die Methode erleichterte die Visualisierung des Ursache- und Wirkungsprinzips in Bezug auf den Effekt, welchen menschliche Aktivitäten auf die Aufwuchsgebiete von *P. platessa* haben können. Klare Tendenzen wurden somit sichtbar: Innerhalb der deutschen Küste wird die Beeinträchtigung benthischer Habitate durch den Ausbau von Offshore-Windparks ansteigen, während die durch bodenberührende Fischereigeräte hervorgerufene Belastung abnehmen wird.

In **Manuskript 2** wurde eine Synergieanalyse zur Erschließung geeigneter Co-Nutzungsflächen für die Kombination von Offshore-Windparks mit Offshore-Aquakultur in der deutschen AWZ der Nordsee durchgeführt. Ziel war es, ökologische und ökonomische Chancen potentieller Managementstrategien zu berechnen. Der Vorteil der angewandten Methode lag in der Ausschließung von marinen Flächen, welche ein erhöhtes Risikopotenzial für die untersuchten Arten aufwiesen sowie in der Ausgabe einer Eignungsskala, welche die

Auswahl zukünftiger Co-Nutzungsflächen erleichtert. Somit konnte eine Vielzahl möglicher Flächen für eine Co-Nutzung identifiziert werden.

In **Manuskript 3** dieser Dissertation wurde eine quantitative Risikoanalyse angewandt. Ziel war es, das ökologische Risiko derzeitiger und zukünftiger Managementstrategien zu berechnen. Derzeitige Managementstrategien beinhalteten sämtliche in der deutschen AWZ aktuell vorliegenden Risiken. Zukünftige Managementstrategien beinhalteten den weiteren Ausbau von Offshore-Windparks. Dieser würde die räumliche Ausdehnung der Aktivitäten großer Baumkurrenkutter um 15%, kleiner Baumkurrenkutter um 3% reduzieren. Das Risiko wurde anhand des derzeitigen und zukünftigen Zustands benthischer Habitate bemessen. Mittels der angewandten Methode konnte die Wahrscheinlichkeit der Störung benthischer Habitate ermittelt werden. In Gebieten der deutschen Bucht, die nicht für bodenberührende Fischereigeräte geschlossen sind, stünde demnach eine um 8% zunehmende Störung entsprechender Habitate bevor, was wiederum Konsequenzen für das Ökosystem Benthos nach sich zöge.

In **Manuskript 4** fand eine räumlich aufgelöste Trade-off Analyse Anwendung. Ziel war es, die ökologischen und ökonomischen Kosten und vor allem den Wert des Nutzens zukünftiger Managementstrategien quantitativ und qualitativ und zu berechnen. Die angewandte Methode erleichterte die Integration einer Vielzahl von Datensätzen und somit einer Vielzahl von menschlichen Nutzungen. Somit konnten Kosten und Nutzen räumlicher Managementszenarien auf transparente Art und Weise abgewogen werden, sodass sie theoretisch direkt in Managemententscheidungen für die deutsche Bucht einbezogen werden könnten. Die berechneten Szenarien resultierten zum einen in zunehmende Umweltrisiken wie dem Rückgang unterstützender Ökosystemdienstleistungen (z.B. Habitate) und zum anderen in möglichen Erträgen durch z.B. einen Anstieg sog. bereitstellender Ökosystemdienstleistungen (z.B. aquatische Lebensmittel).

In **Manuskript 5** dieser Dissertation wurde eine Präferenzanalyse individueller Interessenvertreter für die empirische Aufnahme von Prioritäten ökologischer, ökonomischer und sozio-kultureller Managementstrategien am Beispiel sechs internationaler Fallbeispiele (u.a. Nordsee) angewandt. Die Strategieziele wurden auf verschiedenen Managementebenen abgeleitet. Darüber hinaus konnten Interaktionen wie z.B. Konflikte zwischen den Interessenvertretern ermittelt werden. Die Ergebnisse lieferten u.a. einen Konsens über die zukünftigen Managementziele „Reduktion benthischer Zerstörung“ und „Verstärkung umweltfreundlicher Energien“. Eine eher geringe Übereinstimmung individueller Präferenzen wurde hingegen bei den Zielsetzungen „Wettbewerbsfähigkeit der Aquakultur“, „Wettbewerbsfähigkeit des Fischereisektors“, „Schutz von Zielarten/GES“ und „Sicherstellen hoher Ressourcenrenditen“ festgestellt.

Letztendlich konnten die Risiken und Nutzen (Erträge) alternativer Managementziele für die deutsche Bucht abgeschätzt werden. Kenntnisse über das räumliche Ausmaß der Konsequenzen, welche Managementansätze nach sich ziehen, sind fundamental für politische Entscheidungsträger. Desweiteren lässt sich sagen, dass eine Vielzahl an Methoden verfügbar ist, mit deren sich die Effizienz von Managementstrategien in Richtung EB-MSP abschätzen lassen. Auf Basis dieser Werkzeuge können synergetische oder in Konflikt stehende Interaktionen im Voraus dargestellt, Kompromisslösungen identifiziert und die Kommunikation zwischen Interessenvertretern, Planern und politischen Entscheidungsträgern vereinfacht werden. Nichtsdestotrotz erfordert die Anwendung des Großteils dieser Werkzeuge ein umfangreiches Wissen über marine Systeme und darin ablaufende Prozesse. Die angewandten, analytischen Methoden waren sehr multidisziplinär und erlangten zusätzliche Komplexität, wenn die den Werkzeugen zugrundeliegenden Eingabedaten aufbereitet werden mussten. Dementsprechend ist eine wissenschaftliche Untermauerung solcher Methoden nach wie vor unumgänglich, um Managementstrategien zu evaluieren und

Beratung hinsichtlich eines EB-MSP leisten zu können, welches auf eine nachhaltige Nutzung sowie die Gesundheit des Ökosystems Deutsche Bucht abzielt.

Outline of Publications

The following overview outlines the five publications which are included in this thesis. This outline serves as a clarification of each author's contribution to the respective manuscript.

Manuscript 1

A spatially explicit risk approach to support marine spatial planning in the German EEZ

Antje Gimpel, Vanessa Stelzenmüller, Roland Cormier, Jens Floeter, Axel Temming

Antje Gimpel (AG) performed the analysis and text writing under close cooperation with Vanessa Stelzenmüller (VS), Axel Temming (AT) and Jens Floeter (JF), who critically reviewed the manuscript. Roland Cormier (RC) provided valuable comments. The modelling approach was conducted by AG under supervision of VS, RC and JF.

The manuscript is published in the peer reviewed Journal Marine Environmental Research (2013). doi:10.1016/j.marenvres.2013.02.013

Manuscript 2

A GIS modelling framework to evaluate marine spatial planning scenarios: Co-location of offshore wind farms and aquaculture in the German EEZ

Antje Gimpel, Vanessa Stelzenmüller, Britta Grote, Bela H. Buck, Jens Floeter, Ismael Núñez-Riboni, Bernadette Pogoda, Axel Temming

AG performed the analysis and text writing under close cooperation with VS, AT and JF, who critically reviewed the manuscript. Bela H. Buck (BHB) and Bernadette Pogoda (BP) provided valuable comments. The modelling approach was conducted by AG under supervision of VS, Ismael Núñez-Riboni (IN) and JF.

The manuscript is published in the peer reviewed Journal Marine Policy (2015).
doi: 10.1016/j.marpol.2015.01.012

Manuscript 3

Quantitative environmental risk assessments in the context of marine spatial management: current approaches and some perspectives

Vanessa Stelzenmüller, Heino O. Fock, Antje Gimpel, Henrike Rambo, Rabea Diekmann, Wolfgang N. Probst, Ulrich Callies, Frank Bockelmann, Herman Neumann, Ingrid Kröncke

VS performed the risk assessment and text writing under close cooperation with Heino O. Fock (HOF), AG, Henrike Rambo (HR), Rabea Diekmann (RD), Wolfgang N. Probst (WNP) Ulrich Callies (UC), Frank Bockelmann (FB), Herman Neumann (HN) and Ingrid Kröncke (IK). AG, HR, RD and WNP performed the literature review. The results of the review have been consolidated by AG and illustrated by HS.
The manuscript is published in the peer reviewed ICES Journal of Marine Science (2014). doi:10.1093/icesjms/fsu206

Manuscript 4

Evaluation of spatial management scenarios supporting an ecosystem-based approach to MSP - the German case

Antje Gimpel, Vanessa Stelzenmüller, Jens Floeter, Axel Temming

AG performed all graphical presentations, modelling and text writing under close cooperation with JF and VS. The co-authors, VS, JF and AT critically reviewed the manuscript. The modelling approach was conducted by AG under supervision of VS, AT and JF.
The manuscript will be submitted to the peer reviewed Journal Ecosystem Services (2015).

Manuscript 5

Multiple interests across European coastal waters: the importance of a common language

Jorge Ramos, Katrine Soma, Øivind Bergh, Torsten Schulze, Antje Gimpel, Vanessa Stelzenmüller, Timo Mäkinen, Gianna Fabi, Fabio Grati, Jeremy Gault

Jorge Ramos (JR) performed graphical computations of results and text writing under close cooperation with all other authors. The indicator and questionnaire development as well as the interview surveys have been conducted per Case Study (CS): the North Sea coast CS was conducted by AG, Torsten Schulze (TS), VS and Katrine Soma (KS), the Hardangerfjord CS was conducted by Øivind Bergh (OB), the Atlantic coast CS was conducted by Jerermy Gault (JG), the Algarve coast CS was conducted by JR, the Adriatic Sea coast CS was conducted by Gianna Fabi (GF) and Fabio Grati (FG), and the Baltic Sea coast CS was conducted by Timo Mäkinen (TM). JR performed the preference analysis under supervision of KS.
The manuscript is published in the peer reviewed ICES Journal of Marine Science (2014).
doi: 10.1093/icesjms/fsu095

1. General Introduction

1.1 Ecosystem-based approach to Marine Spatial Planning: a brief introduction

The worlds' oceans storage an enormous amount of carbon, coastal areas serve as resort for recreational activities and marine ecosystems deliver a variety of aquatic products nourishing millions of people (Godfray et al., 2010; FAO, 2014; Halpern et al., 2008a; Shelton et al., 2014). Facing climate change, the ongoing depletion of finite resources and the destruction of natural habitats, which is accompanied with a loss of biodiversity, marine ecosystems are increasingly considered collectively. The term ecosystem, which derived from ancient Greek and can be translated with 'house' (oikós) and 'the combined' (sýstema), was defined by art. 2 of the Convention on Biological Diversity (CBD) as "a dynamic complex of plant, animal and micro-organism communities and their non-living environment interacting as a functional unit" (CBD, 1995). In order to maintain the ecosystem in a healthy, productive and resilient state, assuring the use of the ecosystem services it provides for current and future generations, the concept of Ecosystem-Based Management (EBM) attracts worldwide attention (Halpern et al., 2012; Katsanevakis et al., 2011; Douvere, 2008; McLeod et al., 2005; Obama, 2010; EC, 2012). EBM considers the whole ecosystem, including humans and featuring the pressures they are exerting (McLeod et al., 2005; Obama, 2012).

In 2008 the European Parliament and the Council established the Marine Strategy Framework Directive (MSFD; Directive 2008/56/EC), aiming to achieve or maintain Good Environmental Status (GES) of marine ecosystems. The GES promotes an environmental status of marine waters where these provide ecologically diverse and dynamic oceans and seas which are clean, healthy and productive. Art. 1(3) of the directive points out that "*marine strategies shall apply an ecosystem-based approach to the management of human activities, ensuring that the collective pressure of such activities is kept within levels compatible with the* *achievement of GES [...]*". Necessary measures shall be taken by the European Member States (MS) by the year 2020 at the latest (EC, 2008b).

About the same time, the European Council endorsed the Integrated Maritime Policy (IMP) for the European Union, an approach to ocean management and maritime governance on the environmental pillar of the MSFD. The objectives of the IMP are to reaffirm the maritime dimension of the EU, to support the sustainable development of seas and oceans, to provide better protection of the state of the ecosystem and to develop coordinated, coherent and transparent decision-making in relation to the Union's sectoral policies (EC, 2012). Sustainable development shall meet the needs of the present without compromising the ability of future generations to meet their own needs. Strengthening Europe's focus on sustainable development and economic growth, a resolution implementing an integrated approach to maritime affairs is adopted in 2010: The Europe 2020 strategy features the European demand employment, competitiveness and social cohesion. Key initiatives covered the EU Strategic Energy Technology Plan (SET-plan) also known as '20-20-20' target resolved in 2009. Its overarching goals are (i) a 20 % reduction in EU greenhouse gas emissions from 1990 levels, (ii) a raising share of EU energy consumption produced from renewable resources to 20 % and (iii) a 20 % improvement in the EU's energy efficiency by 2020. In 2011 the Common Fisheries Policy (CFP) was reformed to guarantee for sustainable fisheries, quality food supply and attractive and safer jobs. Responding to the Council and the European Parliament which were requesting for further developments, in 2012 the objective for the coming years called 'Blue Growth: opportunities for marine and maritime sustainable growth' was adopted. All of these policies, directives, strategies and objectives identify Marine Spatial Planning (MSP) as a cross-cutting policy tool that contributes to "*sustainable growth of maritime economies, the sustainable development of marine areas and the sustainable use of marine resources"* while *"applying an ecosystem-based approach as referred to in Article 1(3) of*

Directive 2008/56/EC with the aim of (...) achievement of good environmental status" (EC, 2014a).

MSP integrates ecological, social, and economic interests, interactions between human activities, regardless of whether cross-border or inter-sectoral nature, whether conflict or synergy (Halpern et al., 2008b; Ehler and Douvere, 2009; Foley et al., 2010). As shown in Figure 1, its process is characterized as dynamic and evolving, integrating multiple feedback loops and permanent revisions (Ehler and Douvere, 2009). Since MSP is a public process, the implementation of strategic plans integrates greater accountability and transparency of decision-making by including a wide range of stakeholders from all sectors (Ehler and Douvere, 2009; Wever et al., 2015; Gilliland and Laffoley, 2008; Stelzenmüller et al., 2013). Due to continuous monitoring and evaluation performances it speeds up decision-making. As a strategic tool, MSP can allocate space for upcoming activities such as e.g. aquaculture at sites with both favourable operational characteristics (economic and ecological) as well as lower potential for conflict with other sectors (FAO, 2013; Stelzenmüller et al., in preparation; Guerry et al., 2012; Christie et al., 2014). Consequently, it increases the effectiveness of investments.

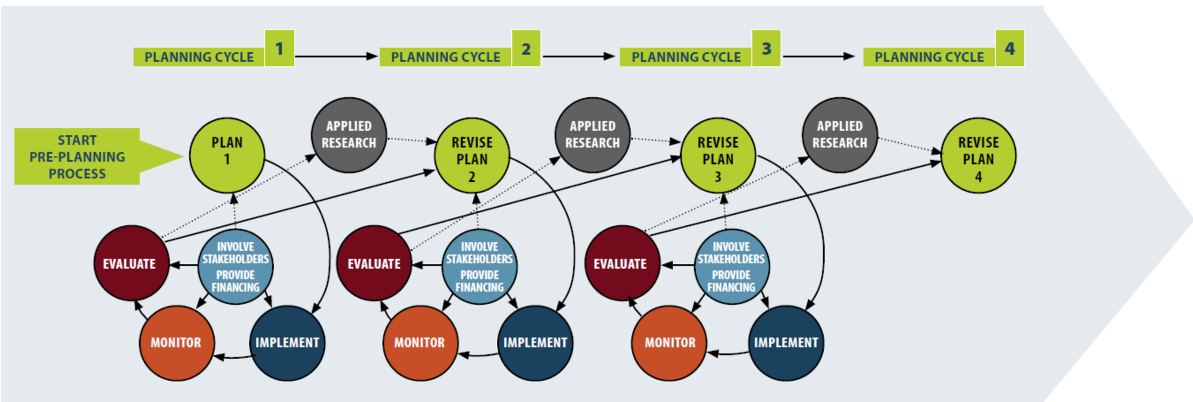


Figure 1: The continuing MSP planning cycle, taken from Ehler and Douvere (2009).

Consequently, the EU adopted a common framework for MSP (Directive 2014/89/EU) in 2014, including a minimum set of requirements. In line with the IMP, the ecosystem approach is an overarching principle for MSP (EC, 2008a). Bringing all those facts about MSP together, the EU commission mentions 10 key principles in its roadmap to MSP in practice:

(1) Using MSP according to area and type of activity, (2) Defining objectives to guide MSP, (3) Developing MSP in a transparent manner, (4) Stakeholder participation, (5) Coordination with Member States - Simplifying decision processes, (6) Ensuring the legal effect of national MSP, (7) Cross border cooperation and consultation, (8) Incorporating monitoring and evaluation in the planning process, (9) Achieving coherence between terrestrial and maritime spatial planning relation with Integrated Coastal Zone Management (ICZM), (10) A strong data and knowledge base (EC, 2008a).

ICZM which is a marine management tool applied to control policy processes affecting coastal zones is figuratively considered (par. 9, the "*planning process should take into account land-sea interactions*") (EC, 2014a; EC, 2008a; EC, 2014b).

As MSP should support the implementation of the MSFD, responsible EU sections were reconsidered collectively: Under the Juncker Commission, the Directorate-General for Maritime Affairs and Fisheries (DG MARE), usually responsible for MSP, and the Directorate-General Environment (DG ENV), usually responsible for the MSFD have been combined "*to reflect the twin logic of "Blue" and "Green" Growth: Protecting the environment and maintaining the European competitiveness*" (EC, 2014c). Blue Growth pursues sustainable growth of maritime economies, the sustainable development of marine areas and the sustainable use of marine resources. The realization of Blue Growth and GES (which is effectively called 'Green Growth') is a tall order. Nevertheless, with legal EU frameworks such as the Water Framework Directive (WFD), the Habitat and Birds Directive (HBD) or the Flora and Fauna Directive (FFH), which is the driver for the designation of Marine Protected Areas (MPAs) such as the Natura 2000 sites, a network of nature protection areas established under the 1992 Habitats Directive, the foundation is laid. Furthermore, the implementation of Ecosystem-Based MSP (EB-MSP) is supported by EU funding instruments such as the European Maritime and Fisheries Fund (EMFF). The EMFF is structured around the pillars of fisheries (CFP), aquaculture and IMP. Another funding instrument is the EU Horizon 2020 program for research and innovation (EC, 2014d). On a final note, scientific and technical support is promoted by the Joint Research Centre - Institute for Environment (JRC IES) of the EU.

1.2 Sustainable use and ecosystem health: towards EB-MSP in the German Bight

According to the European Roadmap to MSP, a "*maritime spatial plan may not need to cover a whole area (e.g. EEZ of a Member State). For densely used or particularly vulnerable areas, a more prescriptive MSP might be needed*" (EC, 2008a). The allocation of the German waters is based on the classification of the United Nations Convention of the Law of the Sea (UNCLOS), an international agreement from 1994 defining the rights and responsibilities of nations with respect to their use of the world's oceans: The territorial sea up to a limit of 12 nautical miles (nm) from a 'baseline' (normally the low water line), the Exclusive Economic Zone (EEZ) beyond these 12 nm that can extend up to 200 nm from the baseline for the territorial sea. "*The competence of a coastal State to undertake MSP in its EEZ is therefore restricted to these issues and may not derogate from the rights enjoyed by other States in such waters including the freedom of navigation and the right to lay submarine cables"* (EC, 2009). As the EU is part of the UNCLOS States Parties, all member states are bound by this convention.

Unlike EU regulations, which are immediately effective finally adopted by the commission, directives just have to conform to minimum requirements. The MSP directive has to be integrated in national legislation by 2016 and has to be completed in 2021 (EC, 2014b). The method of implementation is dedicated to the member states (EC, 2014b). According to the MSP key principle 8 (section 1.1), cross-border coordination with other member states is desired (EC, 2008a). Intergovernmental organizations such as OSPAR (Oslo and Paris convention), HELCOM (Helsinki Commission) or ICES (International Council for the Exploration of the Sea) feature working groups such as the ICES Working Group on Marine Spatial Planning and Coastal Zone Management (WGMPCZM), which develop common standards and action plans on an international scale (Gilliland and Laffoley, 2008; EC, 2008a; Cormier et al., 2010).

MSP in Germany is based on the Federal Land Use Planning Act that was extended to the EEZ. The responsibility lies with the Federal Agency for Shipping and Hydrography (BSH) as a representative of the Federal Ministry of Transport and Digital Infrastructure (BMVI) (Berkenhagen et al., 2010; BSH, 2009a). Spatial plans for the territorial waters (up to 12 nm) were developed by the coastal Federal States. On behalf, the Ministries of Interior and the Ministries of Rural Areas, Nutrition, Agriculture and Consumers Rights of each Schleswig Holstein and Lower Saxony are in pursuit of autonomously developed spatial management strategies (LS, 2005; SH, 2003). While Schleswig Holstein came up with an Integrated Coastal Zone Management Strategy (IKZM, Integriertes Küstenzonenmanagement), Lower Saxony developed a MSP concept (ROKK, Raumordnungskonzept für das niedersächsische Küstenmeer) (Tab. 1).

The German plans are regulatory and enforceable. The federal plan for the North Sea went into effect in September 2009; the federal plan for the Baltic Sea in December 2009 (BSH, 2009a). The plans are based on zoning, creating areas that favour a particular use and areas where certain uses are prohibited (Schultz-Zehden and Gee, 2013; Jay and Gee, 2014; Stelzenmüller et al., in preparation). Where other nations are following an integrated, strategic and participatory planning process (Katsanevakis et al., 2011; Ehler and Douvere, 2009), the German MSP rather grew together, stimulated by the effect of newly developed maps displaying numerous proposals for large-scale Offshore Wind energy Farms (OWF) (UNESCO, 2014).

The plan for the German EEZ of the North Sea refers to a surface area of 28,539 km². Next to other uses, the main human activities regulated are shipping, oil and gas exploitation, cables and pipelines, renewable energy development, and aggregate extraction (Buck et al., 2004; BSH, 2009b). The allocation of fishing activities is not included (Fock, 2011; Stelzenmüller et al., 2011). As marine aquaculture is merely taking place nearshore in terms of mussel and oyster cultures within the Wadden Sea National Park, it is not included as well. Although offshore cultivation is currently conducted in various pilot studies, it is not yet done at commercial scale (Buck et al., 2004; Buck and Krause, 2012).

In line with the Europe 2020 strategy, joint efforts are recently undertaken towards a general principle for spatial planning in the federal territory of Germany. The overarching goal is to concretize and prioritize collective objectives beyond the scope of the federal states. The Conference of Ministers for Spatial Planning (MKRO) is responsible for the preparation of the drafts, followed by consultation of experts. The collaboration led to the "General principles and action strategies for spatial planning in Germany", published in 2006 and 2013 (BMVI, 2015). Such an approach cover principle 3 (Developing MSP in a transparent manner), 4 (Stakeholder participation), 5 (Coordination with member states), 6 (Ensuring the legal effect of national MSP, in as far as a solid administrative framework is concerned), 7 (Cross border cooperation and consultation) and 9 (Achieving coherence between terrestrial and maritime spatial planning) of the EU 10 key principles to MSP in practice (EC, 2008a).

Table 1: Institutions concerned with Marine Spatial Planning (incl. Strategies pursued) and the sections affiliated.

Global	Regional	Supranational	Sectoral		National	Federal States
UNCLOS						
	ICES HELCOM OSPAR					
		EU (Europe 2020 Strategy): *Integrated Maritime Policy (IMP)*				
			DG MARE (Blue Growth): *MSP directive*	DG ENV (GES): *MSFD*		
					BMVI: *German MSP* BSH: *MSP German EEZ*	
						Federal State Authorities: Lower Saxony *ROKK* (12nm) Schleswig Holstein *IKZM* (12nm)

UNCLOS, United Nations Convention on the Law of the Sea; ICES, OSPAR, Oslo and Paris convention; HELCOM, Helsinki Commission; ICES, International Council for the Exploration of the Sea; EU, European Union; DG MARE, Directorate-General for Maritime Affairs and Fisheries; DG ENV, Directorate-General Environment; BMVI, Federal Ministry of Transport and Digital Infrastructure; BSH, Federal Agency for Shipping and Hydrography; ROKK, Spatial Planning Concept for the Coast of Lower Saxony; IKZM, Integrated Coastal Zone Management for the Coast of Schleswig Holstein

1.3 Case study illustration: marine management issues currently a matter of debate

As mentioned in section 1.1 and in line with the IMP and the Europe 2020 Strategy, the ecosystem approach is an overarching principle for MSP (EC, 2008a). According to art. 13(4) of the MSFD, member states need to include "*spatial protection measures, contributing to coherent and representative networks of marine protected areas, adequately covering the diversity of the constituent ecosystems".* Considering the conservation perspective, the member states needed to account for MPA sites to support the achievement of GES requirements of the MSFD. Those were implemented under the Natura 2000 protocol and the FFH directive protecting both, habitats and species (Fock, 2011; EC, 2008b; EC, 2014a). The designation of ultimately 47 % of the total German maritime space and 70 % in coastal waters as Natura 2000 sites required close collaboration of all member states and their respective institutions (see Table 1). When being enforced, activities exerting pressure (e.g. bottom trawling activities) will be displaced.

The implementation of the SET-plan (see section 1.1) boosts the OWF development, speeding up the race for space in the already heavily used offshore and coastal waters of the German Bight: With each wind farm licensed the fisheries loose access to traditional fishing grounds due to the safety requirements imposed by wind farm development. As a result, the (bottom trawling) demersal fisheries will be displaced and forced to concentrate their activities in smaller areas to maintain their level of catch. Located in the North Sea, the German Bight is at the centre of the distribution range of essential fish species such as plaice (*Pleuronectes platessa*) with the Wadden Sea as the most important nursery ground determined by multiple factors such as temperature, salinity, depth, food or dissolved oxygen (Wennhage et al., 2007; Yamashita et al., 2001). Human activities occurring in the North Sea exert a number of pressures on the coastal and marine environment (Halpern et al., 2008), which can be additive, synergistic or antagonistic (Halpern et al., 2008b; Stelzenmüller et al., 2010b). These pressures also have a physical impact on the seabed and are likely to have adverse effects on the integrity of marine habitats (Foden et al., 2011). Those effects include any response to an action's impact. Demersal fisheries occurring on the continental shelf of the southern North Sea are considered as having a major impact on the seafloor (Fock et al., 2011). Leading to an increased frequency of disturbance, essential benthic habitats such as the nursery grounds of plaice are facing an elevated magnitude of impact. It is safe to assume that the degradation of the benthic characteristics of essential habitats will lead to a loss of plaice productivity and decreased landings that would subsequently result in further socio-economic impacts, reasoned by the loss of income. However, the magnitude of the pressure and the probability that it occurs, its quantitative impacts on the ecosystem and the degree of uncertainty involved, especially under alternative management strategies, is hardly predictable. Further, the remaining pressures affecting the integrity of the seafloor, exerted by other drivers than bottom trawling, are not known yet or far less edited in a spatially explicit way. EBM being effective and efficient should account for the cumulative effects of all human activities on the marine environment at meaningful ecological scales (Halpern et al., 2008a; Stelzenmüller et al., 2010c; Stelzenmüller et al., in preparation; Katsanevakis et al., 2011).

Another future management objective addressing sustainable use being currently a matter of debate is offshore aquaculture. Further steps towards the Europe 2020 strategy should involve efforts to create a stable environment attractive to investors. MSP, contributing to the aims of EBM and the development of land-sea links, should facilitate among others the development of experimental and other measures combining the generation of renewable energy and fish farming (EC, 2014b; EC, 2011). In art. 51 of EU regulation no 508/2014 "*the identification and mapping of the most suitable areas for developing aquaculture*" is fostered. The regulation establishes the EMFF in support of MSP, promoting a balanced and inclusive territorial development of fisheries and aquaculture areas (EC, 2014d). In the course of the

EU Horizon 2020 Framework Programme the need for an optimization of contributions of fisheries and aquaculture to food security was raised (EC, 2015). Offshore aquaculture production may contribute to food security and relief some of the pressures on wild stocks. Considering the already mentioned race for space, further attention needs to be paid to the increasing requirements for water resources. In addition, environmental health is of particular importance to the pressures human activities already exert on the marine environment. The coupling of environmental safety and sustainable use of resources with stakeholder's needs and expectations, also mentioned by Katsanevakis et al. (2011), can be addressed by integrating the concept of co-location of marine OWFs and Integrated Multi-Trophic Aquaculture (IMTA). IMTA systems combine aquaculture species to recycle effluent dissolved and particulate nutrients from a higher trophic-level species (fish) to nourish extractive, lower trophic-level species, such as filter feeders (mussels, oysters), polychaetes, sea cucumbers and/or seaweed (Neori et al., 2007; Troell et al., in review). These systems aim at balanced nutrient budgets and minimize the waste production originating from fed aquaculture species through the filtering capacity of other extractive species clearing the water (Troell et al., 2009). Moreover, by using nutrient losses of higher trophic-level species as feeding products, IMTA could provide additional economic benefits (Neori et al., 2007).

1.4 The importance of evaluating marine management strategies

Competition for maritime space and the need for sustainable food production highlight the importance of efficient planning towards coordinated, coherent and transparent decision-making as stipulated by the IMP (Stelzenmüller et al., 2013; Soma et al., 2013; Godfray et al., 2010; Rosenthal et al., 2012; EC, 2012). Assessing marine management strategies decision makers aim to achieve high-level objectives with (e.g. Blue Growth or GES) still constitutes a challenge. MPAs, fishing grounds, aquaculture or IMTAs, maritime infrastructures such as cables, pipelines, shipping lanes and oil, gas and wind installations need to be managed collectively. Different marine and coastal activities have diverse economic, environmental, and socio-cultural objectives, which can lead to conflict when these multidimensional activities coincide spatially or temporally (Stelzenmüller et al., 2013). Aiming to pursue the European 2020 Strategy, the efficiency of management strategies needs to be evaluated by using a transparent approach. Therefore, an economic evaluation of the trade-off between risks (costs) and economic return (services) is needed (Polasky et al., 2008; White et al., 2012a). Subsequently, management strategies can be weighed, integrating all levels of complexity being of economic, environmental or socio-cultural nature. When managers plunge into these diverse levels of complexity, a lack of understanding naturally arises. This might become even more relevant when it comes to policy makers. As a consequence, spatial management decisions are based primarily on economic criteria (Polasky et al., 2008). In fact, scientific advice establishing a strong data and knowledge base, e.g. related to pressure-state relationships, is highly significant to counteract on this effect (EC, 2008a). As required by art. 1(3) of the MSFD, such an approach enables the integration of environmental (previously uncertain) criteria in holistic trade-off assessments towards EB-MSP, finally balancing ecosystem health against sustainable use on a level compatible with the GES (EC, 2008b).

1.5 Concepts and tools to assess spatial marine management strategies

According to art. 6 of the MSP directive, member states shall establish procedural steps towards EBM and sustainable growth while promoting coherence between different planning processes. Considering the number of institutions involved in the German planning processes (Tab. 1), coherence defies coordinated action plans. According to the knowledge of the author, there is no framework proposed on how to handle the new responsibilities assigned. Zamzow (2015) put a finer point to this basic problem describing following situation: In the course of the designation of OWF areas the cable lines required were not accounted for, and posed a “surprising” difficulty for German planners. However, neither the BSH nor the authorities of the Federal States rated a stronger top down control in processes instigated by the European Commission as beneficial. However, in consideration of the ‘General principles and action strategies for spatial planning in Germany’ as defined by the MKRO (section 1.2), annotations how to achieve those actions are hardly given (BMVI, 2015). The provision of transparent tools is needed to align the MSP procedures while supporting EBM towards sustainable growth. Further, working together with stakeholders, managers and policy makers, a common language is of great importance. It implies the usage of similar wordings to align MSP procedures and ensure flawless communication across all (environmental, economic, and social) disciplines.

As described in section 1.4, the evaluation of management strategies is driven by the assessment of criteria describing its efficiency. This assessment is based on the trade-off balancing the risks and return of a strategy. After that, alternative management strategies can be weighed. In order to appropriate the data required for a holistic trade-off assessment, various categories of tools are available (Fig. 2): Depending on the type of management issue, those tools can address: conflicts and/or synergies (1, 2); qualitatively and/or quantitatively assessed risks (3, 4) and the evaluation of multiple objectives at once, e.g. by doing a trade-off analysis and/or a stakeholder preference analysis (5, 6). These entire tools can additionally be distinguished according to the scale they address (local to regional) and their application to current and/or future scenarios. Those tool categories underlie further operational tools utilized to analytically assess the three-dimensional marine space and its components. The selected tools allow the spatially explicit description of conflicts, synergies, risks or benefits emerging from individual management measures. In order to illustrate those tools and methods applicable to evaluate management strategies, they are presented by reference to fisheries, an activity currently not included in the German MSP (BSH, 2009a).

1.5.1 Conflict and Synergy analysis

"*The fisheries and aquaculture sector is facing major challenges*" (FAO, 2014). To gain a tendentious overview about the spatial conflict potential of human uses, conflict categories can be mapped out. Therefore, conflict combinations of fisheries and other activities, analysed using a general scoring scheme, can be combined with a geo-spatial footprint analysis using a Geographic Information System (GIS) (Lee and Stelzenmüller, 2010). In order to assess synergies, selected activities can be evaluated accounting for their spatial overlap. In combination with a GIS, specific sites enabling the co-location of human uses can be mapped out. The integration of a new activity such as aquaculture requires the compilation of multiple factors predicting suitable sites. This approach can be facilitated in application of a Multi-Criteria Evaluation (MCE) technique. Integrating an Ordered Weighted Average (OWA) process, the weightings in factor combination can be subjected to a risk analysis.

Figure 2: Concepts and tools addressing the evaluation of spatial management strategies. The studies conducted during this dissertation address the evaluation of spatial management strategies at different scales (local to regional) and timelines: 1, 3 and 4 represent studies evaluating current (status quo) AND future management strategies, 2, 5 and 6 represent studies just focusing on future management strategies. The dashed blue line shown guides through these studies and simultaneously symbolizes the red thread of this dissertation. The black lines symbolize the steps within each study. The blue circles contain the *tool category*; the purple ones depict the *issues of concern*. The green circles show the *indicators for evaluation* and the orange boxes embody the *concepts and tools used during the evaluation process*. Finally, the ensuing *publications* covering these studies are shown, highlighted by grey circles. Adapted from Ehler and Douvere (2009).

1.5.2 Qualitative and quantitative risk assessment

"*Although concern about bottom trawling is expressed in historical documents going back to the late fourteenth century (…) the proportion of the North Sea surface area trawled at least once a year increased (…) to 60% at the beginning of the twenty-first century*" (Rijnsdorp et al., 2015). To gain a tendentious overview of the recent impacts the ecosystem (or rather its components) is facing, a qualitative risk analysis can be performed. Species Distribution Models (SDM) such as a Generalised Additive Model (GAM), boosted regression tree models or simple GIS overlays can give a first overview about the spatial distribution of essential ecosystem components (e.g. habitats). Subsequently, fisheries and other human uses can be allocated to generic pressure categories (e.g. abrasion) to evaluate their (combined) effects on the ecosystem - accounting for the spatial overlap and sensitivity to those pressures. To integrate the predicted likelihood of fishing pressures, a quantitative risk analysis can be applied. In order to account for uncertainty related to impact prediction, the GIS can be coupled with a Bayesian Belief Network (BBN) (Stelzenmüller et al., 2010a). Finally, the risks coming from (future) spatial management strategies can be assessed.

1.5.3 Assessing multiple management strategies using trade-off and stakeholder preference analysis

"*Conserving biodiversity, while at the same time meeting expanding human needs, is an issue of utmost importance*" (Polasky et al., 2008). In order to define the delivery of marine ecosystem services (e.g., nursery habitats) nearshore and coastal habitats are important for, GIS-based tools can be applied. It enables the mapping and the estimation of changes under different management strategies. Further, trade-offs among services (e.g. food provided by fisheries) can be considered (Guerry et al., 2012). Such estimations require the reduction of services to a common unit (e.g. €). Therefore it facilitates the integration of multiple data sets and human uses, respectively. Costs and benefits of management strategies can be depicted in a transparent form and consequently be integrated in decision making processes (Katsanevakis et al., 2011). An integrated assessment processes brings together knowledge and elements from a variety of disciplinary sources (models, data and assessment methods). To conduct fully integrated assessment processes across environmental, economic, and social dimensions of marine systems a Multi-Criteria-Analysis (MCA) can be adjusted to empirically analyse multiple priorities given to management strategies (Wever et al., 2015; Stelzenmüller et al., in preparation). Further, a ‘common language’ between stakeholders, managers and decision makers can be established leading to increased transparency and therefore acceptance of the society about decisions making in marine and coastal areas (Kelly et al., 2014; Stelzenmüller et al., in preparation).

“Monitoring, evaluation and adaptation are necessary to ensure that marine management measures are both effective and efficient” (Katsanevakis et al., 2011). As mentioned in sections 1.3 and 1.4, the German MSP process is still improvable by scientific advice and Decision Support Tools (DST) that facilitate organizational processes underpinning decision-making processes. Needed are ecosystem-based, integrated, place-based, adaptive, strategic and anticipatory as well as participatory studies conveying the methods, tools and indicators which are already usable (Ehler and Douvere, 2009). The evaluation of management strategies represents a crucial step towards reasonable adaption. These analyses should focus not only on economic values, but also on ecological, social and cultural values associated with coastal communities and the sea, many of which are extremely difficult to measure (Gee and Burkhard, 2010). Since ecosystem values are important to balance sustainable use and ecosystem health, gains and losses need to be weighed carefully. In application of the concepts and tools out there, decision-making towards EB-MSP is supported (White et al., 2012b).

1.6 Thesis objectives

The intention of this dissertation is to showcase the application of spatial explicit tools which allow a transparent evaluation of management strategies for the ecosystem ‘German Bight’, regardless of whether ecological, economic or socio-cultural nature. Further, the aim is to compare those tools in order to identify the ones useful for EB-MSP. Besides, risks and returns coming from alternative management objectives are assessed, focussing on ecosystem health and sustainable use in particular. The effects are facilitated by reference to the fishery sector, demersal fish populations and the benthic ecosystem. Thereby, management strategies are evaluated according to their efficiency towards EB-MSP in the German Bight.

All studies compiled for this dissertation are schematically outlined in Figure 2, and related to the manuscript (MS) the referring analyses were conducted in:

Initially, economic and ecological consequences of human activities spatially coinciding were analysed. In application of a conflict analysis, potential conflicts between human activities due to current and future management strategies were assessed. While the current management strategy involved conflicts occurring in the German Exclusive Economic Zone (EEZ) of the North Sea on a daily business, the future management strategy integrated the realisation of OWF development (**MS 1**). Further, potential synergies accounting for space issues were analysed. This synergy analysis was related to the co-location of offshore aquacultures and wind farms in the German EEZ of the North Sea (**MS 2**).

Subsequently, the environmental risks outgoing from human activities were assessed. To start, a qualitative risk assessment was conducted. The risks have been measured using the pressures human activities exert on important ecosystem components such as the nursery grounds of *Pleuronectes platessa*, a fish species of high commercial meaning. Again, the risks were assessed based on a daily business and, later on, by integrating the realisation of the

OWFs (**MS 1**). Next, a quantitative risk assessment was applied in order to conduct a probabilistic measure of risk using the 'current and future state of benthic communities' while accounting for scientific uncertainty about impact prediction. Here, the current management strategies involved the business as usual, the future one taking a caused spatial shift of 15% of the total fishing frequency of large beam trawlers and 3% of the small beam trawlers as granted due to the realisation of the OWFs (**MS 3**).

Finally, future management strategies were assessed based on socio-cultural, economic and ecological objectives. At first, a spatially explicit trade-off assessment of ecosystem services under multiple management scenarios was conducted. Those involved (i) the utilisation of ES provided by the marine ecosystem of the German Bight of the North Sea on a daily business, (ii) the realisation of the OWF development and spatial closures of Natura 2000 sites as a future perspective, causing a spatial shift of fisheries and an increase of renewable energy production, and (iii) the realisation of co-location as a management option in MSP (**MS 4**).

Consequently, a stakeholder preference analysis was applied in order to empirically prioritize future management strategies. Therefore future management objectives derived from different levels of complexity ((i) the main goal to achieve (i.e. a sustainable sea), (ii) the sets of legislative frameworks/spatial plans consulted (e.g. Water Framework Directive), (iii) the main objectives addressed (ecological, economic and socio-cultural) and, (iv) the operational objectives derived by groups of stakeholders involved (sectors)) were compared against stakeholder interests (**MS 5**).

1.7 References

Berkenhagen, J., Döring, R., Fock, H. O., Kloppmann, M. H. F., Pedersen, S. A., and Schulze, T. 2010. Decision bias in marine spatial planning of offshore wind farms: Problems of singular versus cumulative assessments of economic impacts on fisheries. Marine Policy, 34: 733-736.

BMVI 2015. Bundesministerium für Verkehr und Digitale Infrastruktur. Leitbilder Raumentwicklung. http://www.bmvi.de/SharedDocs/DE/Artikel/G/Raumentwicklung/raumentwicklung-einleitung.html, Retrieved: June 18th 2015, 14:15.

BSH 2009a. Bundesamt für Seeschifffahrt und Hydrographie. Raumordnung in der AWZ. http://www.bsh.de/de/Meeresnutzung/Raumordnung_in_der_AWZ/index.jsp, Retrieved: June the 17th, 2015, 14:15.

BSH 2009b. The Federal Agency for Shipping and Hydrography (Bundesamt für Seeschifffahrt und Hydrographie, BSH), Map North Sea. http://www.bsh.de/en/Marine_uses/Spatial_Planning_in_the_German_EEZ/documents2/MSP_DE_NorthSea.pdf, Retrieved: June 2nd 2014, 17:00.

Buck, B. H., and Krause, G. 2012. Integration of Aquaculture and Renewable Energy Systems. Encyclopaedia of Sustainability Science and Technology, 1: 511-533.

Buck, B. H., Krause, G., and Rosenthal, H. 2004. Extensive open ocean aquaculture development within wind farms in Germany: the prospect of offshore co-management and legal constraints. Ocean & Coastal Management, 47: 95-122.

CBD 1995. Convention on Biological Diversity. https://www.cbd.int/ecosystem/, Retrieved: April 5th 2015, 17:45.

Christie, N., Smyth, K., Barnes, R., and Elliott, M. 2014. Co-location of activities and designations: A means of solving or creating problems in marine spatial planning? Marine Policy, 43: 254-263.

Cormier, R., Kannen, A., Morales Nin, B., Davies, I., Greathead, C., Sarda, R., Diedrich, A., et al. 2010. Risk-based frameworks in ICZM and MSP decision-making processes. ICES ASC, 2010.

Douvere, F. 2008. The importance of marine spatial planning in advancing ecosystem-based sea use management. Marine Policy, 32: 762-771.

EC 2008a. COMMISSION OF THE EUROPEAN COMMUNITIES. COMMUNICATION FROM THE COMMISSION Roadmap for Maritime Spatial Planning: Achieving Common Principles in the EU COM(2008) 791 final.

EC 2008b. European Commission. DIRECTIVE 2008/56/EC OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL of 17 June 2008 establishing a framework for community action in the field of marine environmental policy (Marine Strategy Framework Directive). Official Journal of the European Union, L 164/19.

EC 2009. European Commission Directorate-General for Maritime Affairs and Fisheries. Legal aspects of maritime spatial planning. Summary report. Luxembourg: Office for Official Publications of the European Communities.

EC 2011. European Commission. REGULATION (EU) No 1255/2011 OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL of 30 November 2011 establishing a Programme to support the further development of an Integrated Maritime Policy. Official Journal of the European Union, L 321/1.

EC 2012. European Commission. Progress of the EU's Integrated Maritime Policy – Report from the commission to the European Parliament, the council, the European Economic and social committee and the committee of the Regions. Luxembourg: Publications Office of the European Union: 11 pp.

EC 2014a. European Commission. DIRECTIVE 2014/89/EU OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL of 23 July 2014 establishing a framework for maritime spatial planning. Official Journal of the European Union, L 257/135.

EC 2014b. European Commission. Press release: Commission welcomes Parliament's adoption of Maritime Spatial Planning legislation. IP/14/459.

http://europa.eu/rapid/press-release_IP-14-459_en.htm?locale=en, Retrieved: June 15th 2015, 17:09.

EC 2014c. European Commission. Press release: The Juncker Commission: A strong and experienced team standing for change. IP/14/984. http://europa.eu/rapid/press-release_IP-14-984_en.htm, Retrieved: June 13th 2015, 18:04.

EC 2014d. European Commission. REGULATION (EU) No 508/2014 OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL of 15 May 2014 on the European Maritime and Fisheries Fund and repealing Council Regulations (EC) No 2328/2003, (EC) No 861/2006, (EC) No 198/2006 and (EC) No 791/2007 and Regulation (EU) No 1255/2011 of the European Parliament and of the Council. Official Journal of the European Union, L 149/1.

EC 2015. European Commission. Aquatic Ressources. http://ec.europa.eu/programmes/horizon2020/en/area/aquatic-resources, Retrieved: April the 6th 2015, 14:21.

Ehler, C., and Douvere, F. 2009. Marine Spatial Planning. *In* A Step-by-Step Approach toward Ecosystem-based Management, p. 99. Ed. by I. O. Commission. Intergovernmental Oceanographic Commission.

FAO. 2013. Applying Spatial Planning for Promoting Future Aquaculture Growth. COFI:AQ/VII/2013/6. www.fao.org/cofi/31155-084a1eedfc71131950f896ba1bfb4c302.pdf.

FAO 2014. Food and Agriculture Organization of the United Nations. The State of World Fisheries and Aquaculture. Rome: 223 pp.

Fock, H. O. 2011. Natura 2000 and the European Common Fisheries Policy. Marine Policy, 35: 181-188.

Fock, H. O., Kloppmann, M., and Stelzenmüller, V. 2011. Linking marine fisheries to environmental objectives: a case study on seafloor integrity under European maritime policies. Environmental Science & Policy, 14: 289-300.

Foden, J., Rogers, S. I., and Jones, A. P. 2011. Human pressures on UK seabed habitats: a cumulative impact assessment. Marine Ecology Progress Series, 428: 33-47.

Foley, M. M., Halpern, B. S., Micheli, F., Armsby, M. H., Caldwell, M. R., Crain, C. M., Prahler, E., et al. 2010. Guiding ecological principles for marine spatial planning. Marine Policy, 34: 955-966.

Gee, K., and Burkhard, B. 2010. Cultural ecosystem services in the context of offshore wind farming: A case study from the west coast of Schleswig-Holstein. Ecological Complexity, 7: 349-358.

Gilliland, P. M., and Laffoley, D. 2008. Key elements and steps in the process of developing ecosystem-based marine spatial planning. Marine Policy, 32: 787– 796.

Godfray, H. C., Beddington, J. R., Crute, I. R., Haddad, L., Lawrence, D., Muir, J. F., Pretty, J., et al. 2010. Food security: the challenge of feeding 9 billion people. Science, 327: 812-818.

Guerry, A. D., Ruckelshaus, M. H., Arkema, K. K., Bernhardt, J. R., Guannel, G., Kim, C.-K., Marsik, M., et al. 2012. Modeling benefits from nature: using ecosystem services to inform coastal and marine spatial planning. International Journal of Biodiversity Science, Ecosystem Services & Management, 8: 107-121.

Halpern, B. S., Longo, C., Hardy, D., McLeod, K. L., Samhouri, J. F., Katona, S. K., Kleisner, K., et al. 2012. An index to assess the health and benefits of the global ocean. Nature, 488: 615-620.

Halpern, B. S., McLeod, K. L., Rosenberg, A. A., and Crowder, L. B. 2008a. Managing for cumulative impacts in ecosystem-based management through ocean zoning. Ocean and Coastal Management, 51: 203-211.

Halpern, B. S., Walbridge, S., Selkoe, K. A., Kappel, C. V., Micheli, F., D'Agrosa, C., Bruno, J. F., et al. 2008b. A global map of human impact on marine ecosystems. Science, 319: 948-952.

Jay, S., and Gee, K. 2014. TPEA Good Practice Guide: Lessons for cross-border MSP from Transboundary Planning in the European Atlantic. ICES Document University of Liverpool, Liverpool, UK.

Katsanevakis, S., Stelzenmüller, V., South, A., Sørensen, T. K., Jones, P. J. S., Kerr, S., Badalamenti, F., et al. 2011. Ecosystem-based marine spatial management: Review of concepts, policies, tools, and critical issues. Ocean & Coastal Management, 54: 807-820.

Kelly, C., Gray, L., Shucksmith, R. J., and Tweddle, J. F. 2014. Investigating options on how to address cumulative impacts in marine spatial planning. Ocean and Coastal Management, 102: 139-148.

Lee, J., and Stelzenmüller, V. 2010. Milestone 9: How different planning tools can be used to prioritise efficient use of marine space.

LS 2005. Lower Saxony. Raumordnungskonzept für das niedersächsische Küstenmeer (ROKK). http://www.raumordnung.niedersachsen.de/portal/live.php?navigation_id=1464&article_id=5311&_psmand=7, Retrieved: June the 17th 2015, 14:31.

McLeod, K. L., Lubchenco, J., Palumbi, S. R., and Rosenberg, A. A. 2005. Scientific Consensus Statement on Marine Ecosystem-Based Management. Signed by 221 academic scientists and policy experts with relevant expertise and published by the Communication Partnership for Science and the Sea. http://www.compassonline.org/sites/all/files/document_files/EBM_Consensus_Statement_v12.pdf, Retrieved: July 22nd 2015, 14:00.

Neori, A., Troell, M., Chopin, T., Yarish, C., Critchley, A., and Buschmann, A. H. 2007. The need for a balanced ecosystem approach to blue revolution aquaculture. Environment, 49: Research Library Core, pg. 37.

Obama, B. H. 2010. Administration of Barack H. Obama. Executive Order 13547: Stewardship of the ocean, our coasts, and the great lakes.

Obama, B. H. 2012. National Ocean Council. Draft National Ocean Policy Implementation Plan. https://www.whitehouse.gov/sites/default/files/microsites/ceq/national_ocean_policy_draft_implementation_plan_01-12-12.pdf, Retrieved: June 3rd 2015, 20:00.

Polasky, S., Nelson, E., Camm, J., Csuti, B., Fackler, P., Lonsdorf, E., Montgomery, C., et al. 2008. Where to put things? Spatial land management to sustain biodiversity and economic returns. Biological Conservation, 141: 1505-1524.

Rijnsdorp, A. D., Eigaard, O. R., Hintzen, N. T., and Engelhard, G. H. 2015. Oceans Past V - The evolution of the impact of bottom-trawling on demersal fish populations and the benthic ecosystem. http://www.ices.dk/news-and-events/news-archive/news/Pages/%E2%80%98The-evolution-of-bottom-trawling-impact-on-demersal-fish-populations-and-the-benthic-ecosystem.aspx, Retrieved: June 18th 2015, 18:45.

Rosenthal, H., Costa-Pierce, B., Krause, G., and Buck, B. 2012. Bremerhaven Declaration on the Future of Global Open Ocean Aquaculture - Part II: Recommendations on Subject Areas and Justifications. Aquaculture Forum on Open Ocean Aquaculture Development - From visions to reality: the future of offshore farming. Funded by: Investment in sustainable fisheries co-financed by the European Union (European Fisheries Fund - EFF), Ministry of Economics, Labour and Ports (Free Hanseatic City of Bremen), The Bremerhaven Economic Development Company Ltd., 8.

Schultz-Zehden, A., and Gee, K. 2013. BaltSeaPlan Findings – Experiences and Lessons from BaltSeaPlan. s.Pro, Berlin, Germany pp.

SH 2003. Schleswig Holstein - Ministry of the Interior. Integrated Coastal Zone Management in Schleswig-Holstein. http://www.schleswig-holstein.de/DE/Fachinhalte/L/landesplanung_raumordnung/Downloads/IKZM-Rahmenkonzept.pdf?__blob=publicationFile&v=1, Retrieved June 17th 2015, 14:30.

Shelton, A. O., Samhouri, J. F., Stier, A. C., and Levin, P. S. 2014. Assessing trade-offs to inform ecosystem-based fisheries management of forage fish. Sci Rep, 4: 7110.

Soma, K., Ramos, J., Bergh, O., Schulze, T., van Oostenbrugge, H., van Duijn, A. P., Kopke, K., et al. 2013. The "mapping out" approach: effectiveness of marine spatial management options in European coastal waters. ICES Journal of Marine Science, 71: 2630-2642.

Stelzenmüller, V., Gimpel, A., Gopnik, M., and Gee, K. in preparation. Aquaculture site-selection and marine spatial planning: The roles of GIS-based tools and models. Richard Langan and Bela H. Buck: Aquaculture perspective of multi-use sites in the open ocean. The untapped potential for marine resources in the anthropocene. Springer book.

Stelzenmüller, V., Lee, J., Garnacho, E., and Rogers, S. I. 2010a. Assessment of a Bayesian Belief Network-GIS framework as a practical tool to support marine planning. Mar Pollut Bull, 60: 1743-1754.

Stelzenmüller, V., Lee, J., South, A., and Rogers, S. I. 2010b. Quantifying cumulative impacts of human pressures on the marine environment: a geospatial modelling framework. Marine Ecology Progress Series, 398: 19-32.

Stelzenmüller, V., Lee, J., South, A., and Rogers, S. I. 2010c. Quantifying cumulative impacts of human pressures on the marine environment: A geospatial modelling framework Marine Ecology Progress Series, 398: 19-32.

Stelzenmüller, V., Schulze, T., Fock, H. O., and Berkenhagen, J. 2011. Integrated modelling tools to support risk-based decision-making in marine spatial management. Marine Ecology Progress Series, 441: 197-212.

Stelzenmüller, V., Schulze, T., Gimpel, A., Bartelings, H., Bello, E., Bergh, O., Bolman, B., et al. 2013. Guidance on a Better Integration of Aquaculture, Fisheries, and other Activities in the Coastal Zone: From tools to practical examples. COEXIST project: 79pp.

Troell, M., Chopin, T., Angel, D., and Buck, B. H. in review. IMTA and offshore aquaculture development. Aquaculture Engineering.

Troell, M., Joyce, A., Chopin, T., Neori, A., Buschmann, A. H., and Fang, J.-G. 2009. Ecological engineering in aquaculture — Potential for integrated multi-trophic aquaculture (IMTA) in marine offshore systems. Aquaculture, 297: 1-9.

UNESCO 2014. United Nations Educational, Scientific and Cultural Organization. Initiative on marine spatial planning. Germany (North/Baltic Seas). http://www.unesco-ioc-marinesp.be/msp_practice/germany_north_baltic_seas, Retrieved: April 5th 2015, 23:20.

Wennhage, H., Pihl, L., and Stål, J. 2007. Distribution and quality of plaice (Pleuronectes platessa) nursery grounds on the Swedish west coast. Journal of Sea Research, 57: 218-229.

Wever, L., Krause, G., and Buck, B. H. 2015. Lessons from stakeholder dialogues on marine aquaculture in offshore wind farms: Perceived potentials, constraints and research gaps. Marine Policy, 51: 251-259.

White, C., Halpern, B. S., and Kappel, C. V. 2012a. Ecosystem service tradeoff analysis reveals the value of marine spatial planning for multiple ocean uses. PNAS, 109: 4696–4701.

White, C., Halpern, B. S., and Kappel, C. V. 2012b. Ecosystem service tradeoff analysis reveals the value of marine spatial planning for multiple ocean uses. Proceedings of the National Academy of Sciences of the United States of America, 109: 4696-4701.

Yamashita, Y., Tanaka, M., and Miller, J. M. 2001. Ecophysiology of juvenile flatfish in nursery grounds. Journal of Sea Research, 45: 205 - 218.

Zamzow, T. 2015. Maritime Spatial Planning in the German EEZ and 12 mile zones of the North Sea. A Roadmap towards investigating coherence in planning and process. Internship Report Thünen Institute for Sea Fisheries, Hamburg - Van Hall Larenstein University, Leeuwarden, the Netherlands.

2. Manuscript 1: A spatially explicit risk approach to support marine spatial planning in the German EEZ

Antje Gimpel[a], Vanessa Stelzenmüller[a], Roland Cormier[b], Jens Floeter[c], Axel Temming[c]

[a] Johann Heinrich von Thünen Institute (TI), Federal Research Institute for Rural Areas, Forestry and Fisheries, Institute of Sea Fisheries, Palmaille 9, 22767 Hamburg, Germany

[b] Department of Fisheries and Oceans Canada, P.O. Box 5030, Moncton, NB, E1C 9B6 Canada

[c] Institute for Hydrobiology and Fisheries Science, University of Hamburg, Olbersweg 24, 22767 Hamburg, Germany

Marine Environmental Research 86 (2013) 56-69

2.1 Abstract

An ecosystem approach to marine spatial planning (MSP) promotes sustainable development by organizing human activities in a geospatial and temporal context. This study develops and tests a spatially explicit risk assessment to support MSP. Using the German exclusive economic zone (EEZ) of the North Sea as a case study area, current and future spatial management scenarios are assessed. Different tools are linked in order to carry out a comprehensive spatial risk assessment of current and future spatial management scenarios for ecologic and economic ecosystem components, i.e. *Pleuronectes platessa* nursery grounds. With the identification of key inputs and outputs the suitability of each tool is tested. Here, the procedure as well as the main findings of the spatially explicit risk approach are summarised to demonstrate the applicability of the framework and the need for an ecosystem approach to risk management techniques using geo-spatial tools.

Key words: effects, generalised additive models (GAM), German EEZ, geographic information system (GIS), marine spatial planning (MSP), nursery grounds, *Pleuronectes platessa*, risk assessment, spatial management scenario

2.2 Introduction

An ecosystem approach to marine spatial management integrates ecological, social, and economic interests (Pomeroy and Douvere, 2008; Foley et al., 2010). The main objectives of such an approach are to maintain ecosystem health and services while informing the decision-making processes in regards to the spatial distribution and management of human activities in the marine environment (Douvere, 2008; Foley et al., 2010). It can be used to identify ecosystem vulnerabilities linked to occurring drivers of human activities and resolve conflicts between social and economic interests while allowing for adaptive management strategies to address changing conditions (Ehler and Douvere, 2009; Foley et al., 2010, Crowder and Norse, 2008). Due to the rapid development of offshore renewable energy such as wind farms (which exclude vessel movement within the wind farm area and in the 500 m-wide marginal buffer zone with the exception of vessels for maintenance or research) and with the implementation of marine protected areas (MPA) under the Natura 2000 protocol, vessel movement can be increasingly curtailed (Berkenhagen, 2009; Fock, 2010; European Commission Environment, 2011a). Such management measures can hamper or limit fishing vessels access to fishing grounds as well as displace or concentrate fishing pressures on the remaining areas not closed to fishing (Stelzenmüller et al., 2011). However, the assessment of these effects requires a sound knowledge base of the complex spatial and temporal relationships between human activities and the sensitivities of the marine environment (Stelzenmüller et al., 2008a). Given the spatial context of these effects, geo-spatial analytical tools are required to adequately assess the relationships and to identify the risks.

This paper uses a risk analysis framework to assess current and future spatial management scenarios. It is applied to identify the risks arising from conflicts of current management scenarios between maritime transportation, aggregate extraction, marine protected areas and fisheries as well as risk arising from the introduction of offshore wind farms in the German EEZ. Furthermore, the framework is also used to assess risks to the nursery grounds of *Pleuronectes platessa* in terms of current fishing pressures and future shift in fisheries pressures once the offshore wind farms are introduced as well as pressures from other human activities affecting the integrity of the benthic habitat. Here, the procedure as well as the main findings of the spatially explicit risk approach are summarised to demonstrate the applicability of the framework.

2.3 Material and Methods

2.3.1 Research strategy

The framework presented in this paper links different tools in order to carry out a comprehensive spatial risk assessment of current and future spatial management scenarios for ecologic and economic ecosystem components. With the identification of key inputs and outputs the suitability of each tool is tested. While assessing current management conditions impacting MSP objectives, it is also important to assess future management scenarios in terms of potential risks affecting both, the ecologic and economic components of our case study area.

2.3.2 Case study specifications

Our study area is located in German waters of the North Sea (Figure 1) that has an overall surface area of 41034 km². The risk assessment focused on the drivers of human activities occurring within the German EEZ of the North Sea waters that has a surface area of 28539 km² (BfN, 2011). The Federal Agency for Shipping and Hydrography (Bundesamt für Seeschifffahrt und Hydrographie, BSH) has the responsibility for MSP in the German EEZ

(Berkenhagen, 2009). The objectives of the German MSP initiatives in the EEZ of the North and Baltic Sea are to integrate sustainable development with marine conservation. The main human activities regulated by the plan are the safety and efficiency of navigation, oil and gas exploitation, renewable energy development as well as aggregate extraction (http://www.bsh.de/de/Meeresnutzung/Raumordnung_in_der_AWZ/Dokumente_05_01_2010/Karte_Nordsee.pdf; BSH, 2009a). It is also noted that the allocation of fishing activities is not spatially managed by the German plans (Fock, 2010).

With the development of offshore renewable energies such as wind farms, fisheries are at risk of losing access to traditional fishing grounds because of the safety requirement imposed by wind farms. As a result, the bottom trawling activities of the demersal fisheries could be displaced and forced to concentrate their activities in smaller areas to maintain their level of catch. Given the risk of benthic habitat impacts due to bottom trawls, the increased concentration of these fisheries could put the integrity of essential fish habitats at risk (Seas at risk, 2006; Europa, 2010). In general, demersal fisheries occurring on the continental shelf of the North Sea (Hiddink et al., 2006) and within the southern North Sea are considered as having a major impact on the seafloor (Fock et al., 2011). Paradoxically, fisheries also depend on the integrity of the seafloor and a healthy ecosystem as promoted by the good environmental status requirements of the Marine Strategy Framework Directive (MSFD; European Commission Environment, 2011b). It is evident that the degradation of the benthic characteristics of essential nursery grounds could lead to a loss of productivity and decreased landings that would subsequently result in further socio-economic impacts through the loss of income.

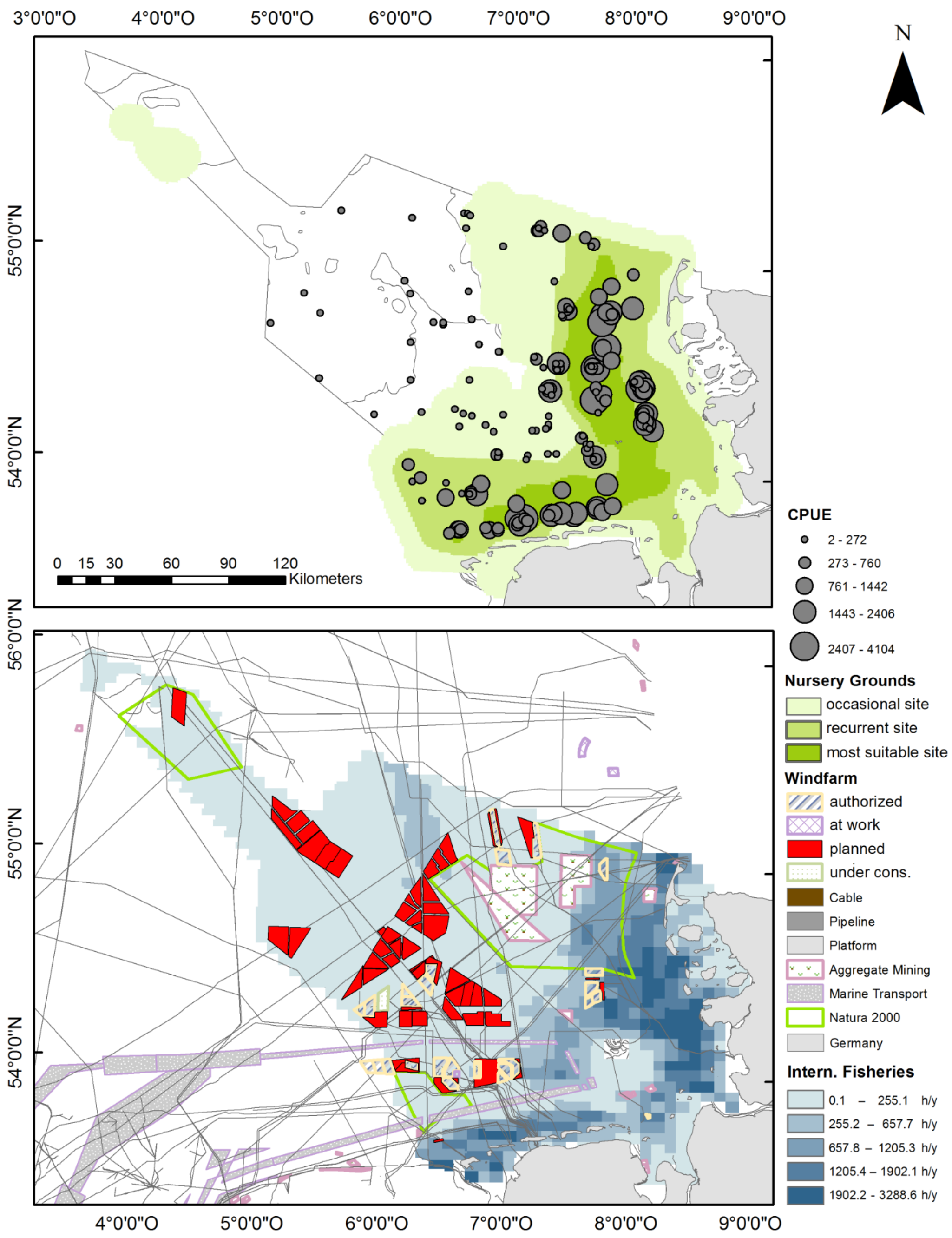


Figure 1: Predicted nursery grounds of plaice: coloured in different green shades, Zone 1 in dark green etc. with GIS-tool "Kernel" (top); human drivers occurring in or surrounding the German EEZ; different colours display each activity considered in this study. The red areas indicate the offshore wind farms planned in future. We implemented also closed areas e.g. nature conservation sites in our study (bottom). We used vector grids with a projected coordinate system ETRS_1989_UTM_Zone_32N_8stellen and the projection Transverse Mercator, the maps are displayed on the scale 1.1:1 (CPUE = Catch per Unit Effort; h/y = hours per year).

Plaice *(Pleuronectes platessa)* is a fish species of high commercial value and is occurring in the North Sea, in the Baltic Sea, in the Skagerrak and the Kattegat. In 2010, the North Sea plaice spawning stock biomass of 522900 t was deemed to be sustainably exploited with landings of approximately 106000 t (57 % landings, 43 % discard) (ICES, 2011a). The most important plaice fishing grounds of the southern North Sea are comprised of the Dogger Bank (Figure 1: the Natura 2000 area in the north) or the northern part of the White Bank area. Plaice is caught using small (< 300 HP) and large (> 300 HP) demersal beam-trawl gears and demersal otter board trawling gears (Fock, 2008). Given that the plaice landings of German fisheries amounted to approximately 3912.7 t in 2010, (BLE, Bundesamt für Landwirtschaft und Ernährung, 2010), plaice can be considered as a significant ecosystem goods and service of the North Sea ecosystem (Duffy, 2006; Silvestri and Kershaw, 2010). Therefore, the preservation and availability of nursery grounds is a crucial factor in maintaining the productivity of this ecosystem goods and service (Wennhage et al., 2007) since plaice recruitment depends on adequate habitat characteristics and abundances. In addition to fisheries, other human activities also exert a number of pressures on the coastal and marine environment (Halpern et al., 2008), which can be additive, synergistic or antagonistic (Stelzenmüller et al., 2010). These pressures also have a physical impact on the seabed and are likely to have adverse effects on the integrity of the nursery grounds (Hiddink et al., 2006; Foden et al., 2010).

2.3.3 Description of risk analysis framework

The intent of a risk analysis is to ensure that decision-making processes are adequately informed in terms of the risks when considering management options designed to eliminate, control or mitigate the risks (US EPA, 2008; Cormier et al., in press). In this case study, we subdivided the risk analysis into risk identification, risk characterization, risk assessment and risk management (Figure 2). The risk analysis framework also includes risk communication as a key principle to ensure transparency and quality of the planning and management processes. As we did not focus on the outcomes of the management process itself but on the applicability of the framework, we did not include this aspect in the case study.

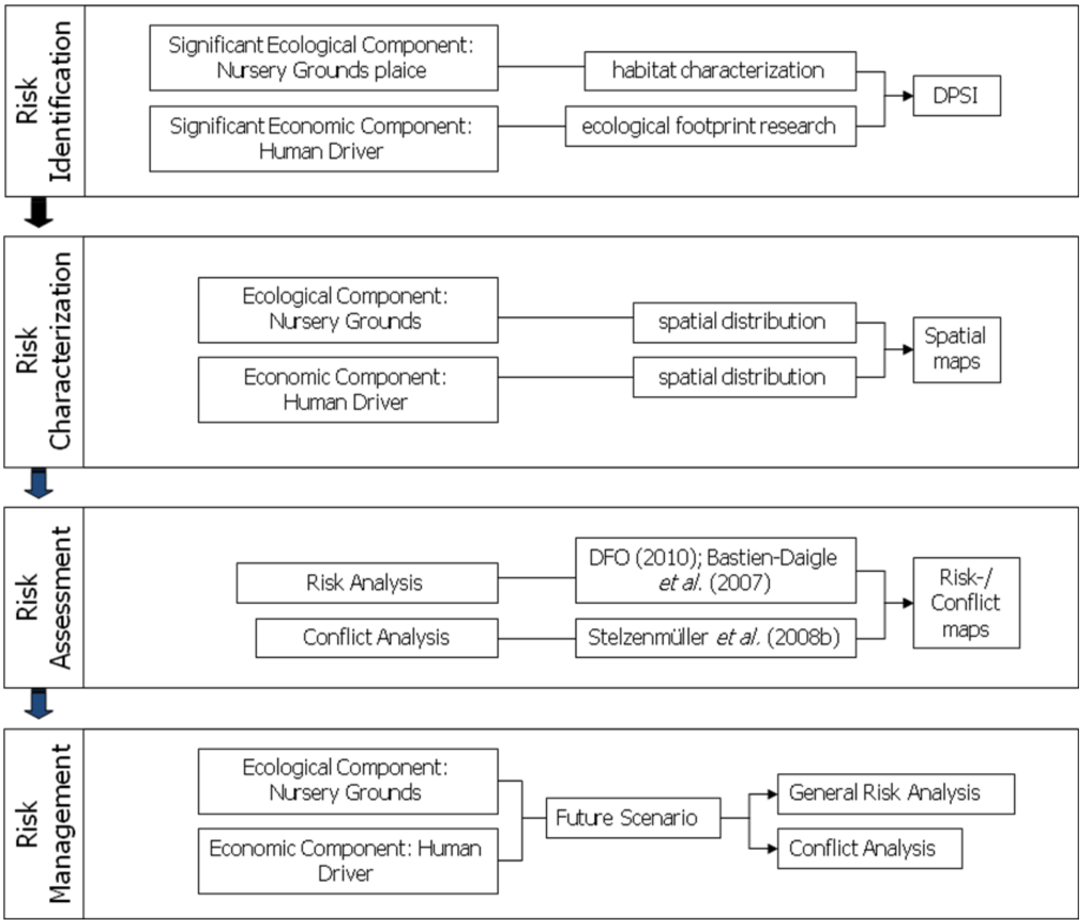


Figure 2: Adapted flow diagram outlining the concept of the Ecosystem-based Risk Management Framework (Cormier et al., in press); adapted to structure our study. The framework comprises six essential steps: risk identification (1), risk characterization (2), risk assessment (3), management options, risk management (4) and an integrated management plan, while we implemented only the first three steps as well as the risk management to structure this study. The four boxes denote these steps (following the thick arrows) we used to assess the effects emerging due to spatial overlaps of both human drivers as well as ecosystem components such as the nursery grounds of plaice (following the thin lines and arrows).

We considered the North Sea as the ecological unit and boundary for the case study. With the help of abiotic and biotic factors, we characterised the most suitable habitat for juvenile plaice to define the nursery grounds of plaice and spatially delineate this ecological component for the assessment. Additionally, we identified the major human activities considered in the analysis within the German EEZ zone of influence. This should be assumed to be a simplification of reality. Below, we describe in detail the step-wise implementation of the risk assessment framework (Figure 2).

2.3.3.1 Risk Identification

A comprehensive habitat characterization for juvenile plaice was conducted (Table 1) to delineate the plaice nursery grounds and to assess sensitivities to pressures caused by drivers of human activities. The characterization was based on a literature research regarding the local conditions in our study area (i.e. sediment composition) and key environmental variables (e.g. temperature). To establish the magnitude and intensity of the pressures related to the human activities, we generated a knowledge base of the benthic ecological footprints for these drivers occurring in the German EEZ (Table 1). The generation of the knowledge base relied on the literature search related to the spatial prediction of the ecological footprints (Figure 1) and resulted in a compilation of geo-referenced data for power and telecommunication cables, pipelines, oil and gas industries, offshore wind farms, shipping, international demersal fishing activities and sediment extraction. These drivers were then categorised into generic pressures under the themes of abrasion, obstruction, extraction, siltation, contamination, smothering and alteration. Pressure categories were used to normalise the data to facilitate the assessment of driver intensities and to allow comparison between different spatial locations (Stelzenmüller et al., 2010). In addition, we used a DPSI (Driver-Pressure-State-Impact) conceptual model and definitions (Figure 3) to illustrate the pathways of effects showing the links between drivers of human activities (Driver) and their respective normalised pressures (Pressure) occurring in the study area (Elliot, 2002; UNEP/GRID-Arendal, 2002). These were then used to predict where the pressures have the potential to cause effects that could change the integrity (State) of the plaice nursery grounds. Finally, the impact (Impact) leading from this changed state was evaluated by accounting for the spatial overlap and sensitivity of the plaice nursery grounds to the pressures.

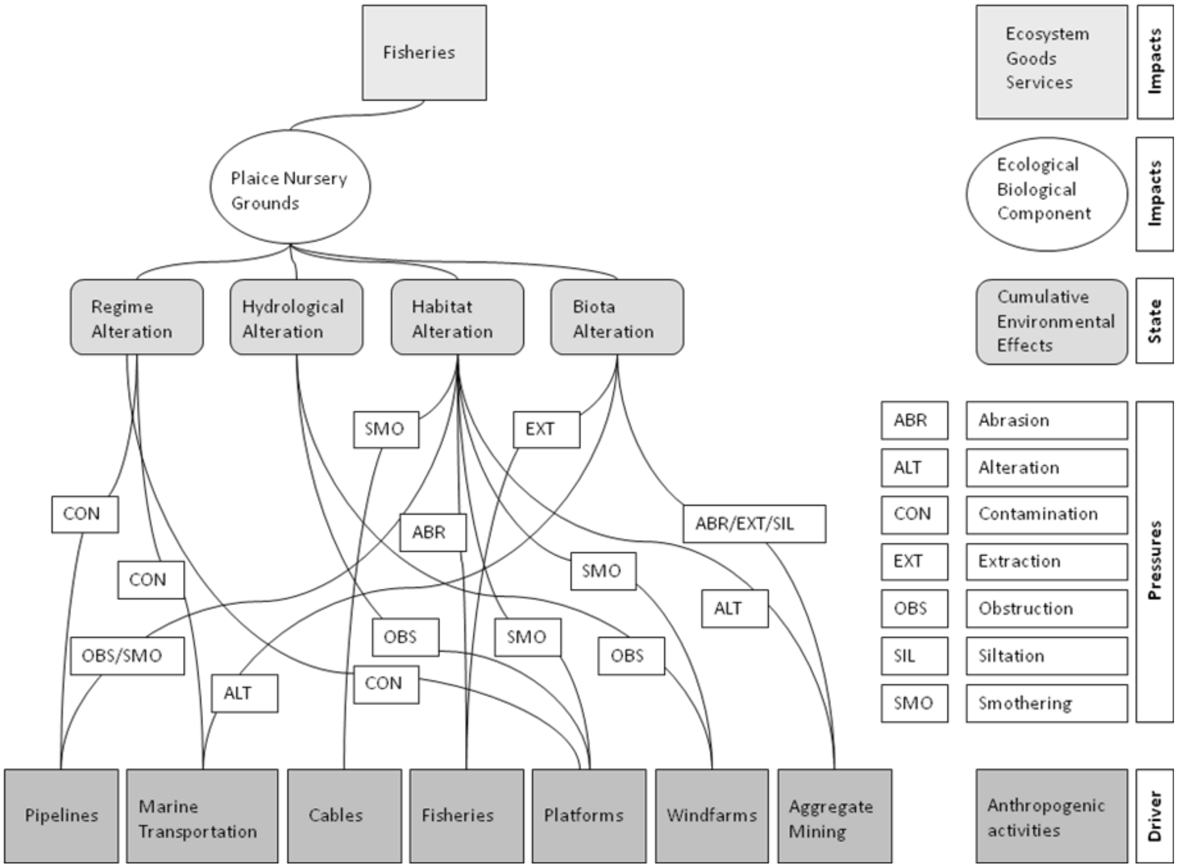


Figure 3: DPSI-model illustrating the effect pathways: The boxes below represent the human driver occurring in the German EEZ applying pressure (displayed in uncoloured small boxes) on the status of the environment, which is described in the squared corners rounded shape boxes. These specific changes impact on ecological components as well as ecosystem goods and services (symbolized with the round box and the squared one above) and the ecosystem, respectively.

2.3.3.2 Risk Characterization

For the risk characterization, we extracted from the ICES Datras data-base catch data of plaice (http://datras.ices.dk/Home/Descriptions.aspx#top) from Dutch Beam Trawl Surveys (2000-2008, 3rd quarter of each year), covering the entire German EEZ. The data included Catch per Unit Effort (CPUE), depth (m), average- (2000-2008) bottom salinity and temperature interpolated with ordinary kriging (Stelzenmüller et al., 2011). Furthermore, we used the occurrence of benthic communities predicted by Pesch et al. (2008) and sediment data provided by the BSH, which were classified after Tauber and Lemke (1995). From the Dutch surveys, we plotted the plaice log transformed CPUE data against the length class to determine the breaking points of the different age classes. To identify the juvenile classes, we only considered individuals smaller than a total length of 150 mm, leaving a data set of 112989 small plaice caught in 9 years out of the earliest life-stages “larvae, early 0-group, late 0-group and I-group”. Taking into account the relationship between the spatial distribution of juvenile plaice and important environmental variables like depth, temperature, salinity, sediment composition and benthos communities, we computed a species distribution model (SDM) using a generalised additive model (GAM) (Hastie and Tibshirani, 1986; Florin et al., 2009; Cotté et al., 2010). We considered all variables for model calibration after testing them with the Spearman Correlation Analysis (Florin et al. 2009; Cotté et al., 2010). For the GAM calculations, we allowed for possible non-linear effects of the environmental variables using natural splines (Venables and Dichmont, 2004) while controlling the risk of over-fitting by limiting the degrees of freedom. From the full set of calculated GAMs we selected the best models by the lowest value of Akaike Information Criterion (AIC: Akaike 1973). The significantly positive partial effect ranges of GAM covariates of the selected model were chosen to define optimal nursery habitat properties (Figure 4). Using these criteria, we developed a GIS raster layer indicating the potential habitat for juvenile plaice.

Subsequently, we buffered the geo data representing the spatial distribution of the drivers regarding their footprint. International demersal fishing activities were mapped by processing VMS data of 2008 following the approach of Fock (2008) and described in detail in Stelzenmüller et al. (2011). To assess the risks of spatial overlaps of the drivers and pressures occurring in our study area, the geo-layers were generated for each of the pressures (Figure 3) using GIS.

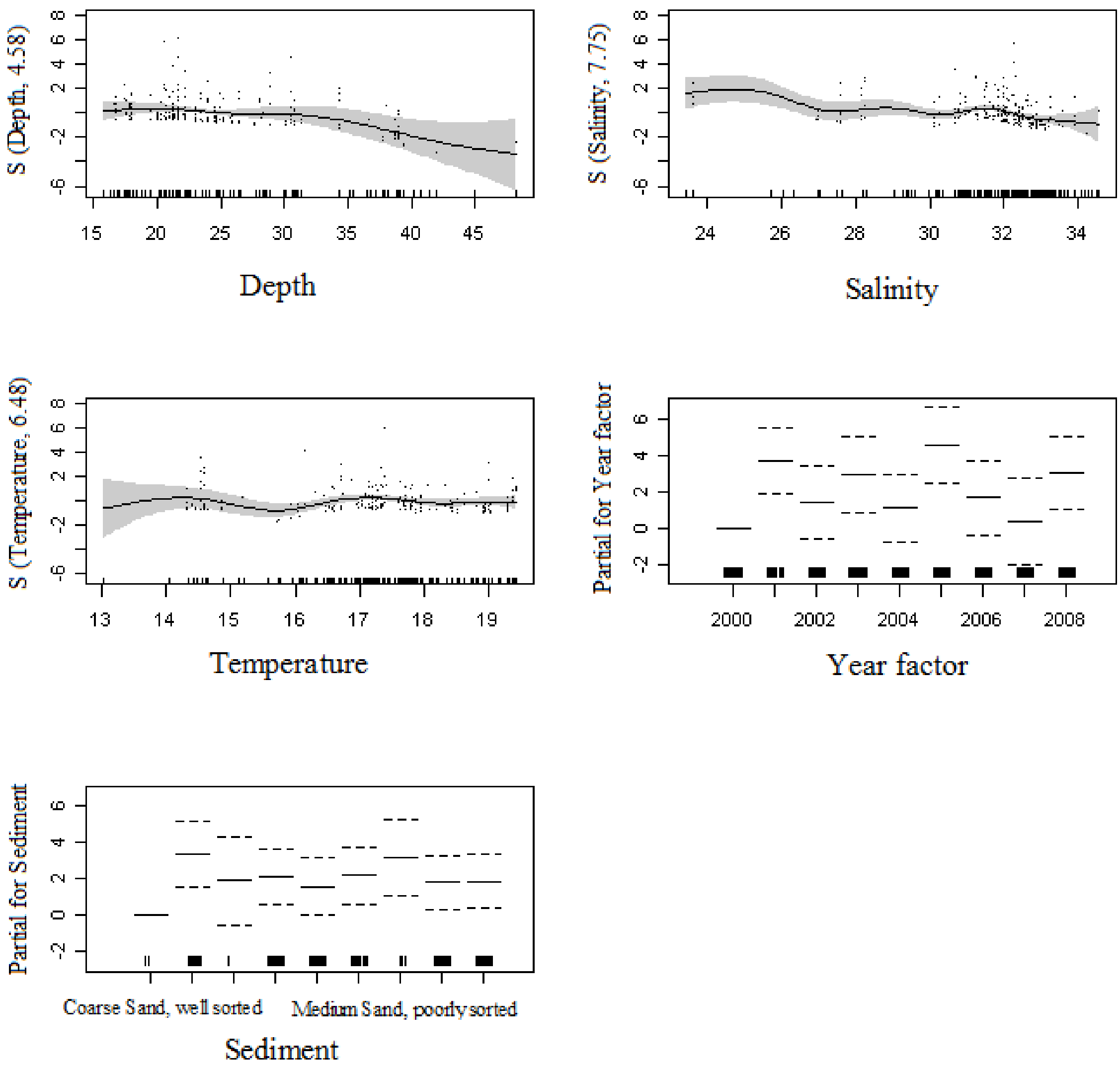


Figure 4: Functional forms of the waiting distance for the smoothed covariates (depth, salinity, temperature, year factor and sediment composition) included in the generalized additive model from the data used in this study (s = smoothed).

2.3.3.3 Risk Assessment

The negative effects of the pressures on the nursery grounds as well as the respective sensitivity of the plaice nursery grounds were assessed using attributes to describe the scale of impact. Figure 5 shows the magnitude of the impact as a result of the spatial prediction. We used the Habitat Management Qualitative Risk Assessment developed by Bastien-Daigle et al. (2007), to ascertain the magnitude and geographic extent of the effects, the duration and the frequency of the effect as well as its reversibility. We adapted the assessment to our defined nursery grounds and evaluated each generic pressure individually. The sensitivity of the nursery grounds, the dependency of plaice on the nursery grounds and the rarity and resiliency of these nursery grounds were estimated using the same framework. The positive effects of the drivers such as the creation of new habitats or shelters for prey species (e.g. construction of reefs around the wind mills pods) were omitted as a simplification of reality. Thus, the sensitivity of the nursery grounds was evaluated using risk scores between "low" = 1; and "high" = 3. The effect caused by the general pressures was assessed using the same classification of risk scores (low = 1, medium = 2, high = 3) (Table 1). To combine the pressure and sensitivity attributes, we calculated a potential level of impact by adding the pressure and sensitivity scores. The maximum level of impact would therefore have a score of 27 (100 %) if every pressure and sensitivity would be assessed as high.

We evaluated the general conflict potential between the drivers using a conflict matrix (Table 2) developed by Stelzenmüller et al. (2008b). The conflict categories range from 0 = "no conflict"; to 5 = "mutually exclusive". The scoring of conflicts was followed by a spatial application of the conflict matrix to produce conflict maps. In a subsequent step, the spatial distribution of human drivers in the study areas and the potential conflicts between them were mapped using GIS (Figure 1 and 6). We also considered closed areas as e.g.

nature conservation sites. An assessment of the adequate grid resolution accounting for the spatial resolution of the available data and computation time at the scale of the study area revealed a grid size resolution of 5 km^2.

2.3.3.4 Risk Management

To assess the potential effects of future spatial management options in the study area, we developed multiple risk scenarios based on the identification of potential conflict areas between drivers as well as between the pressures and the nursery grounds. The risk and conflict analysis considered current drivers and future offshore wind farm development as planned for 2025 (Burkhard et al., 2011) (Figure 1).

2.4 Results

2.4.1 Risk Identification

High quality nursery grounds provide the juvenile plaice with a tidal stream transport, sandy sediments and shelter, which reduce predation rates (Rijnsdorp et al., 1985; Wennhage et al., 2007; Florin et al., 2009). The tidal stream transport constitutes a passive but selective transport by swimming up from the seabed during flood tides and remaining on the seabed during ebb tides, which is used by juvenile plaice to feed on sandy flats and move back to deeper waters on the ebb tide (Rijnsdorp et al., 1985; ICES, 2011b).

Table 1: Attributes used to describe the scale of (negative) effects (due to human pressure) to nursery grounds (NG) and to define the sensitivity of NG, risk scores (severity and duration of risk; 1 = lowest, 3 = highest); modified after Bastien-Daigle et al. (2007).

	Pressure		
	Low (1)	**Medium (2)**	**High (3)**
Magnitude	localized effect on NG, returns to pre-pressure levels in one generation or less, within natural variation	portion of the NG returns to pre-pressure levels in one generation or less, rapid and unpredictable change, temporarily outside range of natural variability	affecting the whole NG, outside the range of natural variation, such that the NG do not return to pre-pressure levels for multiple generations
Geographic Extent	limited to pressure footprint and vicinity	limited to pressure vicinity	extends beyond the pressure area
Duration of Effect	less than one season	one season to one year	more than one year
Frequency of Effects	occurs on a monthly basis or less frequently	occurs on a weekly basis	occurs on a daily basis or more frequently
Reversibility	effects are reversible over short term without active management	effects are reversible over short term with active management	effects are reversible over extended term with active management or effects are irreversible
	Sensitivity		
Sensitivity	NG not sensitive to change and perturbation	NG moderately sensitive to change and perturbation	NG highly sensitive to change and perturbation
Dependence	not used as habitat; or used as migratory habitat only	used as feeding, rearing, and/or spawning habitat	nursery grounds critical to survival of species
Rarity	NG is abundant within its range or community; ecological redundancy is widely present	NG has limited distribution; is confined to small areas; ecological redundancy is present	NG is rare; ecological redundancy is absent
Resiliency	NG is stable and resilient to change and perturbation	NG is stable and can sustain moderate level of change and perturbation	NG is unstable and not resilient to change and perturbation

Table 2: Matrix of potential conflicts developed by Stelzenmüller et al. (2008b); No conflict = 0; Mutually exclusive = 5. The scoring of conflicts was followed by a spatial application using GIS to produce conflict maps (with regard to the footprint of the individual drivers) with a grid size resolution of 5 km².

	Pelagic trawling	Demersal trawling	Fishing fixed gears	Offshore wind farm	Platforms (oil, gas)	Cables	Pipelines	Sediment extraction	Shipping	Closed areas
Pelagic trawling	x	0	5	5	5	3	3	2	2	5
Demersal trawling	0	x	5	5	5	3	3	3	2	5
Fishing fixed gears	5	5	x	2	5	2	2	1	2	5
Offshore wind farm	5	5	2	x	5	2	3	5	5	5
Platforms (oil, gas)*	5	5	5	5	x	1	2	5	5	5
Cables	3	3	2	2	1	x	4	5	0	3
Pipelines	3	3	2	3	2	4	x	5	0	3
Sediment extraction	2	3	1	5	5	5	5	x	2	5
Shipping	2	2	2	5	5	0	0	2	x	4
Closed areas	5	5	5	5	5	3	3	5	4	x

According to Wennhage (2007), the North Sea is at the centre of the distribution range of plaice with the Wadden Sea as the most important nursery ground. The uniqueness of nursery grounds for plaice is high as determined by multiple factors such as temperature, salinity, depth, food or dissolved oxygen (Yamashita et al., 2001). In general, the nursery grounds have a relatively structure-less bottom relief in shallow soft bottom areas (Lerda, 2005). The water depths of our defined I-group nursery grounds lie between 20 metres in the southwest and 30 metres in the north (BSH, 2009a), the salinity between 24 PSU in the southwest and 33 PSU in the north and the average bottom temperature during the 3rd quarter of the year between 17.8 °C in the southwest and 16.8 °C in the north. The water attributes provide key conditions for the plaice juveniles as dynamic systems, aggregated nutrients as well as prey to growth. *Pleuronectidae*, such as plaice, are visual day-feeders, which mainly prey on polychaetes, crustaceans (copepods were numerically the most important prey and also the most frequent item in the stomachs) and molluscs (Amara et al., 2001; Freitas et al., 2010).

The sensitivity of the nursery grounds is mostly related to disturbance of the sediment structure that plaice need for their life-history. Drivers of human activities such as fisheries and sediment extraction generate pressures that may change the natural sediment structure of the nursery grounds and, thus, the recruitment potential of plaice. Ground dragnets destroy the softer seabed up to several centimetres in addition to potential reduction of grain size. The southern North Sea, which borders the Wadden Sea, and outlines nursery grounds for plaice, is a high intensity area of fishing activities. Further, substrate removal and alteration of the seabed topography is caused by aggregate mining (BSH, 2009a; Silvestri and Kershaw, 2010).

We assumed that power and telecommunication cables influence their surrounding area by smothering about 0.09 m, oil and gas platforms about 15 m by contamination, smothering and obstruction (Eastwood et al., 2007). International demersal fishing activities in the study area affect their surrounding area by extraction and abrasion. Pipelines influence their surrounding area by obstruction, smothering and contamination about 0.76 to 1.37 m (Stelzenmüller et al., 2008a). Sediment extraction affects its surrounding area by siltation, abrasion, extraction and alteration about 50 m (Foden et al., 2010). The footprints of offshore wind farms, which influence the area where they occur by obstruction and smothering, are not well studied (Burkhard et al., 2011; OSPAR, 2010). Therefore, we only considered their spatial extent.

2.4.2 Risk Characterization

The best GAM predictor variables are found to be depth (edf = 4.583, $Chi^2 = 3.290$, p = 0.004148), salinity (edf = 7.749, $Chi^2 = 3.241$, p = 0.000945) and temperature (edf = 6.475, $Chi^2 = 2.312$, p = 0.021045) with an overall deviance explained of 29.6 %. The partial effect plots indicates that there is a positive relationship between increased CPUE values and a depth between 22 and 27 m, salinity between 24 and 33 PSU and temperature between 16.8 and 17.8°C (Figure 4).

Using these optimal environmental criteria, we extracted a GIS raster layer from each environmental predictor. It should be noted, however, that we used a GIS depth layer between 20 and 30 m. Thus, we generated a map indicating the most suitable habitat for juvenile plaice comprised of a total area of 17939.12 km^2.

Based on our environmental criteria, we classified the nursery grounds in three zones being (1) the “most suitable site”, (2) the “recurrent site” and (3) the “occasional site”. Each zone is imbedded in the other such that zone 1 is a sub-area of zone 2 and zone 2 is a sub-area of zone 3. Overall, the predicted nursery ground zones cover 98.92 % of juvenile plaice abundances. Zone 2 as the “recurrent site” covers with 56.40 % the highest fraction of juvenile plaice abundances, followed by zone 1, the “most suitable site”, which contains 33

%. Zone 3 covers 9.52 % of the juvenile plaice abundances as the "occasional site" (Figure 1). An example of geo-spatial overlaps between the predicted plaice nursery grounds and human pressures is illustrated in Figure 5. All values are given in Table 3 and 4.

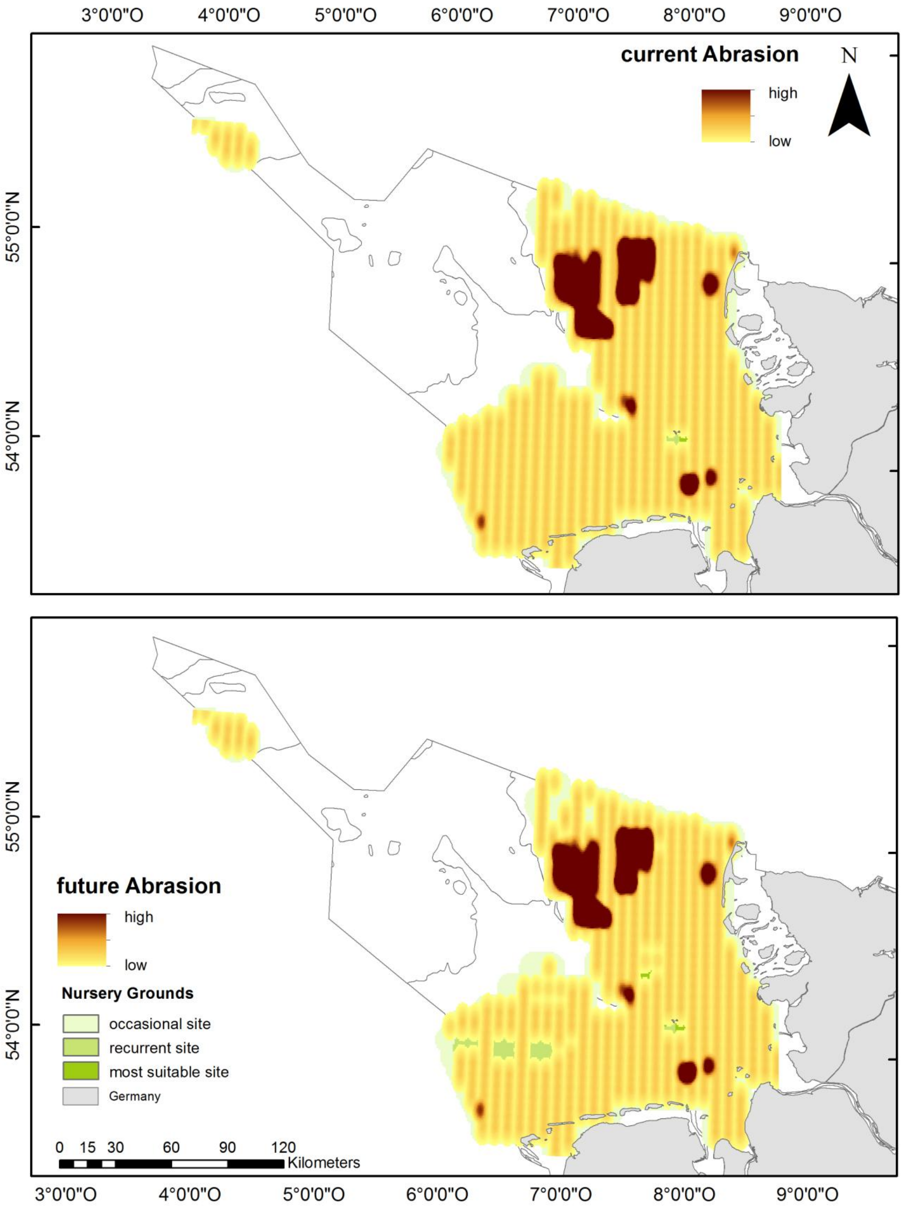


Figure 5: Example abrasion: nursery grounds of plaice, overlaid by all drivers relating to abrasion. (In the GIS we merged drivers exerting the same pressure on the marine environment into a single pressure layer) (top); example future abrasion: nursery grounds of plaice, overlaid by all drivers relating to abrasion (bottom).

In contrast, spatial overlaps of the human drivers i.e. conflicts are illustrated in Figure 6 and are described in section 2.4.3 below.

2.4.3 Risk Assessment

Results show that there are spatial overlaps between pressures occurring in the study area and the nursery grounds with abrasion and extraction affecting 100 % of the habitat (Figure 5). The cumulative aspect of both pressures could lead to a loss of the nursery grounds habitat characteristics and subsequently affect the recruitment potential of juvenile plaice. Only 2.66 % of the contamination and alteration pressures overlap with the nursery ground area. The results of the evaluation regarding the potential negative effects of human pressures on the nursery grounds are summarised in Table 3. From a temporal aspect, we considered these pressures as temporal and spatial constants for the duration of this study.

To combine the pressure and sensitivity attributes, we calculated a potential level of impact indicator that took into account the scale of the impacts for the various pressures and the sensitivity of the nursery grounds. The highest impact scores (21) are reached by abrasion and extraction. Given that siltation results in an impact score of 19 and smothering in an impact score of 20, these impacts are considered as high. Obstruction and contamination are also considered as high, with an impact score of 18 while alteration has an impact score of 16 and is therefore assessed as medium (Table 5).

Table 3: Pressure evaluation: Potential effects used, attributes used to assess the risk level, risk scores and literature cited (severity and duration of risk; 1 = lowest, 3 = highest); Table 1 and 2.

	Magnitude	Geographic Extent	Duration of Effect	Frequency of Effects	Reversibility	References
Abrasion	3 (localized pressure affecting 17939.12 km² and therefore 100 % of the sediment composition)	3 (wide sediment plumes of 200 to 500 m beyond the area the pressure takes place; declines in biodiversity and abundance)	3 (28 month to several years)	1 (not affected more often than five times per year)	3 (alteration of the seabed topography, a change in the sediment structure as well as damage to the bottom-dwelling communities)	Ban and Alder, 2008; Foden *et al.*, 2011; Desprez, 2000; Scharf *et al.*, 2006; Silvestri and Kershaw, 2010; OSPAR, 2010; HELCOM, 2010a, 2010b; BSH, 2009a, 2009b
Extraction	3 (localized pressure affecting 17939.12 km² and therefore 100 % of the sediment composition)	3 (wide sediment plumes of 200 to 500 m beyond the area the pressure takes place; declines in biodiversity and abundance; bycatch)	3 (28 month to several years)	1 (not affected more often than five times per year)	3 (alteration of the seabed topography, a change in the sediment structure as well as damage to the bottom-dwelling communities)	Ban and Alder, 2008; Foden *et al.*, 2011; Desprez, 2000; Scharf *et al.*, 2006; Silvestri and Kershaw, 2010; OSPAR, 2010; HELCOM, 2010a, 2010b; BSH, 2009a, 2009b
Siltation	1 (localized effect on nursery grounds: 1393.84 km², thus 7.77 %)	3 (sediment plume of aggregate mining)	3 (28 month to several years)	1 (not affected more often than five times per year)	3 (physical damage to the seabed topography)	Ban and Alder, 2008; Foden *et al.*, 2011; Desprez, 2000; Silvestri and Kershaw, 2010; OSPAR, 2010; HELCOM, 2010; BSH, 2009a, 2009b
Smothering	3 (covering with 563.26 km² about 3.13 %)	1 (limited to pressure footprint; electromagnetic fields, heat)	3 (altered by favouring opportunistic species for the next years)	3 (affected constantly)	3 (local hydrographic conditions also affect the recovery time of the site decline of biodiversity and abundance of species)	Foden *et al.*, 2011; OSPAR, 2010; HELCOM, 2010a, 2010b; BSH, 2009a, 2009b

	Magnitude	Geographic Extent	Duration of Effect	Frequency of Effects	Reversibility	References
Obstruction	1 (562.62 km² and therefore 3.14 % of the sediment composition affected)	1 (localized effects within its footprint area; destroyed seabed, altered seafloor and the eventually resulting changes in local hydrography)	3 (several years)	3 (affected constantly)	3 (alteration of the seabed topography or increased turbidity; disturbance and loss of habitats)	Foden *et al.*, 2011; OSPAR, 2010; HELCOM, 2010a, 2010b; BSH, 2009a, 2009b
Contamination	1 (477.3 km², thus 2.66 %)	2 (noise; discharge of effluent; a high concentration of ships using the same route)	3 (several years)	2 (high concentration of ships using the same route)	3 (residual contamination based on emissions + accidents causing wider risks)	Ban and Alder, 2008; OSPAR, 2010; HELCOM, 2010a, 2010b; BSH, 2009a, 2009b
Alteration	1 (476.91 km² and therefore 2.66 %)	2 (ballast water = invasive species = affect the nursery grounds at a broader scale than its footprint; noise of shipping, the pollution by oil and hazardous or toxic substances and the discharge and disposal of wastes)	2 (biological disturbances)	3 (high concentration of ships using the same routes)	1 (a stable ecosystem would return to pre-pressure level after short time)	Ban and Alder, 2008; OSPAR, 2010; HELCOM, 2010a, 2010b; BSH, 2009a, 2009b

Table 4: Sensitivity evaluation: Potential influence of effects used, attributes used to assess the sensitivity, risk scores and literature cited (severity and duration of risk; 1 = least, 3 = greatest); Table 1 and 2.

	Sensitivity	Dependence	Rarity	Resiliency	References
Abrasion	3 highly sensitive to change and perturbation (declines in abundance, sediment plumes, 28 month to several years alteration of the seabed topography, a change in the sediment structure)	3 (determining the abundance of plaice; critical to survival of species)	1 (17939.12 km²; abundant)	1 (stable and resilient to change and perturbation)	Ban and Alder, 2008; Desprez, 2000; Florin *et al*., 2009; Foden *et al*., 2011; HELCOM, 2010a, 2010b; BSH, 2009a, 2009b; OSPAR, 2010; Scharf *et al*., 2006; Wennhage *et al*., 2007
Extraction	3 highly sensitive to change and perturbation (declines in abundance, sediment plumes, 28 month to several years alteration of the seabed topography, a change in the sediment structure)	3 (determining the abundance of plaice; critical to survival of species)	1 (17939.12 km²; abundant)	1 (stable and resilient to change and perturbation)	Ban and Alder, 2008; Desprez, 2000; Florin *et al*., 2009; Foden *et al*., 2011; HELCOM, 2010a, 2010b; BSH, 2009a, 2009b; OSPAR, 2010; Scharf *et al*., 2006; Wennhage *et al*., 2007
Siltation	3 highly sensitive to change and perturbation (sediment plumes, 28 month to several years alteration of the seabed topography, a change in the sediment structure)	3 (determining the abundance of plaice; critical to survival of species)	1 (17939.12 km²; abundant)	1 (stable and resilient to change and perturbation)	Ban and Alder, 2008; Desprez, 2000; Florin *et al*., 2009; Foden *et al*., 2011; HELCOM, 2010a, 2010b; BSH, 2009a, 2009b; OSPAR, 2010; Wennhage *et al*., 2007
Smothering	2 moderately sensitive to change and perturbation (electromagnetic fields, heat, altered by favouring opportunistic species for the next years, affected constantly, decline in abundance of species)	3 (determining the abundance of plaice; critical to survival of species)	1 (17939.12 km²; abundant)	1 (stable and resilient to change and perturbation)	Florin *et al*., 2009; Foden *et al*., 2011; HELCOM, 2010a, 2010b; BSH, 2009a, 2009b; OSPAR, 2010; Wennhage *et al*., 2007

	Sensitivity	Dependence	Rarity	Resiliency	References
Obstruction	2 moderately sensitive to change and perturbation (alteration of the seabed topography, affected constantly, disturbance and loss of habitats)	3 (determining the abundance of plaice; critical to survival of species)	1 (17939.12 km²; abundant)	1 (stable and resilient to change and perturbation)	Florin *et al*., 2009; Foden *et al*., 2011; HELCOM, 2010a, 2010b; BSH, 2009a, 2009b; OSPAR, 2010; Wennhage *et al*., 2007
Contamination	2 moderately sensitive to change and perturbation (discharge of effluent, residual contamination based on emissions)	3 (determining the abundance of plaice; critical to survival of species)	1 (17939.12 km²; abundant)	1 (stable and resilient to change and perturbation)	Ban and Alder, 2008; Florin *et al*., 2009; HELCOM, 2010a, 2010b; BSH, 2009a, 2009b; OSPAR, 2010; Wennhage *et al*., 2007
Alteration	2 moderately sensitive to change and perturbation (biological disturbances by invasive species, pollution by oil and hazardous or toxic substances and the discharge and disposal of wastes)	3 (determining the abundance of plaice; critical to survival of species)	1 (17939.12 km²; abundant)	1 (stable and resilient to change and perturbation)	Ban and Alder, 2008; Florin *et al*., 2009; HELCOM, 2010a, 2010b; BSH, 2009a, 2009b; OSPAR, 2010; Wennhage *et al*., 2007

The highest conflict potential between human drivers occurs when closed nature conservation sites overlap with fisheries, aggregate mining or wind farms. Therefore, 8802.75 km² (19.74 %) of the study area is affected by the conflict level "mutually exclusive" (Figure 6, top). An area of 1813.21 km² (4.07 %) is highlighting a very high likelihood of conflict. With a high likelihood of conflict an area of 541.76 km² (1.2 %) is affected, an area of 0.000031 km² is highlighting a medium likelihood of conflict. A low likelihood of conflict is affecting an area of 0.05 km². The absence of conflict is only demonstrated by the combination of shipping and cables or shipping and pipelines (Figure 6).

	Level of Impact	Negligible or low Impact	Low Impact	Medium Impact	High Impact	Very High Impact	
Magnitude of Impact	95 – 100%						Evaluated Impact Score
	60 – 95%				ABR / CON / EXT / OBS / SMO / SIL		
	35 – 60%			ALT			
	5 – 35%						
	0 – 5%						

Table 5: Evaluated impact induced by general pressures, combined with sensitivity of the nursery grounds*. Each variable used to describe the effect of the pressure in terms of their risks on the nursery grounds gains an impact score which ranges from 1 = "low"; to 3 = "high". Subsequently, the scores have been summarized.

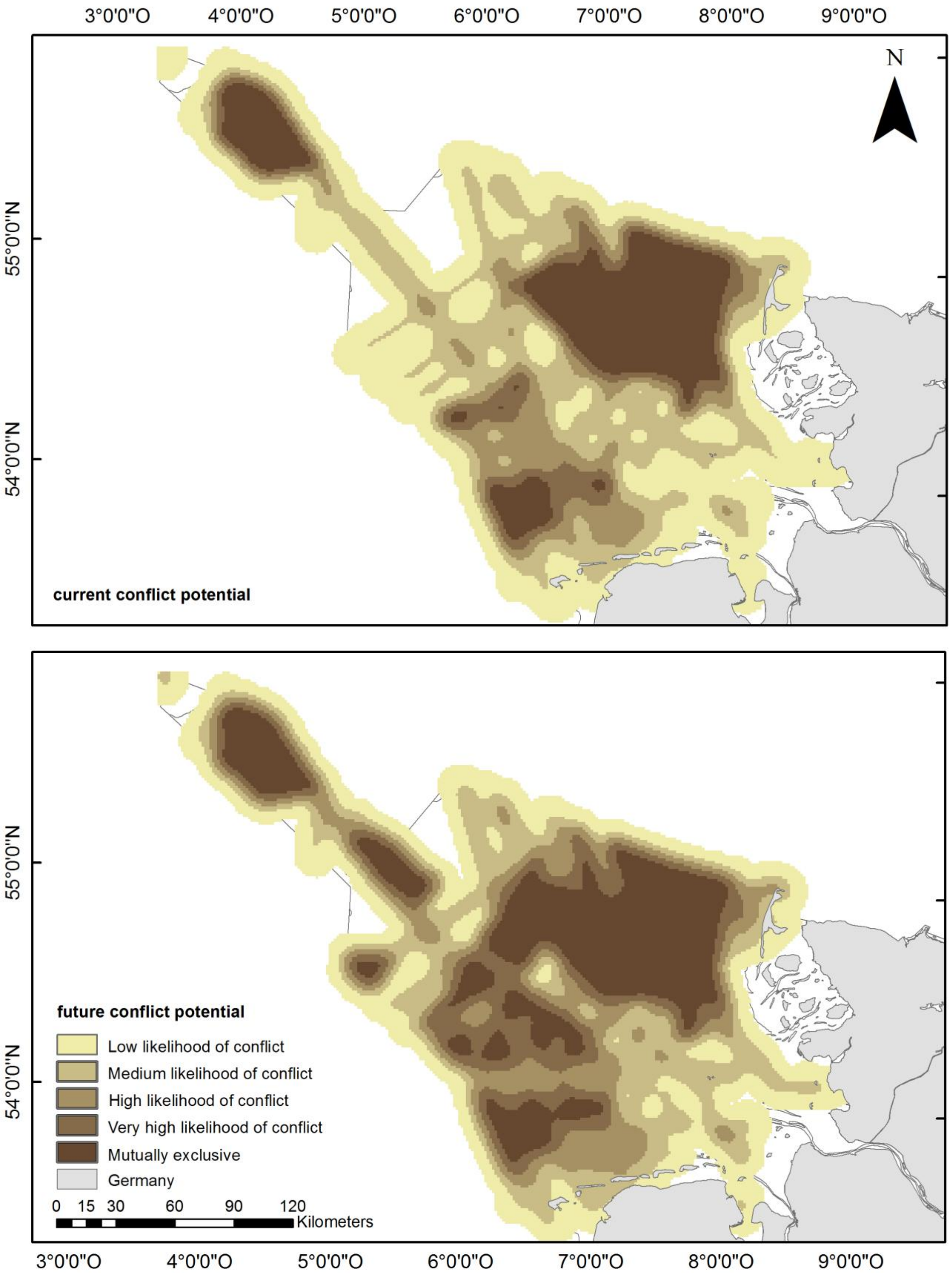


Figure 6: Conflict map: interpolated potential conflict areas scored from 0 = no conflict (coloured in light brown) to 5 = mutually exclusive (coloured in dark brown) (top); future conflict map: potential conflict areas scored from 0 = no conflict to 5 = mutually exclusive. A conflict score of 5 lays above all the other scores (bottom). The potential conflict maps using a mean grid size resolution of 5 km² have been interpolated using the Kernel tool.

2.4.4 Risk Management

For the risk management, we added the future plans for offshore wind farms and repeated the previously described risk and conflict analyses (Table 6 and Figure 1). There is a potential overlap with the nursery grounds and consequently a predicted potential impact. For the most suitable site for juvenile plaice (zone 1), we predict an overlap of 3.73 % with wind farm areas and deem this area as most vulnerable. Zone 3 has an overlap of 0.87 % and is the least affected by wind farms. Adding the future wind farms to our generic pressure list, the area where abrasion occur, decreases and the area where smothering and obstruction occur, increases (Table 6 and Figure 3):

Smothering received a high magnitude of pressure (1251.34 km² = 6.98 %) affecting the nursery grounds (0.88 % in zone 1; 2.37 % in zone 2; 3.73 % in zone 3). Obstruction received a medium magnitude result (1250.86 km² = 6.97 %) affecting the sediment composition of the nursery grounds (0.88 % in zone 1; 2.36 % in zone 2; 3.73 % in zone 3). As vessel traffic is prohibited in the areas of the wind farms, the abrasion pressure, induced by demersal fisheries, decreases. Using this scaling of effects, abrasion received a high magnitude of localised pressure (15350.36 km² = 85.57 %) affecting the nursery grounds (15.13 % in zone 1; 58.51 % in zone 2; 85.57 % in zone 3).

The calculated risk scores show that future abrasion decreases in magnitude of impact but remains at a “high risk of impact” level. Future smothering and obstruction increase or respectively double in magnitude. Although the risk score for obstruction increases by a factor of one, both pressures still remain “high”. The existing wind farm area, which is 675.9 km², is not just doubled; it even increases by 122.33 % due to the planned wind farms on the nursery grounds surface (Figure 1).

The highest conflict potential between human drivers after implementing the planned offshore wind farms occurs between wind farming and fisheries. The whole area where demersal fisheries take place is 38779.49 km². Combining the existing and the planned offshore wind farms, an area of 4525.64 km² would be covered with offshore wind farms. Consequently, 3595.10 km² of the area where demersal fisheries occur would be displaced in a potentially new conflict situation with the driver wind farm, leading to a potential loss of 9.27 % of the fishing grounds. Additionally, the conflict potential of wind farming in combination with areas closed for nature protection would also increase (Table 6, Figure 6).

The area with low likelihood of conflict (0.05 km²), with medium likelihood of conflict (0.000031 km²), with high likelihood of conflict (541.76 km²) as well as the areas with very high conflict potential (1813.21 km²; 4.07 %) in our future scenario analysis remain the same. The highest conflict potential between human drivers, occurring in combination with closed areas, fisheries, aggregate mining or wind farming, increases by 50 percent and affects 13356.04 km² (29.94 %) for "mutually exclusive" conflict levels (Table 6 and Figure 6).

2.5 Discussion

We applied a spatially explicit risk analysis framework to assess both potential impacts of major human pressures on predicted plaice nursery grounds and risks of future spatial management scenarios on nursery grounds and human activities. This was accomplished by combining the risk management framework developed by Cormier et al. (in press) with spatial analysis tools such as basic mapping using a GIS and spatial distribution modeling using GAMs. It must be emphasised that the purpose of this study is to demonstrate the suitability of the risk management approach in combination with generic tools and methods.

Table 6: Outcome for the future scenario, divided in the risk analysis (top) and conflict analysis (bottom). The red boxes denote an increase of the different components, the blue boxes a decrease of the different components and the white boxes stand for no change for the different components.

		Risk potential (Nursery grounds)
Drivers	Pipelines	→
	Marine Transportation	→
	Cables	↑
	Fisheries	↓
	Platforms	→
	Wind farms	↑
	Aggregate Mining	→

Pressures	Abrasion	↓
	Alteration	→
	Contamination	→
	Extraction	↓
	Obstruction	↑
	Siltation	→
	Smothering	↑

States	Regime Alteration	↓
	Hydrological Alteration	↑
	Habitat Alteration	↑
	Biota Alteration	↓

		Conflict potential (Wind farms)
Human activities	Pipelines	↑
	Marine Transportation	→
	Cables	↑
	Fisheries	↑
	Platforms	→
	Nature Conservation	↑
	Aggregate Mining	→

The risk assessment framework (Cormier et al., in press) was helpful in structuring both, the risk analysis and the conflict analysis:

Performing an initial risk identification ensured a comprehensive knowledge base concerning potential impacts affecting both, the plaice nursery grounds and the human activities. As a proxy of the ecosystem, we considered in our analysis solely human activities occurring within the German EEZ. Taking e.g. the whole Wadden Sea into account, might have been consistent towards a comprehensive ecosystem approach but would also have been associated with numerous conservation laws (national level down to state level) as well as additional different tourism and aquaculture activities which have had to be considered. Therefore, we kept the case study definitions simple. The assessment of the driver footprints revealed that the information regarding the wind farms is lacking. Using the DPSI-approach proved to be very useful for defining the general pressures because it helped to structure the links between drivers, the pressures and the effects that may impact the nursery grounds (e.g. ecosystem component). In addition, it helped visualise the pathway between the drivers that could potentially impact the nursery grounds. Here, we only considered the most important pressures in relation to their spatial extent. As we focused on drivers which are located offshore, nutrient and organic matter enrichment due to input of fertilizers or organic matter and marine litter were not included.

The basic mapping of ecosystem components within the risk characterization facilitated the identification of potential risks and conflicts, although the intensity of the general pressures is not reflected within this study as the available data were limited. The VMS data represented only the variability of one year (2008), it can be assumed that the average fishing effort distribution is representative of the typical fisheries intensity in the area (Stelzenmüller et al., 2008a). It has to be mentioned, that the habitat features of the nursery grounds of plaice represented just a part of the aspects of ecosystem structure. Ecosystem-based risk assessment would also include a representation of different trophic level species (Samhouri and Levin, 2012).

During the risk assessment, the habitat management qualitative risk assessment developed by Bastien-Daigle et al. (2007) appeared to be useful to characterise and estimate the exposure of the effect due to human pressures. It also provided the basis for assessing the sensitivity, rarity and resiliency of the nursery grounds. Since we considered the regional effects of localised human activities, it reflected the risk potential of present regional management actions. For our analysis, we considered all drivers and pressures to be temporarily and spatially constant for a time period of one year. Furthermore, we assessed the level of risk and impact only spatially and to a limited extent although the scale of impact varies with different pressures and with the sensitivity of the nursery grounds to these pressures. As mentioned in section 2.3, the positive effects of the drivers were not considered within the findings of the paper (cf. human "pressures"), even though the inclusion of positive effects could neutralise the impacts of the pressures in this study. Given the typical increase in productivity and biodiversity for several meters around the windmills, it might be assumed that offshore wind farms are highly attractive to juvenile plaice (Meißner and Sordyl, 2006). Including such positive effects would change the conclusions regarding the characterisation of the impacts found in our study. Consequently, further investigations would be needed to identify the net effects given our risk approach regarding negative impacts instead of overall outcomes that consider both the impacts and benefits of introducing wind farms. Given that the nursery grounds are associated to soft bottom areas, their vulnerability is affected mainly by fisheries or dredging, respectively (Fock et al., 2008; BSH 2009a; BSH, 2009b). The framework by Bastien-Daigle et al. (2007) focused on nine different attributes for evaluating the general pressures and their effects. However, there are several additional indirect effects caused by human drivers such as an increased turbidity, caused by obstruction, which could affect larval stages through reduced feeding efficiency (Florin et al., 2009). In turn, slow growth rates of juvenile plaice would affect the recruitment. Another indirect effect is caused by the demersal fisheries, which lead to larger by catch of non-target fish species (HELCOM, 2010b) as for juvenile plaice. The conflict matrix by Stelzenmüller et al. (2008b), which was used within the risk assessment to generate the potential conflict maps, highlighted five different conflict combinations. Possible co-existences (such as wind farms and aquaculture), were not identified and need further investigations. The areas of potential conflicts highlighted in this study represent a wide range of overlapping drivers acting in the German EEZ. Whereas nature conservation sites overlapping with fisheries resulted in the conflict level "mutually exclusive", circa 20 % of the whole area was affected by this level (Stelzenmüller et al., 2008b). As the nature conservation sites may contain habitats and species which are not sensitive to fisheries, wind farming or sediment extraction, detailed studies would be required to take these factors into account as well as the duration of activities.

Our future spatial management scenarios took into account existing offshore wind farm development objectives for 2025. As a direct effect for the plaice nursery grounds, the drivers for smothering and obstruction increased as a result of the wind farm development objectives. However, as vessel traffic is prohibited in wind farm areas in addition to 500 m-wide marginal buffer zones (Berkenhagen, 2009), abrasion and extraction decreased from a magnitude of 100 % to 85.57 %. It can therefore be expected, that the direct risks for plaice are on a limited scale. However, as an indirect effect, areas of fisheries would be displaced to other grounds since the wind farm areas are inaccessible (Fock, 2010; Stelzenmüller et al., 2011). In need for new areas, this driver would increase its pressure for abrasion and extraction towards the most sensitive juvenile plaice nursery grounds. In the future, comprehensive studies that would include all direct and indirect effects as well as positive effects on the nursery grounds would be needed. Regarding our conflict analysis, the future development of offshore wind farms could lead to a strong increase of the conflict potential “mutually exclusive”. Results reveal that the area where wind farms would exist will likely more than double:

In the entire German EEZ, an area of 4525.64 km² will be allocated to wind farms, which will lead to a loss of 9.27 % of potential fishing grounds. As already mentioned, the spatial allocation of fishing activities is currently not managed by the German plans. Thus, the future risks of offshore wind farm development are not clear as it will result in a loss of fishing grounds access. In addition, 31.5 % of the EEZ are covered by designated Natura 2000 sites (Fock, 2010) resulting in further bottom trawling over sensitive benthic habitats because of the displaced fisheries (Fock et al., 2011). Since profitable fishing grounds in the North Sea are relatively stationary for numerous of fish species, fisheries may need to change their target species. This might be possible for larger fishing fleets but not for individual fishermen or small fishing associations given the costs involved. As a potential loss of fishing grounds could lead to an increased competition and conflicts, catch rates will likely decrease and individual fishermen as well as small fishing associations will suffer economically (Berkenhagen, 2009).

Results have shown that more research is required on the development of objective evaluation methods regarding the interaction and conflict levels between human activities. However, the risk assessment framework allowed the identification of potential indirect effects related to the spatial management options (e.g. the level of geo-spatial analysis of risks highlighted the displacement of the trawling intensity over sensitive plaice nursery grounds). In summary, the risk approach can be used to inform the decision making processes in support of an ecosystem-based MSP, even though the ecosystem-based risk framework used in this study is not a tool in itself:

It helps to structure and simplify the integration of spatially explicit tools and consequently, facilitates risk and conflict analyses of spatial management options. Spatially explicit tools such as GIS facilitate the visualization of current and future conditions. The SDM is useful in describing the relationship of the distribution of an organism to its environment. Combining these tools on the base of mapping could assist in identifying areas of conservation potential, reducing and solving conflicts and thus supporting the implementation of an ecosystem-based management approach.

2.6 Conclusion

In conclusion it can be said, that there are many different tools assessing the risk; there is no single tool that fits all. The spatial risk approach is a tangible framework toward ecosystem-based management and the reduction of conflicts among user groups. The value for marine spatial planning is not just in the development of the framework but also in the process, as the framework is not providing any more quality of outputs than what the quality is of the inputs. Done well, the risk assessment process can help to examine future risks and clarify potential conflicts by involving future management scenarios while demonstrating the need for an ecosystem approach to risk management techniques using geo-spatial tools.

In Germany, offshore MSP requires an integrated assessment process considering all ecosystem functions and the potential impacts of the direct, indirect and combined effects of human drivers. This would include a comprehensive analysis defining principal areas for all vessels operating in a given planning area (Fock, 2008; Stelzenmüller et al., 2011). Such an integrated assessment is also promoted by the MSFD obliging EU member states to achieve GES by 2020. In turn, this requires member states to conduct an initial assessment of the current state of the marine environment by 2012 and to develop a strategy for the assessment of the GES by 2018. A crucial part of the strategy will include the implementation of management measures to achieve GES. The combined alignment of

MSP and GES management strategies should be considered in future planning processes. In addition, a coherent planning and assessment system that integrate coastal (under the jurisdiction of the Federal States) and offshore areas should be considered.

2.7 Acknowledgements

We would like to thank the whole team at Fisheries and Oceans Canada. They provided us with very valuable advice on the application of the risk assessment framework.

Furthermore, we would like to thank the University of Hamburg, TI SF and ICES for the provision of funding for this study. The data were provided by the TI SF and BSH, Hamburg, Germany in raw, uninterpreted form.

2.8 References

Akaike H. 1973. Information theory and an extension of the maximum likelihood principle. Second International Symposium on Information Theory. Akadémiai Kiadó. Budapest. 267-281.

Amara RA, Laffargue P, Dewarumez JM, Maryniak C, Lagardère F, Luczac C. 2001. Feeding ecology and growth of O-group flatfish (sole, dab and plaice) on a nursery ground (Southern Bight of the North Sea). *Journal of Fish Biology* 58: 788–803.

Ban N, Alder J. 2008. How wild is the ocean? Assessing the intensity of anthropogenic marine activities in British Columbia, Canada. *Aquatic Conservation: Marine and Freshwater Ecosystems* 18: 55–85.

Bastien-Daigle S, Hardy M, Robichaud G. 2007. Habitat Management Qualitative Risk Assessment: Water Column Oyster Aquaculture in New Brunswick. Canadian Technical Report of Fisheries and Aquatic Sciences 2728.

Berkenhagen J, Döring R, Fock HO, Kloppmann MHF, Pedersen SA, Schulze T. 2009. Decision bias in marine spatial planning of offshore wind farms: Problems of singular versus cumulative assessments of economic impacts on fisheries. *Marine Policy* 34: 733-736.

BfN, Bundesamt für Naturschutz. 2011. Schutzgebiete Übersicht und Kurzfakten. http://www.bfn.de/habitatmare/de/schutzgebiete-uebersicht.php [09 November 2011]

BLE, Bundesamt für Landwirtschaft und Ernährung. 2010. http://www.ble.de/SharedDocs/Downloads/02_Kontrolle_Zulassung/04_Fischerei/01_Fischwirtschaft/Anlandestatistik2010.pdf?__blob=publicationFile [01 December 2011]

BSH, Bundesamt für Seeschifffahrt und Hydrographie. 2009. Non technical summary (North Sea). http://www.bsh.de/en/Marine_uses/Spatial_Planning_in_the_German_EEZ/documents2/Report-NorthSea.pdf [12 October 2012]

BSH, Bundesamt für Seeschifffahrt und Hydrographie. 2009. Spatial Plan for the German Exclusive Economic Zone in the North Sea - Text section. http://www.bsh.de/en/Marine_uses/Spatial_Planning_in_the_German_EEZ/documents2/Report-NorthSea.pdf [18 August 2011]

Burkhard B, Opitz S, Lenhart H, Ahrendt K, Garthe S, Mendel B, Windhorst W. 2011. Ecosystem based modelling and indication of ecological integrity in the German North Sea - Case study offshore wind parks. *Ecological Indicators* 11: 168–174.

Cormier R et al. In press. Marine and Coastal Ecosystem-Based Risk Management Handbook. ICES Cooperative Research Report.

Cotté C, Guinet C, Taupier-Letage I, Petiau E. 2010. Habitat use and abundance of striped dolphins in the western Mediterranean Sea prior to the morbillivirus epizootic resurgence. *Endangered Species Research* 12: 203-214.

Crowder L, Norse E. 2008. Essential ecological insights for marine ecosystem-based management and marine spatial planning. *Marine Policy* 32: 772-778.

Desprez M. 2000. Physical and biological impact of marine aggregate extraction along the French coast of the Eastern English Channel: short- and long-term post-dredging restoration. *ICES Journal of Marine Science* 57: 1428–1438.

DFO, Department of Fisheries and Oceans Canada. 2010. Species at risk act, Risk based listing framework, Tool Guidelines, final draft_V2.

Duffy JE. 2006. Marine ecosystem services. http://www.eoearth.org/article/Marine_ecosystem_services [18 August 2011]

Douvere, F. 2008. The importance of marine spatial planning in advancing ecosystem-based sea use management. *Marine Policy* 32: 762 – 771.

Eastwood PD, Mills CM, Aldridge JN, Houghton CA, Rogers SI. 2007. Human activities in UK offshore waters: an assessment of direct, physical pressure on the seabed. *ICES Journal of Marine Science* 64: 453–463.

Ehler C, Douvere F. 2009. Marine Spatial Planning, A Step-by-Step Approach toward Ecosystem-based Management, Intergovernmental Oceanographic Commission, Manual and Guides No. 53, ICAM Dossier No. 6.

Elliot M. 2002. The Role of the DPSIR approach and conceptual models in marine environment management: an example for offshore wind power. *Marine Pollution Bulletin* 44: iii–vii.

Europa. 2010. Europe's seas: Commission sets out criteria for good environmental status. http://europa.eu/rapid/pressReleasesAction.do?reference=IP/10/1084&format=HTML&aged=1&language=DE&guiLanguage=de [18 August 2011]

European Commission Environment. 2011. http://ec.europa.eu/environment/nature/natura2000/index_en.htm [01 December 2011]

European Commission Envrionment. 2011. http://ec.europa.eu/environment/water/marine/ges.htm [01 December 2011]

Florin AB, Sundblad G, Bergström U. 2009. Characterization of juvenile flatfish habitats in the Baltic Sea. *Estuarine Coastal and Shelf Science* 82: 294-300.

Fock HO. 2008. Fisheries in the context of marine spatial planning: Defining principal areas for fisheries in the German EEZ. *Marine Policy* 32: 728–739.

Fock HO. 2010. Natura 2000 and the European Common Fisheries Policy. *Marine Policy* 35: 181–188.

Fock HO, Kloppmann M, Stelzenmueller V. 2011. Linking marine fisheries to environmental objectives: a case study on seafloor integrity under European maritime policies. *Environmental Science and Policy* 14: 289-300.

Foden J, Rogers SI, Jones AP. 2010: Recovery of UK seabed habitats from benthic fishing and aggregate extraction - towards a cumulative impact assessment. *Marine Ecology Progress Series* 411: 259-270.

Foden J, Rogers SI, Jones AP. 2011. Human pressures on UK seabed habitats: a cumulative impact assessment. *Marine Ecology Progress Series* 428: 33–47.

Foley MM., Halpern BS, Micheli F, Armsby MH, Caldwell MR, Crain CM, Prahler E, Rohr N, Sivas D, Beck MW et al. 2010. Guiding ecological principles for marine spatial planning. *Marine Policy* 34: 955-966.

Freitas V, Campos J, Skreslet S, van der Veer HW. 2010. Habitat quality of a subarctic nursery ground for 0-group plaice (Pleuronectes platessa L.). *Journal of Sea Research* 64: 26–33.

Halpern BS, Walbridge S, Selkoe KA, Kappel CV, Micheli F, D'Agrosa C, Bruno JF, Casey KS, Ebert C, Fox HE et al. 2008. A Global Map of Human Impact on Marine Ecosystems. *Science* 319: 948-952.

Hastie T, Tibshirani R. 1986. Generalised additive models. *Statistical Science* 1: 297-318.

HELCOM. 2010. Ecosystem Health of the Baltic Sea 2003–2007: HELCOM Initial Holistic Assessment. Balt. Sea Environ. Proc. No. 122.

HELCOM. 2010. Towards a tool for quantifying anthropogenic pressures and potential impacts on the Baltic Sea marine environment: A background document on the method, data and testing of the Baltic Sea Pressure and Impact Indices. Balt. Sea Environ. Proc. No. 125.

Hiddink JG, Jennings S, Kaiser MJ. 2006. Indicators of the Ecological Impact of Bottom-Trawl Disturbance on Seabed Communities. *Ecosystems* 9: 1190–1199.

ICES, International Council for the Exploration of the Sea. 2011. Advice June 2011, Book 6. http://www.ices.dk/committe/acom/comwork/report/2011/2011/ple-nsea.pdf [16 August 2011]

ICES, International Council for the Exploration of the Sea. 2011. http://www.ices.dk/marineworld/fishmap/ices/pdf/plaice.pdf [1 December 2012]

Lerda S. 2005. Economic valuation of coastal habitats sustaining plaice fisheries. http://users.ictp.it/~eee/workshops/smr1684/a_lerda.pdf [09 November 2011]

Meißner K, Sordyl H. 2006. Literature Review of Offshore Wind Farms with Regard to Benthic Communities and Habitats. In: Ecological Research on Offshore Wind Farms: International Exchange of Experiences - Part B: Literature Review of Ecological Impacts. *BfN-Skripten* 186: 1-47.

OSPAR. 2010. Quality Status Report 2010, OSPAR Commission, London, 176 pp.

Pesch R, Pehlke H, Jerosch K, Schröder W, Schlüter M. 2008. Using Decision Trees to Predict Benthic Communities within and near the German Exclusive Economic

Zone (EEZ) of the North Sea. *Environmental Monitoring and Assessment* 136: 313-325.

Pomeroy R, Douvere F, 2008. The engagement of stakeholders in the marine spatial planning process. *Marine Policy* 32: 816–822.

Rijnsdorp AD, van Stralen M, van der Veer HW. 1985. Selective Tidal Transport of North Sea Plaice Larvae Pleuronectes platessa in Coastal Nursery Areas. *Transactions of the American Fisheries Society* 114: 461-470.

Samhouri JF, Levin PS. 2012. Linking land- and sea-based activities to risk in coastal ecosystems. *Biological Conservation* 145: 118–129.

Scharf FS, Manderson JP, Fabrizio MC. 2006. The effects of seafloor habitat complexity on survival of juvenile fishes: Species-specific interactions with structural refuge. *Journal of Experimental Marine Biology and Ecology* 335: 167–176.

Seas at Risk. 2006. Defining Good Environmental Status in the context of the European Marine Strategy Directive - What constitutes a healthy marine environment? http://www.seas-at-risk.org/1mages/NGO%20paper%20on%20Good%20Environmental%20Status%20%28FINAL%29_2.pdf [18 August 2011]

Silvestri S, Kershaw F. 2010. Framing the flow: Innovative Approaches to Understand, Protect and Value Ecosystem Services across Linked Habitats, UNEP World, Conservation Monitoring Centre, Cambridge, UK.

Stelzenmüller V, Ellis JR, Rogers SI. 2008. Towards a spatially explicit risk assessment for marine management: Assessing the vulnerability of fish to aggregate extraction. *Biological Conservation* 143: 230–238.

Stelzenmüller V, Lee J, Rogers SI. 2008. Report on milestone 7: Step by step demonstration of the capability of the planning tools using a realistic marine planning scenario. Cefas, project A1420 on MSP tools, 25 pp.

Stelzenmüller V, Lee J, South A, Rogers SI. 2010. Quantifying cumulative impacts of human pressures on the marine environment: A geospatial modelling framework. *Marine Ecology Progress Series* 398: 19-32.

Stelzenmüller V, Schulze T, Fock HO, Berkenhagen J. 2011. Integrated modelling tools to support risk based decision making in marine spatial management. *Marine Ecology Progress Series* 441: 197-212.

Tauber F, Lemke W. 1995. Map of sediment distribution in the Western Baltic Sea (1: 100,000), Sheet "Darß". *Ocean Dynamics* 47: 171-178.

UNEP/GRID-Arendal. 2002. DPSIR framework for state of Environment Reporting, Maps and Graphics Library. http://maps.grida.no/go/graphic/dpsir_framework_for_state_of_environment_reporting [01 December 2011]

US EPA. 2008. Guidelines for Assessing Regional Vulnerabilities, EPA/600/R-08/078. http://www.epa.gov/reva/docs/guidelines_reva_20080627.pdf [01 December 2011]

Venables WN, Dichmont CM. 2004. A generalised linear model for catch allocation: an example from Australia's Northern Prawn Fishery. *Fisheries Research* 70: 409-426.

Wennhage H, Pihl L, Stal J. 2007. Distribution and quality of plaice (Pleuronectes platessa) nursery grounds on the Swedish west coast. *Journal of Sea Research* 57: 218-229.

Yamashita Y, Tanaka M, Miller JM. 2001. Ecophysiology of juvenile flatfish in nursery grounds. *Journal of Sea Research* 45: 205-218.

3. Manuscript 2: A GIS modelling framework to evaluate marine spatial planning scenarios: Co-location of offshore wind farms and aquaculture in the German EEZ

Antje Gimpel[a], Vanessa Stelzenmüller[a], Britta Grote[b], Bela H. Buck[b], Jens Floeter[c], Ismael Núñez-Riboni[a], Bernadette Pogoda[b], Axel Temming[c],

[a] Thünen Institute (TI), Federal Research Institute for Rural Areas, Forestry and Fisheries, Institute of Sea Fisheries, Palmaille 9, 22767 Hamburg, Germany

[b] Alfred Wegener Institute Helmholtz Centre for Polar and Marine Research (AWI), Bussestrasse 27, 27570 Bremerhaven, Germany

[c] Institute for Hydrobiology and Fisheries Science, University of Hamburg, Olbersweg 24, 22767 Hamburg, Germany

Marine Policy 55 (2015) 102-115

3.1 Abstract

The concept of co-location of marine areas receives an increased significance in the light of sustainable development in the already heavily used offshore marine realm. Within this study, different spatial co-location scenarios for the coupling of offshore aquacultures and wind farms are evaluated in order to support efficient and sustainable marine spatial management strategies. A Geographic Information System (GIS) and Multi-Criteria Evaluation (MCE) techniques were combined to index suitable co-sites in the German exclusive economic zone of the North Sea. The MCE was based on criteria such as temperature, salinity or oxygen. In total, 13 possible aquaculture candidates (seaweed, bivalves, fish and crustaceans) were selected for the scenario configuration. The GIS modelling framework proved to be powerful in defining potential co-location sites. The aquaculture candidate oarweed (*Laminaria digitata*) revealed the highest suitability scores at 10 to 20 m depth from April to June, followed by haddock (*Melanogrammus aeglefinus*) at 20 to 30 m depth and dulse (*Palmaria palmata*) and Sea belt (*Saccharina latissima*) at 0 to 10 m depth between April and June. In summary, results showed several wind farms were de facto suitable sites for aquaculture since they exhibited high suitability scores for Integrated Multi-Trophic Aquaculture (IMTA) systems combining fish species, bivalves and seaweeds. The present results illustrate how synergies may be realised between competing needs of both offshore wind energy and offshore IMTA in the German EEZ of the North Sea. This might offer guidance to stakeholders and assist decision-makers in determining the most suitable sites for pilot projects using IMTA techniques.

Keywords: Aquaculture, Co-location, GIS, Marine Spatial Planning, Offshore Site Selection, Wind farms

3.2 Introduction

Given the heavy exploitation of wild fish stocks in combination with an increasing demand for aquatic products, offshore aquaculture production may contribute to food security and relief some of the pressures on wild stocks. However, to deliver on these promises and secure production well into the future, further attention needs to be paid to the increasing requirements for water resources as well as market demands, logistics and technical developments (Godfray et al., 2010; FAO, 2014). Aquaculture poses a conflict potential in combination with other (traditional) activities such as fisheries or tourism by competing on space (Christie et al., 2014). With an increase of designated areas for offshore wind development, planned until 2025, the race for space will gain more importance in offshore and coastal waters of the German North Sea. Fisheries are at risk of losing access to traditional fishing grounds due to the safety requirements imposed by wind farm development, leading to potentially decreased landings (Gimpel et al., 2013). Competition for maritime space and the need for sustainable food production highlight the importance of efficient adaptive management, to avoid potential conflicts as well as create synergies between different activities (Stelzenmüller et al., 2013; Soma et al., 2013; Godfray et al., 2010; Rosenthal et al., 2012). Considering the recent European Maritime Spatial Planning (MSP) Directive, the implementation of Blue Growth, a long term strategy promoted by MSP to support sustainable growth in the marine environment, is required by 2020 (EC, 2014a). In the light of Good Environmental Status (GES) requirements of the Marine Strategy Framework Directive (MSFD), different uses made of the marine resources should be conducted at a sustainable level, individually as well as cumulatively (EC, 2014b). Therefore, the concept of co-location (also referred to as co-use or multi-use (Grote and Buck, 2014)) of marine offshore areas currently receives increased significance (Buck et al., 2004). The possibility of co-location depends on site specific characteristics and adaptive management (Christie et al., 2014; Kaiser et al., 2010). In this field, case studies are essential to explore co-location-options, like (Benassai et al., 2014) for offshore aquaculture in combination with wind farms in Danish waters or by (Buck et al., 2004) as well as (Buck and Krause, 2012) for co-management options and legal constraints for offshore aquaculture possibilities in German waters. According to (Wever et al., 2015), not only the research community but also the policy makers are interested in 'sustainable, resource- and space-efficient solutions'. Further, stakeholders' apprehensions are generally referred to biological, economical or technological issues, which need to be eliminated using concrete, transparent tools or, even better, pilot projects for research. Besides, regulations for aquaculture in Offshore Wind Farm (OWF) areas are unclear or even completely lacking (Buck et al., 2004).

Within the interdisciplinary project Offshore Site Selection (OSS), a co-location roadmap is generated for future uses (existing and further) of marine areas in German waters to regulate and reduce the impact on the ecosystem (Buck and Krause, 2013; Grote and Buck, 2014). One objective constitutes the definition of potential areas in the German Exclusive Economic Zone (EEZ) of the North Sea for the co-utilization of OWFs and Integrated Multi-Trophic Aquaculture (IMTA).

IMTA systems combine aquaculture species to recycle effluent dissolved and particulate nutrients from a higher trophic-level species (fish) to nourish extractive, lower trophic-level species, such as filter feeders (mussels, oysters), polychaetes, sea cucumbers and/or seaweed (Neori et al., 2007; Troell et al., in review). These systems aim at balanced nutrient budgets and minimize the waste production originating from fed aquaculture species through the filtering capacity of other extractive species clearing the water (Troell et al., 2009). Moreover, by using nutrient losses of higher trophic-level species as feeding products, IMTA could provide additional economic benefits (Neori et al., 2007). Concerning the GES standards given by the European Commission (EC, 2014b), IMTA systems intend to maintain the functioning and resilience of ecosystems while aiming to prevent the decline of biodiversity such as wild fish stocks caused by human activities (Barrington et al., 2008).

Selecting offshore areas for IMTA brings advantages such as enhanced water quality due to higher levels of dissolved oxygen, less impact by other human activities and opportunities to increase the scale and expansion for aquaculture (Benassai et al., 2011; Buck et al., 2004; Benassai et al., 2014; Troell et al., 2009; Buck, 2007). In spite of the risks (currents, strong wave action, harsh offshore wind conditions) and disadvantages (increasing environmental costs in comparison to onshore aquaculture due to logistics) (Troell et al., 2009), offshore aquacultures have already been successfully undertaken for haddock, halibut and mussels in the US waters (UNH, 2014) as well as oysters and mussels within the German Bight (Pogoda et al., 2011). There can be positive effects concerning shared logistics and infrastructure and restrictions for other types of activities due to the security zone around the OWFs. Further, next to 'room-in-room-solutions' the provision of OWFs structures to build on has been discussed (Buck et al., 2004; Benassai et al., 2014; Joschko et al., 2008; Michler-Cieluch et al., 2009), though, according to (Buck et al., 2008) and (Benassai et al., 2014), this purpose would require alterations to OWF technologies. As this not only leads to increased costs, but also to extraordinary forces acting upon aquaculture cages and potentially destroying OWF structures, latest plans do refrain from banking on such doubled benefits.

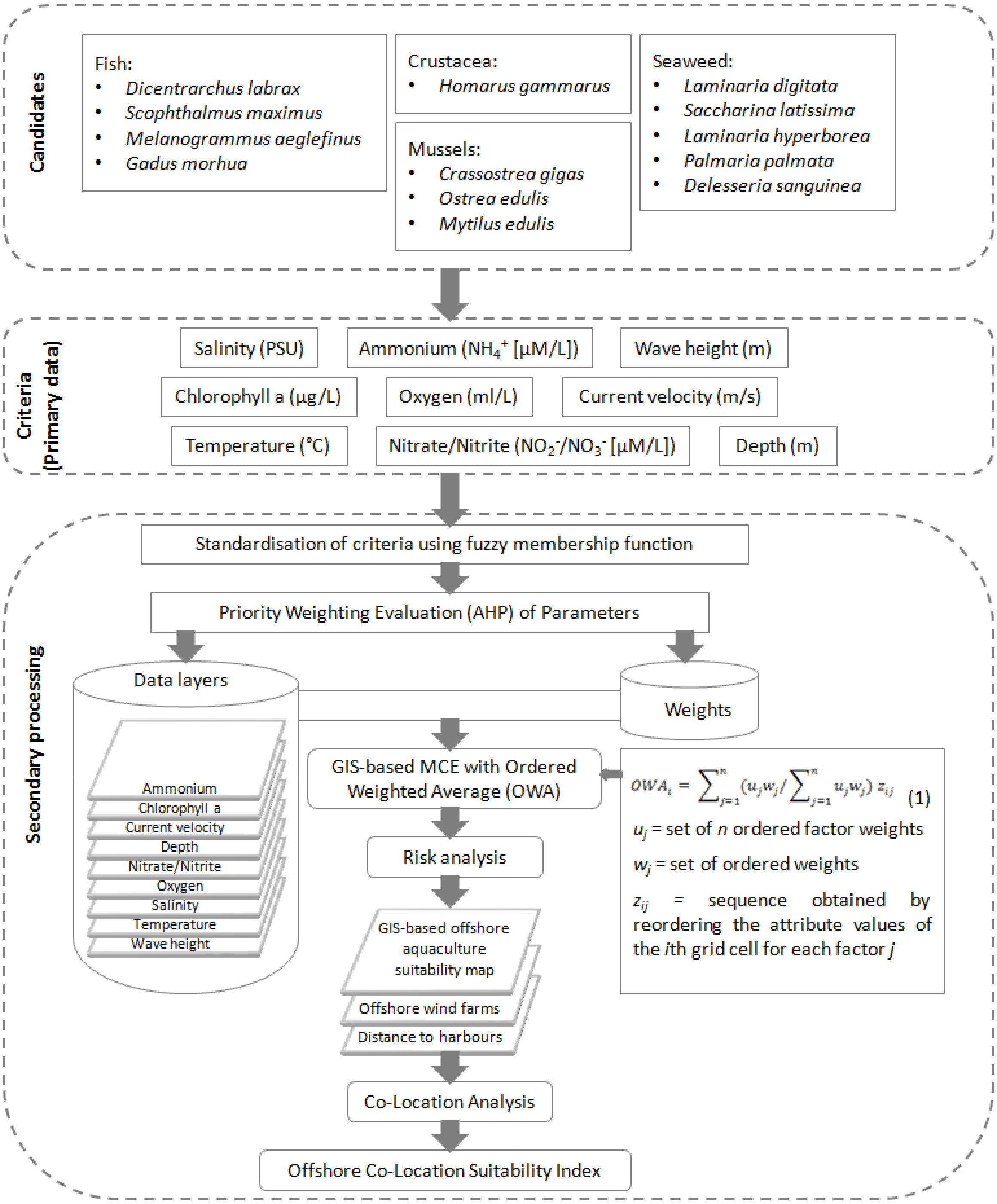


Figure 1: GIS-based modelling framework. Overall methodological approach used to index potential co-locations of offshore wind farms in combination with offshore aquaculture, adapted from (Ouma and Tateishi, 2014).

This study contributes to the indexation of potential areas for the co-location of OWF areas and offshore aquacultures in a spatio-temporal manner. The suitable sites were identified in application of a GIS based Multi-Criteria Evaluation (MCE), which has been previously used for land based site selection by (ElMahdi and Kheireldin, 2004), (Al-Yahyai et al., 2012) and (Gorsevski et al., 2012) and for offshore site selection by (Perez et al., 2005). Subsequently, different approaches to criteria aggregation were examined by using the Ordered Weighted Averaging (OWA) technique (Malczewski, 2006). In this way, the risk of making the wrong decision in aggregating the criteria which determine the suitability of aquaculture sites has been addressed. Using the GIS MCE led to continuous scaling between the risk averse and risk taking OWA operators (Boroushaki and Malczewski, 2008; Gorsevski et al., 2012), providing basic decision scenarios for the evaluation of co-locations in the German EEZ of the North Sea. The study does not account for environmental carrying capacity or environmental impacts, nor does it consider economic viability both of which will influence the success of any offshore aquaculture development.

In this paper, the procedure as well as the main findings of the GIS-based modelling framework (Fig. 1) are summarised to demonstrate the applicability of the methodological approach in a marine ecosystem. Finally, the different spatial co-location scenarios for the coupling of offshore aquacultures and OWF areas are evaluated in order to explore the practical application of co-located offshore aquacultures in combination with OWFs.

3.3 Material and Methods

3.3.1 Case study specifications

The study area comprised the German EEZ of the North Sea with a surface area of 28,539 km² (Fig. 2). Next to other uses, the main human activities regulated by the German MSP are shipping, oil and gas exploitation, cables and pipelines, renewable energy development, and aggregate extraction (Buck et al., 2004; BSH, 2009). The allocation of fishing activities is not included in the German MSP (Gimpel et al., 2013; Fock, 2011; Stelzenmüller et al., 2011). Currently, marine aquaculture is only taking place nearshore in terms of mussel and oyster cultures within the Wadden Sea National Park. Offshore cultivation is currently conducted in various pilot studies, however, it is not yet done at commercial scale (Buck et al., 2004; Buck and Krause, 2012).

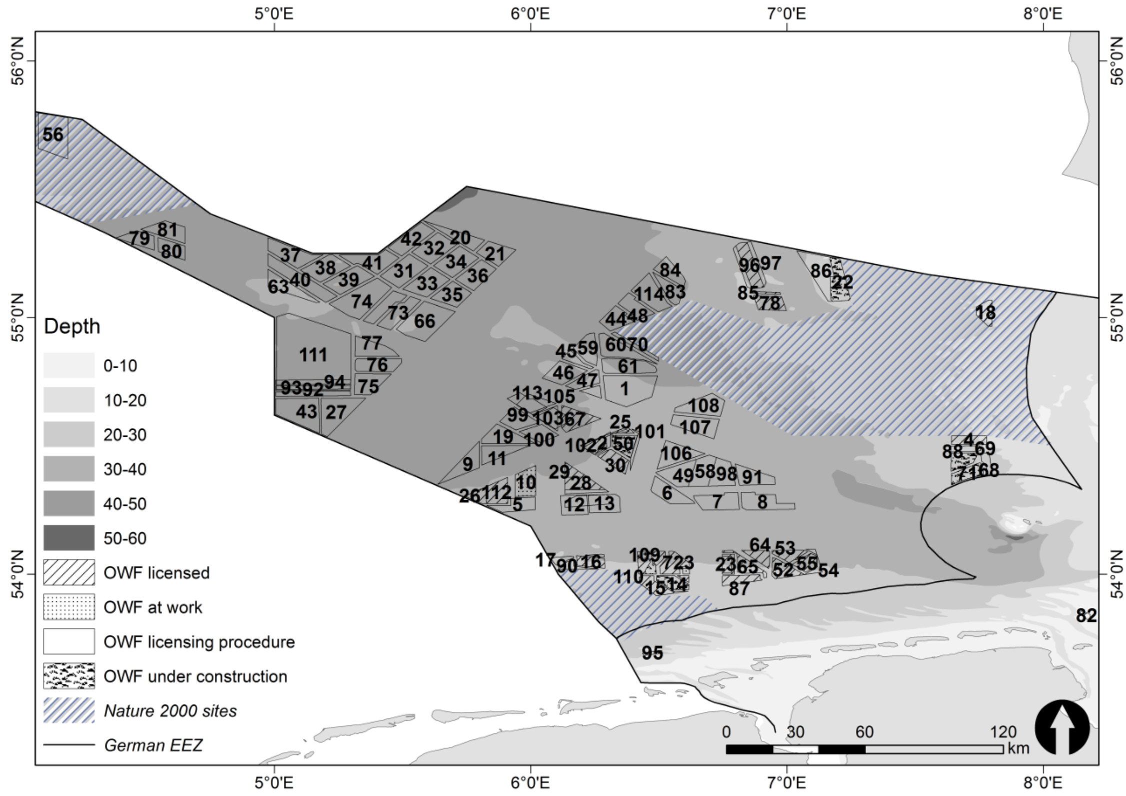


Figure 2: Map of OWF areas in the German EEZ of the North Sea. The OWFs are numbered and hachured per status. The depth levels are scaled in gray. Shaded districts show the Nature 2000 areas. Note that the OWF areas (effective from December 2013; BSH), the depth levels where the OWFs occur and the Nature 2000 sites constituted a physical constraint applied, limiting suitable sites for co-use with aquaculture. OWF 18, 56, 82 and 95 have not been considered during this study, as they appear within the 12 nm zone or in Nature 2000 sites.

The respective study area was subdivided into a set of grid cells accounting for the spatial resolution of available data and computation time. This revealed a grid size resolution of 9.26 km². Within this study, the terms offshore, onshore and nearshore were applied as follows: offshore is beyond 12 nm from the shoreline, nearshore (and coastal/inshore) is between 12 nm and shoreline and onshore is 3 km inland from the shoreline.

3.3.2 Aquaculture candidates and environmental criteria

In total, 21 species of seaweed, bivalves, fish, and crustaceans were identified as adequate aquaculture candidates accounting for their native occurrence in the German North Sea, their resistance to hydrodynamic conditions in offshore environments as well as their economic potential for the EU market. From those, the 13 most promising ones were selected for the scenario configuration. From the literature and experimental data, parameters have been selected for the targeted species (Tab. 1). In order to provide a fundamental data base of environmental variables, hydrographic data from 2002 to 2012 were extracted from National Oceanic and Atmospheric Administration (NOAA) and combined with data provided by the Federal Maritime and Hydrographic Agency (BSH), covering the entire German EEZ of the North Sea: temperature (°C), salinity (PSU), nitrate/nitrite (NO_2^-/NO_3^- [µM/L]), chlorophyll a (µg/L), oxygen (ml/L) and ammonium (NH_4^+ [µM/L]). Because preferences of selected candidates may differ with respective depth layers (Tab. 1) and time scales, gaps in the vertical profiles of the water column were filled as follows:

An average profile for each variable was calculated with all available data in a yearly quarter. This average profile was displaced to minimise the sum of squared differences between average and individual (gappy) profiles. Missing values in the individual profiles were replaced by values of the displaced average profile at the corresponding depth. As the raw data of ammonium were insufficient from April to June, no interpolation was possible.

To complete the set of environmental variables, modelled current velocity (m/s) data from 1958 to 2004 and wave height (m) data from 1958 to 2007 (Weisse and Plüβ, 2005) were used to derive depth stratified mean values per quarter to account for seasonality (1st quarter from January to March; 2nd quarter from April to June, etc.).

To generate the *criteria* for the GIS MCE, the environmental variables were interpolated onto a regular grid encompassing the southern part of the North Sea with universal kriging (Fig. 3). Empirical variograms were calculated and theoretical variogram functions were fitted using weighted least squares, accounting for directional influences, i.e. trends and anisotropy. The fitted omnidirectional and directional Gaussian, spherical or exponential covariance models were examined with the help of cross validation and a Goodness Of Fit (GOF) parameter yielding the best fitting models [31, 32].

3.3.3 Standardisation and priority weighting of criteria

The criteria were standardised on a scale of 1 to 10 (10 = high suitability) using fuzzy membership functions (Eastman, 2001). The function selected governed the shape of the suitability curve and the control points restricted its start/end (Tab. 1). In other words, the starting point represents the inflection point as the membership function rises above 0. For example, current velocity for oarweed (*Laminaria digitata)* starts to be suitable at 0.51 m/s for this species. At 1m/s and with a suitability of 10 the peak of the bell-shaped fuzzy membership function is approached, at 1.48m/s the suitability falls below 1 again, and finally approaches 0 at the end point of 1.54m/s. Consequently, current velocity below 0.51 and above 1.54 m/s gained a suitability of 0 during the standardisation procedure for *L. digitata*. The choice of function and control points was based on expert knowledge and literature research.

Table 2: Aquaculture candidates; the respective modelled depth used for aquaculture site selection; basic site selection criteria for individual aquaculture candidates; fuzzy membership functions with corresponding parameterisation (start/end points) based on literature research and expert knowledge; modified after (Eastman, 2001)*.

Aquaculture candidates	**Modelled depth**	**Basic criteria for site-selection**	**Parameterisation (start&end point)**		**Fuzzy membership function (sigmoidal)**	**Reference**
Fish						
European sea bass (*Dicentrarchus labrax)*	10 - 20 m 20 - 30 m	Ammonium (µM/L)	0	100	monotonically decreasing	FAO (2014)
		Current velocity (m/s)	0	2	monotonically decreasing	
		Oxygen (ml/L)	3.5	∞	monotonically increasing	
		Salinity (PSU)	3	38	bell shaped	
		Temperature (°C)	5	28	monotonically increasing	
Turbot (*Scophthalmus maximus)*	10 - 20 m 20 - 30 m 30 - 40 m 40 - 50 m	Ammonium (µM/L)	0	100	monotonically decreasing	Moksness et al. (2004); Person-Le Ruyet et al. (2006); Daniels and Watanabe (2010)
		Current velocity (m/s)	0	0.5	monotonically decreasing	
		Oxygen (ml/L)	3.5	∞	monotonically increasing	
		Salinity (PSU)	10	35	bell shaped	
		Temperature (°C)	12	18	bell shaped	
Haddock (*Melanogrammus aeglefinus)*	10 - 20 m 20 - 30 m 30 - 40 m 40 - 50 m	Ammonium (µM/L)	0	100	monotonically decreasing	Moksness et al. (2004); Chambers and Howell (2006)
		Current velocity (m/s)	0.3	0.9	bell shaped	
		Oxygen (ml/L)	3.5	∞	monotonically increasing	
		Salinity (PSU)	31	35	bell shaped	
		Temperature (°C)	1	20	bell shaped	

Aquaculture candidates	Modelled depth	Basic criteria for site-selection	Parameterisation (start&end point)		Fuzzy membership function (sigmoidal)	Reference
Atlantic cod (*Gadus morhua)*	10 - 20 m 20 - 30 m 30 - 40 m 40 - 50 m	Ammonium (µM/L)	0	100	monotonically decreasing	Jobling (1988); Moksness et al. (2004); Chambers and Howell (2006)
		Current velocity (m/s)	0	2	monotonically decreasing	
		Oxygen (ml/L)	2.45	∞	monotonically increasing	
		Salinity (PSU)	8	35	bell shaped	
		Temperature (°C)	1	23	bell shaped	
Crustacea						
European lobster (*Homarus gammarus*)	30 - 40 m 40 - 50 m	Current velocity (m/s)	0	0.25	monotonically decreasing	Rosenberg et al. (1991); MarLIN (2014)
		Oxygen (ml/L)	1	∞	monotonically increasing	
		Salinity (PSU)	20	40	bell shaped	
		Temperature (°C)	-1	30	bell shaped	
		Wave height (m)*	0	3	monotonically decreasing	
Bivalves						
Pacific oyster (*Crassostrea gigas*)	0 - 10 m 10 - 20 m	Chlorophyll a (µg/L)	0	∞	monotonically increasing	Pogoda et al. (2011); MarLIN (2014)
		Current velocity (m/s)	0.1	0.8	bell shaped	
		Oxygen (ml/L)	2	∞	monotonically increasing	
		Salinity (PSU)	10	35	bell shaped	
		Temperature (°C)	-1	35	bell shaped	
		Wave height (m)*	0	5	monotonically decreasing	

Aquaculture candidates	Modelled depth	Basic criteria for site-selection	Parameterisation (start&end point)		Fuzzy membership function (sigmoidal)	Reference
European oyster (*Ostrea edulis*)	0 - 10 m 10 - 20 m	Chlorophyll a (µg/L)	0	∞	monotonically increasing	Pogoda et al. (2011); Cano et al. (1997); MarLIN (2014)
		Current velocity (m/s)	0.1	0.8	bell shaped	
		Oxygen (ml/L)	2	∞	monotonically increasing	
		Salinity (PSU)	18	40	bell shaped	
		Temperature (°C)	0	19	bell shaped	
		Wave height (m)*	0	5	monotonically decreasing	
Blue mussel (*Mytilus edulis*)	0 - 10 m 10 - 20 m	Chlorophyll a (µg/L)	0	∞	monotonically increasing	Buck (2007); Karayücel and Karayücel (2000); MarLIN (2014)
		Current velocity (m/s)	0.51	1.54	bell shaped	
		Oxygen (ml/L)	2	∞	monotonically increasing	
		Salinity (PSU)	18	32	bell shaped	
		Temperature (°C)	-10	29	bell shaped	
		Wave height (m)*	0	4	monotonically decreasing	
Seaweed						
Oarweed (*Laminaria digitata*)	0 - 10 m 10 - 20 m	Current velocity (m/s)	0.51	1.54	bell shaped	Mc Hugh (2003); Bolton and Lüning (1982); Lüning (1990); MarLIN (2014)
		Salinity (PSU)	15	40	monotonically increasing	
		Temperature (°C)	1	22	bell shaped	
		Wave height (m)*	0	6	monotonically decreasing	
Sugar kelp (*Saccharina latissima*)	0 - 10 m 10 - 20 m	Ammonium (µM/L)	0	20	bell shaped	Lüning (1990); Bolton and Lüning (1982); Buck and Buchholz (2004); MarLIN (2014)
		Current velocity (m/s)	0.08	1.52	bell shaped	
		Nitrate/Nitrite (mg/L)	3	30	bell shaped	

Aquaculture candidates	Modelled depth	Basic criteria for site-selection	Parameterisation (start&end point)		Fuzzy membership function (sigmoidal)	Reference
		Salinity (PSU)	18	35	monotonically increasing	
		Temperature (°C)	10	18	bell shaped	
		Wave height (m)*	0	6.4	monotonically decreasing	
Cuvie (*Laminaria hyperborea*)	0 - 10 m 10 - 20 m	Current velocity (m/s)	0.51	1.54	bell shaped	Mc Hugh (2003); Bolton and Lüning (1982); Lüning (1990)
		Salinity (PSU)	20	40	bell shaped	
		Temperature (°C)	10	20	bell shaped	
		Wave height (m)*	0	4	monotonically decreasing	
Dulse (*Palmaria palmata*)	0 - 10 m 10 - 20 m	Current velocity (m/s)	0.51	1.54	bell shaped	Mc Hugh (2003); Werner and Dring (2011); Lüning (1990); MarLIN (2014)
		Salinity (PSU)	30	40	bell shaped	
		Temperature (°C)	7	17	bell shaped	
		Wave height (m)*	0	4	monotonically decreasing	
Sea beech (*Delesseria sanguinea*)	0 - 10 m 10 - 20 m	Current velocity (m/s)	0.51	1.54	bell shaped	Mc Hugh (2003); Lüning (1990); MarLIN (2014)
		Salinity (PSU)	18	40	bell shaped	
		Temperature (°C)	1	23	monotonically decreasing	
		Wave height (m)*	0	1	monotonically decreasing	

*Note: wave height was not considered below 10m

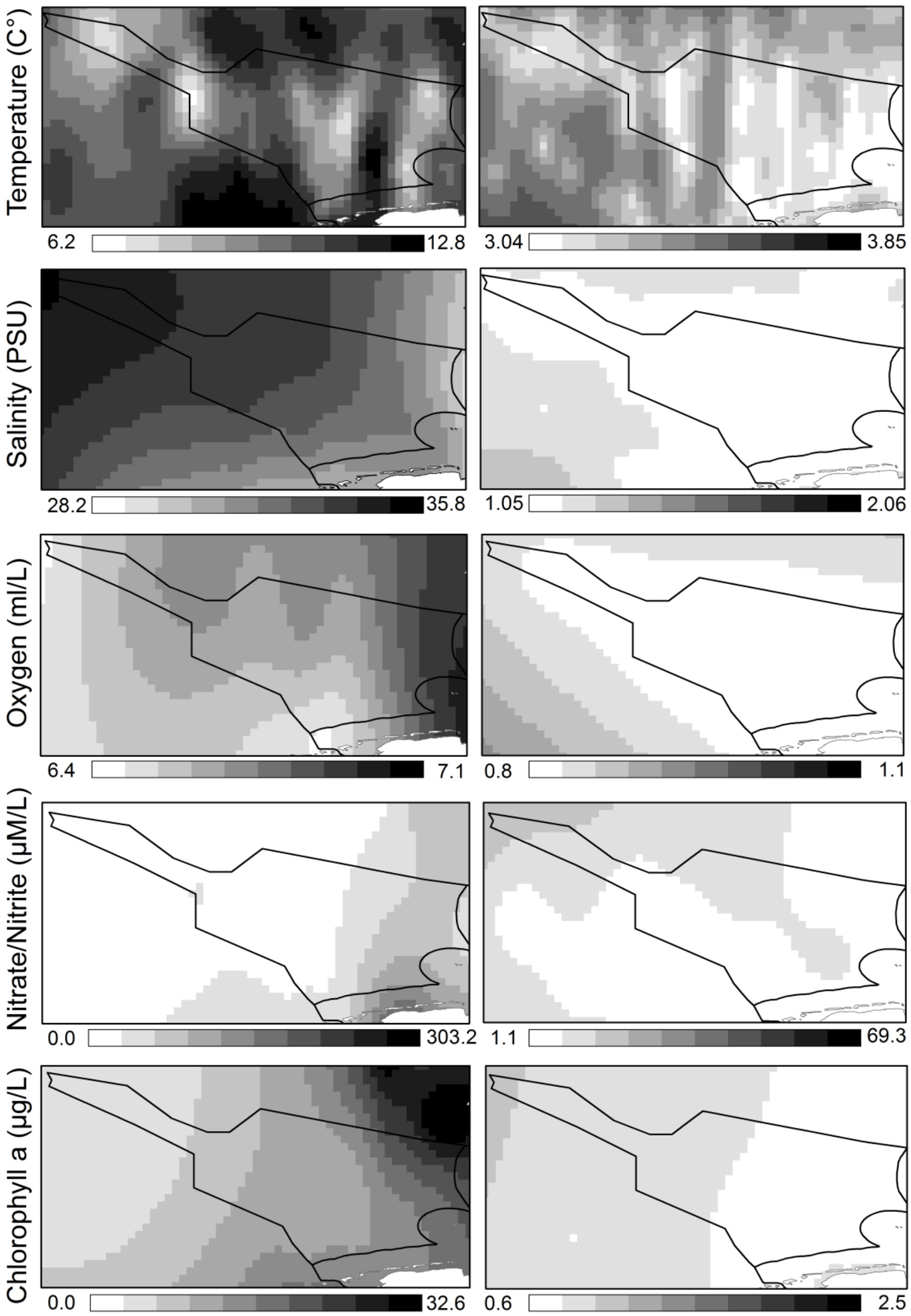


Figure 3: Maps of environmental variables. Left: Results from interpolation using universal kriging. The corresponding kriging error is given to the right. Data are shown at a depth of 0 to 10m and from the 2nd quarter (between 1st of April and 30th of June) in the German EEZ of the North Sea.

3.3.3 Standardisation and priority weighting of criteria

The criteria were standardised on a scale of 1 to 10 (10 = high suitability) using fuzzy membership functions (Eastman, 2001). The function selected governed the shape of the suitability curve and the control points restricted its start/end (Tab. 1). In other words, the starting point represents the inflection point as the membership function rises above 0. For example, current velocity for oarweed (*Laminaria digitata)* starts to be suitable at 0.51 m/s for this species. At 1m/s and with a suitability of 10 the peak of the bell-shaped fuzzy membership function is approached, at 1.48m/s the suitability falls below 1 again, and finally approaches 0 at the end point of 1.54m/s. Consequently, current velocity below 0.51 and above 1.54 m/s gained a suitability of 0 during the standardisation procedure for *L. digitata*. The choice of function and control points was based on expert knowledge and literature research.

The pairwise comparison method of the Analytical Hierarchy Process (AHP) was used to weight the *factors* (standardised criteria) by priority (Jiang and Eastman, 2000; Gorsevski et al., 2012). The weighting of the factors was based on optimal growth under farmed conditions and was judged by experts. The pairwise comparison matrix (Tab. 2) employs an underlying scale from “less important” to “more important”. Importance is rated on a nine point continuous scale. Scaling temperature and wave height equally, as done for *L. digitata*, both factors would be rated with 1. Scaling wave height as “moderately more important” than salinity, the factor wave height would be rated with 3 and the factor salinity with 0.3. These preferences are summarised by normalising the eigenvector associated with the maximum eigenvalue of the pairwise comparison matrix. The eigenvector then gives the relative weights of the factors. Furthermore, the Consistency Ratio (CR) to measure the degree of consistency in judgement of the pairwise comparison was calculated: if CR < 0.1, the ratio indicates a reasonable level of consistency (Boroushaki and Malczewski, 2008).

Table 2: A pairwise comparison matrix of the Analytical Hierarchy Process (AHP) for the calculation of factor weights for aquaculture offshore site selection by the example *Laminaria digitata*, modified after (Gorsevski et al., 2012). The pairwise comparison matrix employs an underlying scale from "less important" to "more important". The intensity of importance is judged by priority ratings, which are provided on a nine point continuous scale. The consistency ratio (CR) < 0.1 indicates consistent judgements.

Less important				*More important*				
1/9	*1/7*	*1/5*	*1/3*	*1*	*3*	*5*	*7*	*9*
extreme	*very strong*	*strong*	*moderate*	*equal*	*moderate*	*strong*	*very strong*	*extreme*

Environmental factors	**E_1**	**E_2**	**E_3**	**E_4**	*Weights*	*CR*
Temperature (E_1)	*1.00*	*1.00*	*1.00*	*3.00*	**0.30**	0.00
Wave height (E_2)	*1.00*	*1.00*	*1.00*	*3.00*	**0.30**	
Current velocity (E_3)	*1.00*	*1.00*	*1.00*	*3.00*	**0.30**	
Salinity (E_4)	*0.33*	*0.33*	*0.33*	*1.00*	**0.10**	
$\sum$	*3.33*	*3.33*	*3.33*	*10.00*		

Considering the lacking information about ammonium, the weighting for the second quarter has been calculated excluding this factor. This exclusion was applied for various fish species, such as Atlantic cod (*Gadus morhua*), European sea bass (*Dicentrarchus labrax*), haddock (*Melanogrammus aeglefinus*), turbot (*Scophthalmus maximus*) and one seaweed species, the sugar kelp (*Saccharina latissima*). Furthermore, the factor wave height has a minor impact at depths below 10 m and was exempted from all weightings for depth layers below 10 m. Next to these factors, physical constraints such as the appropriate depth (m) for each candidate as well as the Nature 2000 areas (BfN, Federal Agency for Nature Conservation (Bundesamt für Naturschutz)) (excluding human activities) were applied, defining the area suitable for co-location.

3.3.4 GIS-based Multi-Criteria Evaluation (MCE) with Ordered Weighted Averaging (OWA) technique

Using the OWA, a range of weighting designs were modelled to address the risk of making the wrong decision in aggregating the factor values determining the suitability of aquaculture sites. The OWA is a family of multi criteria combination procedures:

$$OWA_i = \sum_{j=1}^{n} \left(u_j w_j / \sum_{j=1}^{n} u_j w_j \right) z_{ij} \qquad (1)$$

where $u = (u_1, ..., u_{jn})$ is the set of n ordered factor weights (based on expert knowledge) for individual global weighting; $w = (w_1, ..., w_n)$ is the set of ordered weights for individual local weighting; and $z_i = (z_{i1}, ..., z_{in})$ is the sequence obtained by reordering the values of the ith grid cell for each factor j (Malczewski, 2006). An example for *L. digitata*, where the factor values (4, 2, 8, 3) were associated with the factor weights (0.3, 0.3, 0.3 and 0.1; Tab. 2): According to a descending order of the factor values z (8, 4, 3, 2) the corresponding weights were then reordered (u = 0.3, 0.3, 0.1, 0.3) per grid cell and subsequently combined with a set of ordered weights w. Following (Malczewski, 2006), Regular

Increasing Monotone (RIM) quantifiers α (Tab. 3) were used to generate the ordered weights w. Including these RIM quantifiers α, the OWA is redefined (Malczewski, 2006):

$$\mathrm{OWA_i} = \sum_{j=1}^{n}((\sum_{k=1}^{j} u_k)^{\alpha} - (\sum_{k=1}^{j-1} u_k)^{\alpha})\, z_{ij} \quad (2)$$

for $k = 1,2,..., l; l \leq n.$

Two features can be used to characterise the OWA operators. The first is the attitudinal character (ORness). The ORness represents the degree of risk to misinterpret factor attributes (on a scale of 0 to 1) and can be achieved through Eqs. (3):

$$\mathrm{ORness} = 1 - (\frac{1}{n-1} \sum_r (n-r)\, w_r) \quad (3)$$

where n is the number of factors, r is the order of factors, and w_r is the weight for the factor of the rth order (Gorsevski et al., 2012). The ORness can be specified using α (Jiang and Eastman, 2000; Malczewski, 2006; Boroushaki and Malczewski, 2008). More precisely, by changing α, different degrees of ORness can be obtained: Using the previous example of *L. digitata*, a quantifier of α = 0.0001 (OR operator) would result in a set of OWA weights = 1, 0, 0, 0 (Tab. 3). The OWA value of the respective grid cell would then be 8 but the ORness would result in 1, as a maximum of risk underestimating the factor attributes (i.e. low factor values) is reached.

The second feature to characterise the OWA operators is the degree of dispersion (Tradeoff). The Tradeoff, on a scale of 0 to 1, represents to which level a good performance of one factor can substitute a poor performance of another factor (compensation). The Tradeoff can be obtained through Eqs. (4):

$$\mathrm{Tradeoff} = 1 - \sqrt{(n \sum_r (w_r - 1/n)^2)/n-1} \quad (4)$$

where n is the number of factors, r is the order of factors, and w_r is the weight for the factor of the rth order (Gorsevski et al., 2012). The Tradeoff depends on the weights distributed across all factors used in a weighting combination (Al-Yahyai et al., 2012; Malczewski, 2006; Boroushaki and Malczewski, 2008; Gorsevski et al., 2012). Full compensation would result in a Tradeoff of 1 (Gorsevski et al., 2012; Jiang and Eastman, 2000). Within this study, choosing a quantifier of α = 100 (AND operator) leads to a Tradeoff of 0, because the performance of the factor weight cannot be compensated by other OWA weights. Choosing an α of 1 results in OWA weights equal to a weighted linear combination (WLC) and therefore to the same weighting scheme as given by expert opinion (0.3, 0.3, 0.1, 0.3) in section 3.3.3. Here, with 0.8 a high Tradeoff degree and therefore nearly full compensation would be reached. Choosing an α of 100 leads to the most conservative approach of estimating factor values as the factor limiting the suitability (i.e. the lowest one) is weighted discretely. However, the degree of dispersion would then be 0, too, as there can be no compensation of the performance by other OWA factor weights.

Table 3: Ordered Weighted Average (OWA) operators; fuzzy quantifiers and order weights used to control levels of ORness (risk underestimating factor values) and Tradeoff (compensation between factor values) for the factors predicting suitable sites for *Laminaria digitata*, modified after (Malczewski, 2006) and (Al-Yahyai et al., 2012).

Operator	Quantifier	OWA weights	ORness	Tradeoff	GIS combination procedure
OR	α = 0.0001	1,0,0,0	1.00	0.00	OWA (max)
MIDOR	α = 0.1	.89,.03,.05,.04	0.92	0.15	OWA
AVG	α = 0.5	.55,.08,.2,.16	0.67	0.59	OWA
WLC	α = 1	.3,.1,.3,.3	0.47	0.80	OWA (WLC)
MIDAND	α = 2	.09,.07,.33,.51	0.25	0.58	OWA
AND	α = 100	0,0,0,1	0.00	0.00	OWA (min)

3.3.5 Risk and co-location analysis

The aquaculture suitability modelling resulted in a compilation of geo-referenced layers between the risk averse (AND) and the risk taking (OR) OWA operators comprising the whole German EEZ of the North Sea. Within this study, two scenarios have been regarded as determinative: the AND scenario, which can be seen as the most conservative approach, where the factor attribute limiting the suitability was weighted discretely; and the WLC scenario, where the OWA weights were equal to the weights determined by experts based on optimal growth under farmed conditions (see section 3.3.3).

A combination of both weighting designs was used during risk analysis. With the aim to define a low degree of *risk* (ORness) estimating the factor attributes disproportionately and a low degree of *compensation* (Tradeoff) between the factor weights, the α parameter was raised up to 100 (AND operator). Thus, the grid cells containing factor values of 0 were identified and excluded from further assessments. To calculate the optimal growth under farmed conditions, α was specified as 1 (WLC operator), and the factor values were weighted on the base of expert opinion (as previously done by using the AHP) for the remaining grid cells. In other words, only if one cell was indexed as suitable during the risk analysis, compensation was allowed and the WLC score was recorded for the GIS-based offshore aquaculture suitability map.

Accounting for the spatial overlap of the aquaculture suitability layers with the respective geo-referenced OWF areas provided by the BSH (effective from December 2013), an offshore co-location suitability index has been developed. As the study area comprised the German EEZ of the North Sea, OWF areas outside of the EEZ (or inside the 12 nm zone) were not considered.

Within this study, the scenarios and suitability scores have not only been stratified by depth, moreover the seasonality has been accounted for. The reason is that some aquaculture candidates might be cultivated onshore and reared offshore at a stadium when getting more resilient. As offshore aquaculture leads due to logistics to increasing environmental costs in comparison to onshore and land based aquaculture, the factor distance of the OWF areas to the next harbour has been incorporated.

3.4 Results

3.4.1 Standardisation and priority weighting of criteria

The factor values resulting out of the standardisation process revealed multiple limitations in sites suitable for aquaculture. Most limitations for species such as *S. maximus* or European lobster (*Homarus gammarus)* were the result of low (maximum) parameterisations for current velocity. Further, the maximum parameterisation of temperature or wave height, especially in the case of sea beech (*Delesseria sanguine),* led to limited suitability.

With CR < 0.1 the priority weighting revealed fair results for all fish species (*D. labrax*, *S. maximus*, *M. aeglefinus*, *G. morhua*), *H. gammarus* and all seaweed species (*L. digitata*, *S. latissima*, cuvie *Laminaria hyperborea*, dulse *Palmaria palmata*, *D. sanguinea*). Whereas the pairwise comparison for all bivalve species (blue mussel *Mytilus edulis,* European oyster *Ostrea edulis* and Pacific oyster *Crassostrea gigas*) revealed a CR of 0.14, indicating an imbalanced weighting composition of the respective optimal growth factors (Boroushaki and Malczewski, 2008). Leaving out the factor wave height at 10 to 20 m depth, a CR of 0.03 was assessed.

3.4.2 GIS-based Multi-Criteria Evaluation (MCE) with Ordered Weighted Averaging (OWA) technique

Different spatial co-location scenarios were constructed by using a range of aggregation approaches, i.e. OWA operators (Fig. 4). Changing the α parameter and therefore the order weights of various factors led to multiple levels of *risk* (ORness) over- or underestimating individual factor attributes. In addition, it leads to several degrees of *compensation* (Tradeoff) between the factor weights. For all candidates, the AND scenario has been characterised by zero values in the case of both, the level of risk as well as the degree of compensation.

Weighting *L. digitata*, the degree of dispersion between the factor weights approached nearly full compensation (= 0.8) when α was set to 1, whereas the risk level was assessed as 0.53. Calculating these features for *S. latissima* resulted in a risk level of 0.47 and a compensation of 0.82. Full compensation (= 1) in combination with a low risk level (= 0.5) was reached by weighting all factors equally as done for *P. palmata* and *L. hyperborea*. When assessing the candidates *D. sanguine* and *H. gammarus*, the compensation resulted in 0.67 and the risk level of 0.56.

Calculating these features for all bivalve species resulted in a compensation of 0.77 and a risk level of 0.43. The lowest value (= 0.39) was obtained when assessing the risk level for *S. maximus*, the compensation resulted in 0.78. The same level of compensation was calculated for *D. labrax*, while the degree of risk resulted in 0.5. When assessing the aquaculture candidates *G. morhua* and *M. aeglefinus*, a compensation of 0.76 and a risk level of 0.49 were reached.

All other scenarios obtained by using the OWA operators resulted in intermediate degrees of risk and compensation between the OR and the AND scenario (Tab. 3).

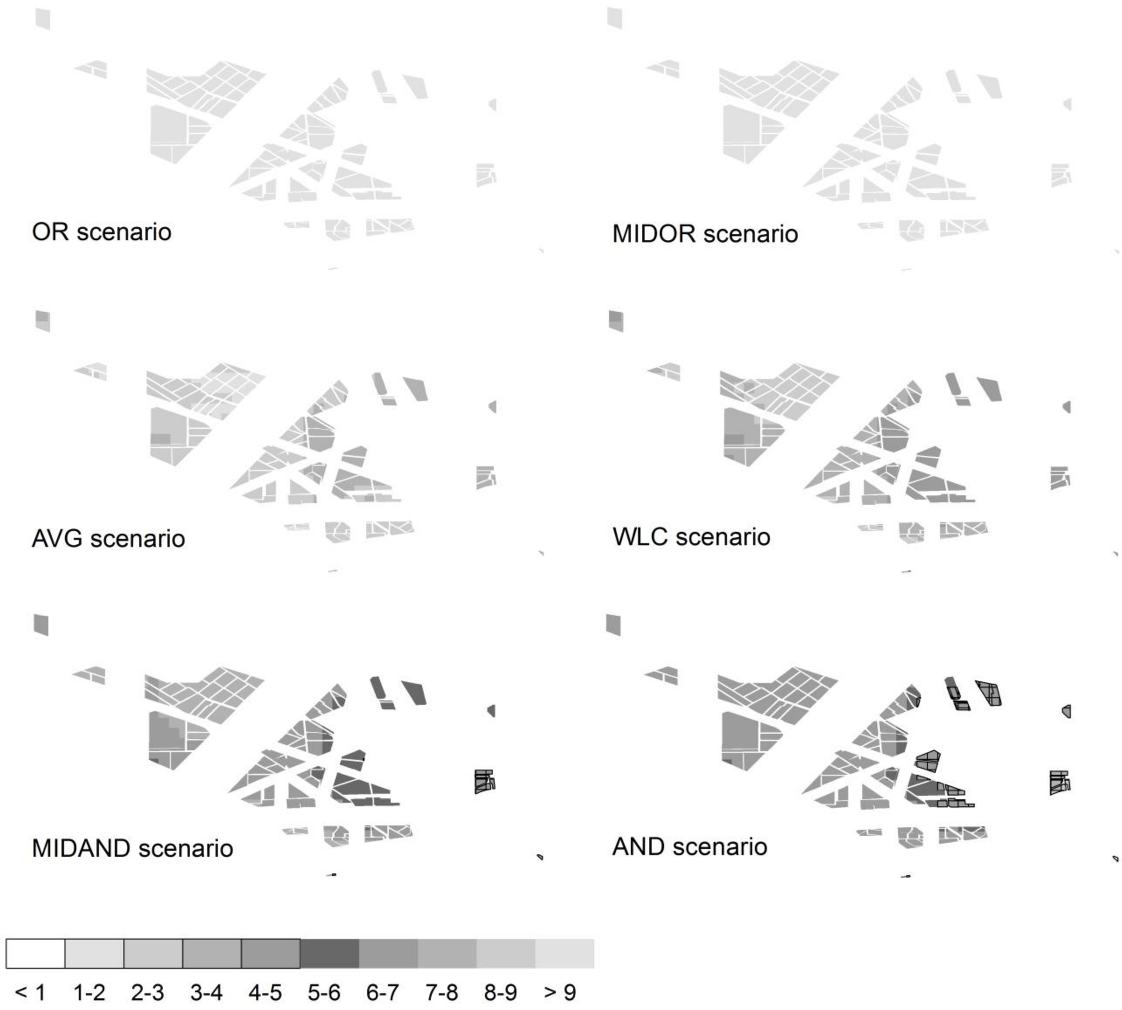


Figure 4: Maps of generated OWA scenarios for *Laminaria digitata.* (For illustration purposes results are shown for *L. digitata*). The OWA scenarios to index sites suitable for the co-location of *L. digitata* aquaculture and OWFs. The OWA operators shown in Table 3 were used to assess multiple levels of ORness (risk misestimating factor values) and Tradeoff (compensation between factor values). Data are given at 0 to 10m depth, reporting aquaculture suitability (0 – 10, 10 = most suitable) from the 2nd quarter (between 1st of April and 30th of June) in the German EEZ of the North Sea.

3.4.3 Risk and co-location analysis

As described in section 3.3.5, only those grid cells were recorded for the GIS-based offshore aquaculture suitability map, which have been indexed as suitable during the risk analysis. This procedure meant a loss of suitable aquaculture sites for most of the candidates. When the offshore co-location suitability index was developed by accounting for i) overlaps between the aquaculture sites and the OWF areas and ii) distance of the OWF to the next harbour, the actual extent of loss became visible:

For the seaweed candidate *L. digitata*, the OWFs 4, 68, 69, 71 and 88 were assessed to be not or just partial suitable during the 3rd quarter and even more OWFs have been indexed as unsuitable during the 1st quarter. Next to variations with season the predicted suitability scores differed in comparison to the depth layers. *L. digitata* scored highest at a depth of 10 to 20 m (Fig. 5 and 6).

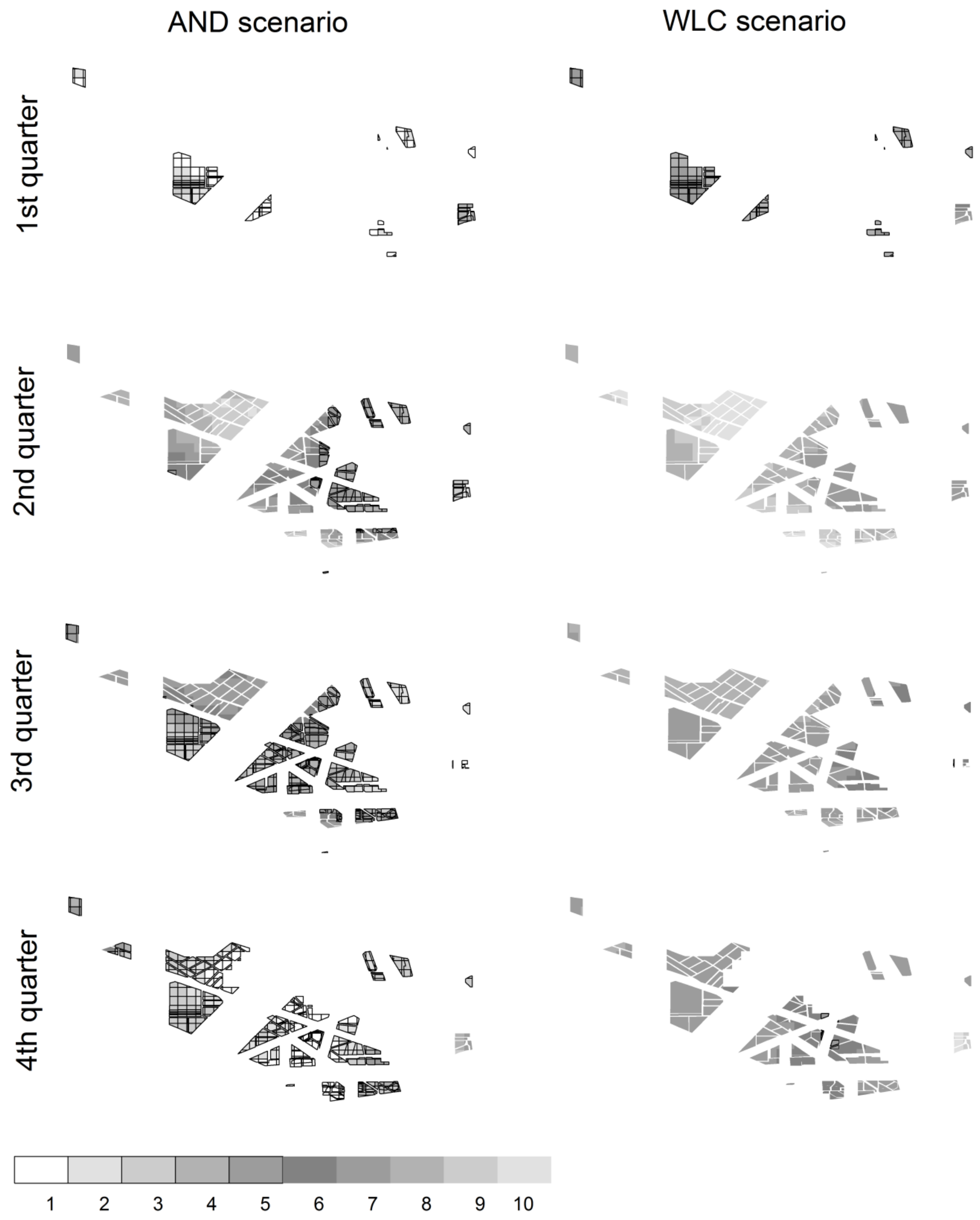


Figure 5: Maps of predicted co-location suitability for *Laminaria digitata*. (For illustration purposes results are shown for *L. digitata*). To index the suitability of co-location sites, two scenarios were regarded as determinative: the AND scenario (left) and the WLC scenario (right). Data are given per quarter (1st quarter = 1st of January to 31st of March, etc.), reporting aquaculture suitability (0 – 10, 10 = most suitable) for *L. digitata* at a depth of 10 to 20m in combination with OWF areas in the German EEZ of the North Sea.

Assessing *S. latissima* resulted in suitable OWF sites during the 2^{nd} and the 4^{th} quarter at 0-10 m depth. *L. hyperborea* showed the most suitable sites in the 3^{rd} quarter, a few during the 2^{nd} and 4^{th} quarter and none in the 1^{st} quarter. A depth of 10 to 20 m ensued higher suitability scores than 0 to 10 m. For the seaweed candidate *P. palmata*, the 2^{nd} quarter was assessed to be highly suitable, the 3^{rd} and 4^{th} quarter as partially suitable (Fig. 7). A significant difference between the depth levels for *P. palmata* could not be ascertained, whereas *D. sanguinea* only showed suitable sites at 10 to 20 m depth from the 2^{nd} to the 4^{th} quarter (Fig. 6). Assessing the bivalve species during the 2^{nd} and 3^{rd} quarter resulted in similar suitable sites for *O. edulis* and *C. gigas*, at both modelled depth levels (Fig. 6). The aquaculture candidate *M. edulis* only showed suitable sites at the OWFs 4, 68, 69, 71 and 88, all situated in front of the German coast, though at all depth levels assessed from the 2^{nd} to the 4^{th} quarter. Another 'stable' candidate proved to be *G. morhua* at all depth levels and quarters, but especially showing high scores in between 30 to 40 m (Fig. 6 and 7). Assessing *D. labrax* resulted in a comparably low loss of suitable sites with the OWFs 4, 68, 69, 71 and 88 during the 1^{st} quarter. The candidate scored highest at 10 to 20 m depth. While *M. aeglefinus* showed least suitable sites during the 2^{nd} and 3^{rd} quarter at all modelled depth levels (Fig. 7), *S. maximus* featured two suitable OWF sites (71 and 88) at 10 to 20 m depth and 20 to 30 m depth, both during the 3^{rd} quarter. Assessing *H. gammarus* did not yield any suitable OWF.

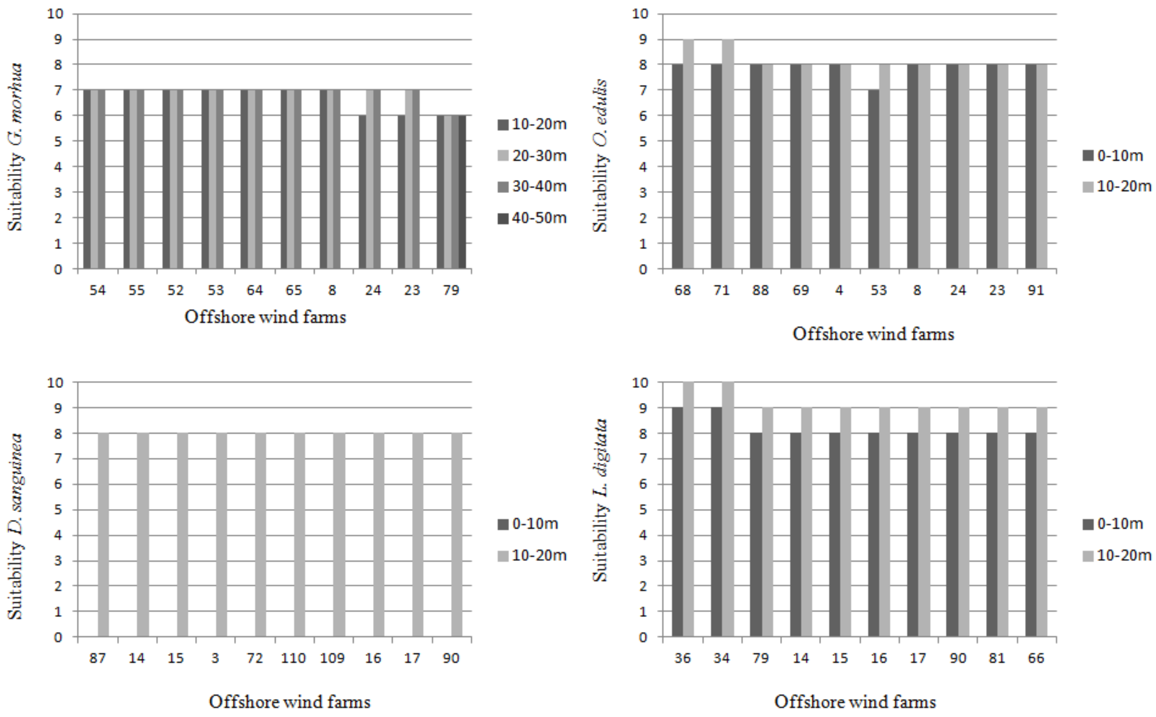


Figure 6: Predicted co-location suitability per depth. OWF identifiers (Figure 1) representing the 10 most suitable OWF areas (0 – 10, 10 = most suitable) for multiple candidates. The scores are gained using the WLC scenario and shown at the 2nd quarter, greyed out corresponding to the depth layer assessed. The suitability and the distance to the next harbour determined the order of the OWF identifiers*.

*Note: wave height was not considered below 10m, ammonium was not considered assessing *G. morhua*

In most of the cases the highest suitability scores were reached in the 2nd quarter (Fig. 6). Nevertheless, *D. Labrax*, *S. maximus* and *L. hyperborea* scored highest during the 3rd quarter. The differences between the quarters can be high, as for example for *M. aeglefinus* in the OWF areas 68 and 71, or comparatively low, as shown for *G. morhua*, where the suitability scores did not vary significantly with season (Fig. 7).

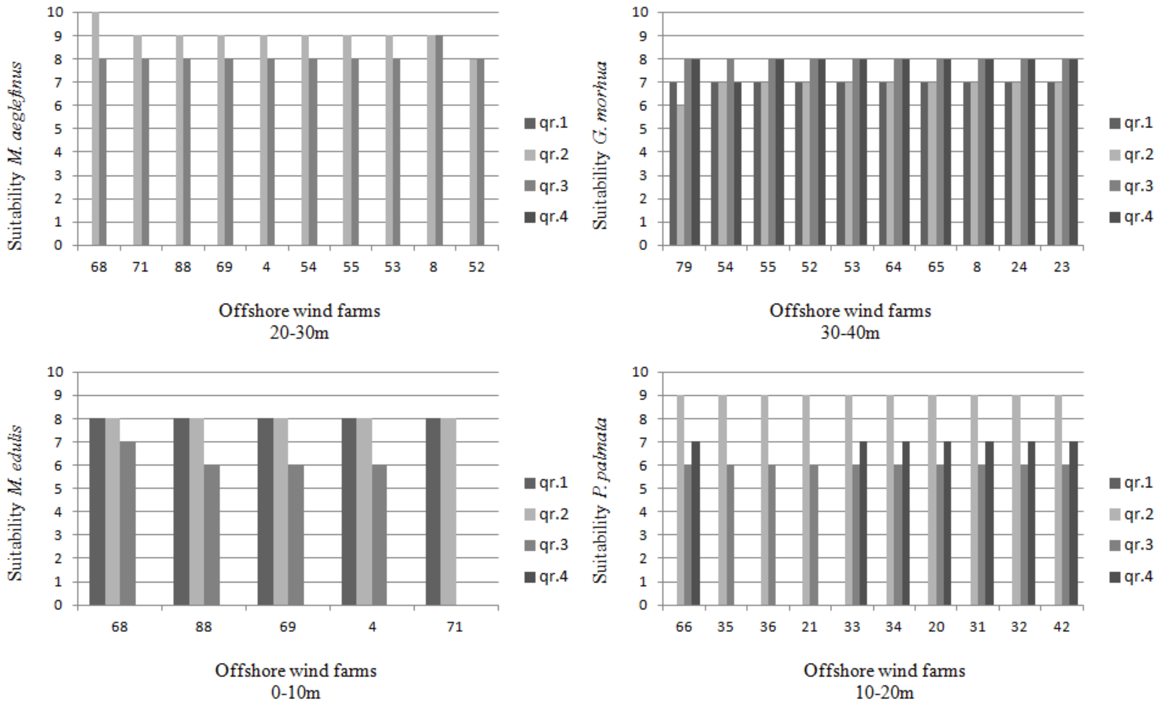


Figure 7: Predicted co-location suitability per quarter. OWF identifiers (Figure 1) representing the 10 most suitable OWF areas (0 – 10, 10 = most suitable) for multiple candidates. The scores are gained using the WLC scenario and shown at the most suitable depth layer, greyed out corresponding to the quarter assessed (‘qr. 1’ = between 1st of January and 31th of March etc.). The suitability and the distance to the next harbour determined the order of the OWF identifiers*.

*Note: wave height was not considered below 10m, ammonium was not considered in the 2nd quarter for *M. aeglefinus*

The suitability scores for selected OWF areas per quarter are shown at 0 to 10 m depth in Table 4 and at 10 to 20 m depth in Table 5. Several OWFs provided robust sites suitable for multiple aquacultures such as 69 or 88, exhibiting a possible combination of six or ten aquaculture candidates, respectively. Table 4 enables the decision maker to choose from a set of candidates and between the most suitable sites for *M. edulis*, *O. edulis*, *C. gigas*, *L. hyperborea*, *P. palmata*, *L. digitata*, *D. sanguinea* or *S. latissima*.

Table 4: Predicted multifunctional use matrix (depth: 0 – 10m). OWF identifier; distance to the nearest harbour (1 = Bremerhaven, 2 = Cuxhaven); suitability (0 – 10, 10 = most suitable) of multiple aquaculture candidates at selected OWF areas from the 2nd quarter (between 1st of April and 30th of June) and the corresponding IQR (given in brackets), used to represent the temporal variation. The last column ∑ quotes the number of times the OWF was selected as suitable per candidate for each group (bivalves and seaweed).

OWF	Distance	Harbour	*C. gigas*	*O. edulis*	*M. edulis*	∑	*L. hyperborea*	*L. digitata*	*P. palmata*	*S. latissima*	*D. sanguinea*	∑
88	91.54	2	8 (8)	8 (6.5)	8 (3.5)	3	0 (6)	6 (0.75)	4 (3.25)	8 (2)	0 (0)	3
69	94.18	2	9 (8.25)	8 (7.25)	8 (2.75)	3	0 (6)	6 (0.75)	4 (3)	8 (2)	0 (0)	3
54	110.9	1	7 (5.5)	7 (5.5)	0 (0)	2	6 (6)	7 (0.5)	6 (2.25)	9 (3.75)	0 (0)	4
55	113.1	1	7 (5.5)	7 (5.5)	0 (0)	2	6 (6)	7 (1.75)	6 (1.5)	9 (3.75)	0 (0)	4
53	117.4	1	7 (5.5)	7 (5.5)	0 (0)	2	6 (6)	8 (2)	6 (1.5)	9 (2.25)	0 (0)	4
72	144.4	1	8 (8)	8 (6.5)	8 (8)	3	0 (1.5)	6 (2)	4 (4.75)	8 (2)	0 (0)	3
5	188.9	1	8 (8)	8 (6.5)	8 (3.5)	3	0 (6)	6 (0.75)	4 (3.25)	8 (2)	0 (0)	3
70	197.1	1	8 (8)	8 (6.5)	8 (3.5)	3	0 (6)	6 (0.75)	4 (3.25)	8 (2)	0 (0)	3
19	200.3	1	8 (8)	8 (7.25)	8 (7.25)	3	0 (4.5)	6 (0.75)	4 (2)	8 (2)	0 (0)	3
11	201.1	1	6 (5.25)	6 (6)	0 (0)	2	6 (6)	8 (2)	7 (2.5)	8 (5.75)	0 (0)	4

Table 5: Predicted multifunctional use matrix (depth: 10 – 20m). OWF identifier; distance to the nearest harbour (1 = Bremerhaven, 2 = Cuxhaven); suitability (0 – 10, 10 = most suitable) of multiple aquaculture candidates at selected OWF areas from the 2nd quarter (between 1st of April and 30th of June) and the corresponding IQR (given in brackets), used to represent the temporal variation. The last column ∑ quotes the number of times the OWF was selected as suitable per candidate for each group (fish, bivalves and seaweed).

OWF	Distance	Harbour	*D. labrax*	*S. maximus*	*M. aeglefinus*	*G. morhua*	∑	*C. gigas*	*O. edulis*	*M. edulis*	∑	*L. hyperborea*	*L. digitata*	*P. palmata*	*S. latissima*	*D. sanguinea*	∑
88	91.54	2	5 (2.75)	0 (1.5)	9 (8.25)	7 (1)	3	8 (8)	8 (6.5)	8 (2.75)	3	0 (6)	6 (1)	5 (4.25)	7 (1.75)	6 (0.75)	4
69	94.18	2	5 (2.75)	0 (0)	8 (8)	7 (1)	3	9 (8.25)	9 (6.75)	8 (2.75)	3	0 (6)	6 (1)	5 (4)	6 (2.5)	6 (0.75)	4
54	110.9	1	6 (1.5)	0 (0)	9 (8.25)	7 (1)	3	7 (5.5)	8 (5)	0 (0)	2	6 (6.25)	8 (1)	7 (2.5)	7 (7)	7 (0.75)	5
55	113.1	1	5 (0.75)	0 (0)	9 (8.25)	7 (1)	3	7 (5.5)	7 (4.75)	0 (0)	2	6 (6.25)	8 (2)	7 (2.5)	8 (7.25)	7 (1.75)	5
53	117.4	1	5 (0.75)	0 (0)	9 (9)	7 (1)	3	7 (5.5)	7 (4.75)	0 (0)	2	6 (6.25)	8 (2)	7 (2.5)	7 (6.25)	7 (1.75)	5
65	124	1	5 (0.75)	0 (0)	8 (8.25)	7 (1)	3	7 (5.5)	7 (4.75)	0 (0)	2	5 (5.5)	8 (2)	7 (2.5)	7 (6.25)	7 (1.75)	5
91	139.4	1	5 (0.75)	0 (0)	8 (8.25)	7 (0.25)	3	7 (6.25)	8 (5)	0 (0)	2	5 (5.25)	7 (1.5)	6 (3)	7 (2.5)	6 (1.25)	5
5	188.9	1	5 (0.75)	0 (0)	6 (6.5)	6 (0.5)	3	8 (8)	8 (6.5)	8 (2.75)	3	0 (6)	6 (1)	5 (4.25)	7 (1.75)	6 (0.75)	4
70	197.1	1	5 (0.75)	0 (0)	7 (8)	6 (0.5)	3	8 (8)	8 (6.5)	8 (2.75)	3	0 (6)	6 (1)	5 (4.25)	7 (1.75)	6 (0.75)	4
9	208.5	1	5 (1.25)	0 (0)	6 (6.5)	6 (0.5)	3	7 (6.25)	8 (5.75)	0 (0)	2	5 (5.25)	7 (2.25)	7 (3.25)	7 (1.75)	7 (1.5)	5

Table 5 features a selection of suitability scores for *G. morhua*, *D. Labrax*, *M. aeglefinus*, *S. maximus*, *M. edulis*, *O. edulis*, *C. gigas*, *L. hyperborea*, *P. palmata*, *L. digitata*, *D. sanguinea* and *S. latissima*. The OWF areas shown in Table 5 exhibited comparably high suitability scores for *M. aeglefinus*, (6-9), *O. edulis*, (7-9) and *L. digitata* (6-8).

3.5. Discussion

Competition for maritime space has highlighted the need for efficient management and synergies between different activities. Within the scope of the project OSS, sites suitable for co-location of individual offshore aquacultures in combination with OWFs were indexed. In the present publication the applicability of the GIS-MCE to assess the suitability of such sites is demonstrated.

3.5.1 The weighted GIS-based modelling framework

Uncertainty in standardised data derived by expert knowledge and uncertainty from the interaction of ranked criteria can be smoothed out using the fuzzy membership functions. These functions have already been successfully applied in a range of analyses using MCE (Boroushaki and Malczewski, 2008; Malczewski, 2006; Gorsevski et al., 2012). Nevertheless, in the course of this study, using the AHP resulted in inconsistent judgments ($CR < 0.1$) for the three bivalve species blue mussel *M. edulis*, European oyster *O. edulis* and Pacific oyster *C. gigas*. This inconsistency might be explained by a high preference only given to chlorophyll a (0.34). These original weights need to be revised in the future.

Employing the OWA technique resulted in a range of aggregation methods to combine factor attributes determining the suitability of aquaculture sites. Applying the OR operator revealed a high risk (ORness) to overestimate the factor with the highest corresponding value. Hence, the OR scenario might be interpreted as the most optimistic and risky evaluation strategy. Consequently, the OR operator is not applicable for the determination of suitable aquaculture sites. The risk of ignoring essential factors is just too high. On the contrary, applying the AND operator yielded a low risk in overestimating factors as it is based on the lowest factor value. Therefore, it can be interpreted as the most conservative and risk averse evaluation strategy and is regarded as applicable for the determination of suitable aquaculture sites. Applying the WLC or other operators between OR and AND resulted in intermediate risk levels for the aquaculture candidates. The lowest risk level (turbot *S. maximus*) was given when applying the WLC operator and can be explained by the fact, that high factor values were combined with low order weights and vice versa.

As the degree of compensation (Tradeoff) depends on the weights distributed across all factors included into a weighting combination, both, the OR and the AND scenario resulted in a Tradeoff of 0. Whereas the WLC scenario yielded multiple degrees of compensation, individually depending on the factor weights distributed when evaluating each candidate. Full compensation (in combination with a low risk level) was reached when all factors were weighted equally as for dulse *P. palmata* and cuvie *L. hyperborea* in the WLC scenario assessment. In summary, all modelled outputs were consistent and demonstrated overlaps among the scenarios for individual aquaculture candidates. Moreover, results from the AND scenario yielded usually the same aquaculture candidates as most suitable as revealed with the WLC scenario.

3.5.2 Suitability for co-location of offshore aquaculture and wind farms

Considering both, the AND scenario and the WLC scenario during risk analysis was justified by the following facts:

The weighting of the factors was judged by experts. In reality, from a biological perspective, factors such as highly preferred temperature cannot compensate unsuitable oxygen concentrations. All factors determine the suitability of an area. Moreover, interpreting these results from an economic point of view, the AND scenario was the most certain approach as the factor limiting the suitability was weighted discretely. Therefore, a complete failure leading to the loss of organisms and in this way to the loss of aquaculture revenues can be classified as improbable. Furthermore, when using the AND scenario, expert judgement in weighting the factors determining aquaculture suitability is treated with caution because it can be incomplete, affected by natural randomness or imprecise due to vague, underspecified or context-dependent terms (Perera et al., 2011).

However, the focus of this study has been on defining areas for the co-location of OWF areas and offshore aquacultures. When providing basic management scenarios for decision makers, there is a principal need to include cost effective settings. Therefore, the factors were weighted according to optimal growth under farmed conditions by expert judgment. The WLC scenario resulted in the same weighting scheme as given by expert opinion and can therefore be interpreted as the most cost effective one. Even so, using the WLC, the highest degrees of compensation were reached and the level of risk estimating the factor attributes varied disproportionately.

Apart from this, all following results discussed below were based on the offshore co-location suitability index, which was developed by accounting for i) overlaps between the aquaculture sites and the OWF areas and ii) distance of the OWF to the next harbour.

Fish

European sea bass (*D. labrax)* showed high suitability in combination with OWF areas over all depth layers and quarters. Analysing this candidate during the 2nd quarter at the OWF areas 4, 54, 55, 69 and 88, which are all situated near the transition zone from offshore to nearshore (12 nm boarder) at 10 to 20 m depth, resulted in low interquartile ranges (IQR). Therefore, the results depict a similar suitability in all seasons. Even higher suitability scores were yielded for Atlantic cod (*G. morhua*). This agrees with (Chambers and Howell, 2006), who showed that cultivating cod all year-round (1.5 years) submerged 12 m below the surface resulted in a survival rate of 92%. The same cultivation approach was tested successfully for haddock (*M. aeglefinus*). Within this study, the high IQR given in Tab. 5 disagree with these results, which can also be said about the best suitability scores yielded at 20 to 30 m depth (Fig. 7). The most efficient scenario shown for haddock scored the highest suitability in the 2nd quarter where ammonium was not considered. Ammonium usually limits the suitability for fish species cultivated in onshore recirculation aquaculture systems due to toxic effects of ammonia in high concentrations. Nevertheless, these concentrations get resolved in offshore areas by strong currents, moreover benefitting from strong wave action and harsh offshore winds. Therefore, recirculation aquaculture system conditions are hard to compare to offshore conditions. Furthermore, the results out of the 3rd quarter showed comparably high scores for haddock. The low values for turbot can be explained by the parameterisations chosen during standardisation procedure. Limitations were given due to the parameterisation of temperature and the rather low current velocity turbot (*S. maximus*) can withstand.

Crustaceans

Analysing European lobster (*H. gammarus*) did not result in any suitable aquaculture site at all. The factor reducing the suitability was identified to be current velocity, which has been defined as unsuitable > 0.25 m/s.

Bivalves

The suitability scores for blue mussels (*M. edulis*) at 0 to 10 m depth (Fig. 7) are confirmed by a study of (Maar et al., 2009) and (Buck, 2007), who demonstrated a 7 to 18 times higher biomass for blue mussels located higher up in the water column (on collectors or artificial reefs, such as turbine pillars) than those located deeper (on the scour protection), caused by an enhanced advective food supply. It has to be noted for this type of cultivation, that mussels have two cultivation stages, (1) collecting the seed and (2) grow out to market size. The latter stage is the important one in the context of this study.

In general, offshore mussel cultures might be feasible as indicated by the high suitability scores Pacific oyster (*C. gigas*) and European oyster (*O. edulis*) yielded at the OWF areas 5 or 70. These ones are stable over the year within both modelled depth layers (Tab. 4 and 5, Fig. 6). There are already nearshore mussel cultures in the Wadden Sea of the German EEZ, in nearshore waters of Ireland and Scotland and in offshore waters of the USA, France, the Netherlands, New Zealand, Japan, and China (Troell et al., 2009).

Seaweed

According to (Troell et al., 2009), the seaweed candidates of the genus *Laminaria* (*L. digitata*, *S. latissima* and *L. hyperborea)* prefer low water temperatures. Indeed, *L. hyperborea* showed higher values at 10 to 20 m depth but scored highest in the 3rd quarter, while the lowest temperature occurs during the 4th and 1st quarter. At this point, it has to be taken into account that seaweed cultivation strongly depends on the season. If the seaweed is part of an IMTA approach and also a candidate to be sold on the EU market, it has to be harvested latest by the end of the 2nd quarter. If the seaweed is cultivated within a bioremediation concept and is only used to extract nutrients from the water column, it can be on-site year around (Buchholz et al., 2012).

The whole study area became only suitable for *D. sanguine* in deeper waters. This can be explained by the fact, that wave height, which limits the suitability for seaweed species, becomes a minor impact at depths below 10 m. Furthermore, according to (Troell et al., 2009), seaweed offshore cultures are adaptable to limited light conditions and can therefore be cultivated at greater depth. Moreover, if seeded elaborately on the rope and transferred at sea at a juvenile stage, holdfasts will not be dislodged and cauloids will not break leading to a resistance to harsh conditions (Buck and Buchholz, 2004). These might be interesting facts for culturing the seaweeds *P. palmata*, *L. digitata*, *D. sanguine* and *L. hyperborea*, as these candidates scored highest at 10 to 20 m depth.

During a study by (Handå et al., 2013), *S. latissima* was cultivated in the most effective way at a depth of 2 and 5 m during early summer time. These results match the suitability scores attained during the 2nd quarter at 0 to 10 m depth (Tab. 4). Judged by the fact, that *S. latissima* has a preference to take up ammonium, the factor was included in the determination of suitable sites (Tab. 1). The high scores could therefore be explained by the fact, that there was no ammonium considered in the 2nd quarter. However, as ammonium has been weighted with 0.01 during the AHP, this fact might be disregarded.

The range of suitability scores for identified aquaculture candidates across the OWF areas per quarter reflected the dependence of each candidate on local conditions in the study area and key environmental criteria as given in Tab. 1. Furthermore, the differences in the suitability between depth layers and quarters justified the scenario settings as described in section 3.3.2. Consequently, if strong currents and waves limit the suitability over the year, affected candidates might be initially cultivated onshore and subsequently applied offshore at a more resilient stage. This might be possible for fish and oyster species showing high suitability in certain quarters but also high IQRs, such as at the OWF areas 88 and 69 (Tab. 5). When evaluating the OWFs 34 and 36 (Fig. 6), a kind of patchiness gets visible, as these OWFs are situated next to each other further offshore, where the temperature is lower than nearshore. The OWF areas 4, 68, 69 and 71 and 88 are situated near the transition zone from offshore to nearshore (12 nm boarder) at a depth of 10 to 20 m, exhibiting higher temperatures as well as nutrient enriched water columns.

With the focus on IMTA, the results given in Tab. 4 and 5 are quite revealing: The suitability scores given at the OWF areas situated near the 12 nm boarder (depth: 10 to 20 m; 54, 55, 69 and 88) indicate a possible set of aquaculture candidates consisting of *M. aeglefinus*, *O. edulis* and *L. digitata*, favourable for IMTA techniques at least in one quarter. These results might be explained by nutrient rich water layers due to river inflows. IMTA techniques bring along a number of advantages such as better growth rates of *Laminaria* species cultured near fish farms (Handå et al., 2012; Handå et al., 2013). Nevertheless, seaweed cultures need a large space at the ocean's surface, whereas mussels have to be cultured in high numbers if they shall function as biofilters to remediate particles out of the water column (Troell et al., 2009) and to ensure environmental balance or economic benefits. Limitations might be possible regarding aquaculture technologies such as IMTA constructions or alterations needed concerning the OWF structures. Within this study, as described in section 3.3.4, all OWF areas have been included, even those already at work. Furthermore, we included all depth levels given in Table 1, although the cultivation at different water depths will require different technologies, some of them more feasible than others. Information about offshore installations, alterations required or other further details might be provided by (Buck et al., 2006), (Buck and Buchholz, 2004), (Benassai et al., 2014) or (Troell et al., 2009).

The potential of a site for co-location depends on biological, ecological and hydrological factors. Furthermore, commercial, legal and social factors have to be addressed (Christie et al., 2014). The present GIS-MCE modelling approach is a first step to analyse potential synergies within the German EEZ of the North Sea. Subsequent steps need to comprise (1) an analysis of profitability on coupling offshore IMTA candidates, (2) the assessment of the environmental carrying capacity, (3) an environmental impact assessment, (4) the analysis of the economic viability of co-locations, (5) the analysis of co-management strategies (e.g. (Buck et al., 2004)) and (6) an integrated assessment process of the German

MSP concerning measures to grant facilities for volunteering co-location (OWF and IMTA) developers in comparison to mono-use OWF developers.

Following the biological site-selection presented in this paper, next steps comprise the analysis of the economic viability and the analysis on the integration of the co-location concept in existing maritime spatial planning processes.

3.6 Conclusion

The resulting suitability scores reveal several possible sets of seaweed, bivalves and fish candidates, favourable for IMTA techniques at least in one quarter. The present results illustrate how competing needs might be balanced in planning for both offshore wind energy and offshore IMTA in the German EEZ of the North Sea.

In conclusion, the GIS-based framework is a suitable tool to analyse synergies regarding space issues among user groups, to offer guidance to stakeholders and assist decision-makers in determining the most suitable sites for pilot projects using IMTA techniques. The co-location of OWFs in combination with offshore IMTA systems might be seen as a milestone towards sustainable MSP, ensuring the continuity of aquatic resources for future generations, however, final decisions still need to be made by decision makers.

3.7 Acknowledgements

The German Federal Office for Agriculture and Food (Bundesanstalt für Landwirtschaft und Ernährung, BLE) funded this study, which is a part of the project *Offshore Site Selection* (OSS) (313-06.01-28-1-73.010-10). The data were freely provided by National

Oceanographic and Atmospheric Administration (NOAA), Helmholtz-Zentrum Geesthacht - Centre for Materials and Coastal Research (HZG) and the German Federal Maritime and Hydrographic Agency (Bundesamt für Seeschifffahrt und Hydrographie, BSH), Hamburg (Germany) in raw, uninterpreted form.

3.8 References

Al-Yahyai, S., Charabi, Y., Gastli, A., and Al-Badi, A. 2012. Wind farm land suitability indexing using multi-criteria analysis. Renewable Energy, 44: 80-87.

Barrington, K., Ridler, N., Chopin, T., Robinson, S., and Robinson, B. 2008. Social aspects of the sustainability of integrated multi-trophic aquaculture. Aquaculture International, 18: 201-211.

Benassai, G., Mariani, P., Stenberg, C., and Christoffersen, M. 2014. A Sustainability Index of potential co-location of offshore wind farms and open water aquaculture. Ocean & Coastal Management, 95: 213-218.

Benassai, G., Stenberg, C., Christoffersen, M., and Mariani, P. 2011. A sustainability index for offshore wind farms and open water aquaculture. Second international Conference on physical coastal processes, management and engineering, Naples, Italy, 27 to 29 April 2011.

BfN Federal Agency for Nature Conservation (Bundesamt für Naturschutz). Natura 2000 Marine Protected Areas in the EEZ. 2014, http://www.bfn.de/0314_meeresschutzgebiete.html.

Boroushaki, S., and Malczewski, J. 2008. Implementing an extension of the analytical hierarchy process using ordered weighted averaging operators with fuzzy quantifiers in ArcGIS. Computers and Geosciences, 34: 399-410.

BSH 2009. The Federal Agency for Shipping and Hydrography (Bundesamt für Seeschifffahrt und Hydrographie, BSH). Map North Sea, http://www.bsh.de/en/Marine_uses/Spatial_Planning_in_the_German_EEZ/documents2/MSP_DE_NorthSea.pdf.

Buchholz, C. M., Krause, G., and Buck, B. H. 2012. Seaweed and Men. Seaweed Ecophysiology & Ecology by C. Wiencke & K. Bischof (Eds.), Ecological Studies of Springer.

Buck, B. H. 2007. Experimental trials on the feasibility of offshore seed production of the mussel Mytilus edulis in the German Bight: installation, technical requirements and environmental conditions. Helgoland Marine Research, 61: 87-101.

Buck, B. H., Berg-Pollack, A., Assheuer, J., Zielinski, O., and Kassen, D. 2006. Technical realization of extensive aquaculture constructions in offshore wind farms: consideration of the mechanical loads, Proceedings of the 25th International Conference on Offshore Mechanics and Arctic Engineering, OMAE 2006. 25th International Conference on Offshore Mechanics and Arctic Engineering, 4-9 June 2006, Hamburg, Germany, Ocean, Offshore, and Arctic Engineering, ASME. New York, NY: American Society of Mechanical Engineers.

Buck, B. H., and Buchholz, C. M. 2004. The offshore-ring: A new system design for the open ocean aquaculture of macroalgae. Journal of Applied Phycology, 16: 355-368.

Buck, B. H., and Krause, G. 2012. Integration of Aquaculture and Renewable Energy Systems. Encyclopaedia of Sustainability Science and Technology, 1: 511-533.

Buck, B. H., and Krause, G. 2013. Short Expertise on the Potential Combination of Aquaculture with Marine-Based Renewable Energy Systems. SeaKult-Sustainable Futures in the Marine Realm. Expertise für das WBGU-Hauptgutachten „Welt im Wandel: Menschheitserbe Meer“. Wissenschaftlicher Beirat der Bundesregierung Globale Umweltveränderungen (WBGU). 58 pp.

Buck, B. H., Krause, G., Michler-Cieluch, T., Brenner, M., Buchholz, C. M., Busch, J. A., Fisch, R., et al. 2008. Meeting the quest for spatial efficiency: Progress and Prospects of Extensive Aquaculture within Offshore Wind Farms. Helgoland Marine Research, 62: 269-281.

Buck, B. H., Krause, G., and Rosenthal, H. 2004. Extensive open ocean aquaculture development within wind farms in Germany: the prospect of offshore co-management and legal constraints. Ocean & Coastal Management, 47: 95-122.

Chambers, M., and Howell, W. 2006. Preliminary information on cod and haddock production in submerged cages off the coast of New Hampshire, USA. ICES Journal of Marine Science, 63: 385-392.

Christie, N., Smyth, K., Barnes, R., and Elliott, M. 2014. Co-location of activities and designations: A means of solving or creating problems in marine spatial planning? Marine Policy, 43: 254-263.

Eastman, J. R. 2001. Guide to GIS and Image Processing. Idrisi Production, 2: 1 - 141.

EC 2014a. European Commission. Maritime Affairs. Policy. Blue growth., http://ec.europa.eu/maritimeaffairs/policy/blue_growth/.

EC 2014b. European Commission. Environment. Marine directive. Good Environmental Status., http://ec.europa.eu/environment/marine/good-environmental-status/index_en.htm.

ElMahdi, A., and Kheireldin, K. 2004. GIS and Mulit-Criteria Evaluation for Integrated Water Resources. Conserving Soil and Water for Society: Sharing Solutions, Brisbane, July 2004, Paper No. 1014: 1 - 7.

FAO 2014. Food and Agriculture Organization of the United Nations. The State of World Fisheries and Aquaculture. Rome: 223 pp.

Fock, H. O. 2011. Natura 2000 and the European Common Fisheries Policy. Marine Policy, 35: 181-188.

Gimpel, A., Stelzenmuller, V., Cormier, R., Floeter, J., and Temming, A. 2013. A spatially explicit risk approach to support marine spatial planning in the German EEZ. Marine environmental research, 86: 56-69.

Godfray, H. C., Beddington, J. R., Crute, I. R., Haddad, L., Lawrence, D., Muir, J. F., Pretty, J., et al. 2010. Food security: the challenge of feeding 9 billion people. Science, 327: 812-818.

Gorsevski, P. V., Donevska, K. R., Mitrovski, C. D., and Frizado, J. P. 2012. Integrating multi-criteria evaluation techniques with geographic information systems for landfill site selection: a case study using ordered weighted average. Waste Management, 32: 287-296.

Grote, B., and Buck, B. H. 2014. The IMTA-approach for nutrient balanced aquaculture: Evaluating the potential of North Sea species from onshore RAS to offshore environments. Aquaculture (in review).

Handå, A., Forbord, S., Wang, X., Broch, O. J., Dahle, S. W., Størseth, T. R., Reitan, K. I., et al. 2013. Seasonal- and depth-dependent growth of cultivated kelp (Saccharina latissima) in close proximity to salmon (Salmo salar) aquaculture in Norway. Aquaculture, 414-415: 191-201.

Handå, A., Min, H., Wang, X., Broch, O. J., Reitan, K. I., Reinertsen, H., and Olsen, Y. 2012. Incorporation of fish feed and growth of blue mussels (Mytilus edulis) in close proximity to salmon (Salmo salar) aquaculture: Implications for integrated multi-trophic aquaculture in Norwegian coastal waters. Aquaculture, 356-357: 328-341.

Jiang, H., and Eastman, J. R. 2000. Application of fuzzy measures in multi-criteria evaluation in GIS. International Journal of Geographical Information Science, 14: 173-184.

Joschko, T. J., Buck, B. H., Gutow, L., and Schröder, A. 2008. Colonization of an artificial hard substrate by Mytilus edulis in the German Bight. Marine Biology Research, 4: 350-360.

Kaiser, M. J., Yu, Y., and Snyder, B. 2010. Economic feasibility of using offshore oil and gas structures in the Gulf of Mexico for platform-based aquaculture. Marine Policy, 34: 699-707.

Maar, M., Bolding, K., Petersen, J. K., Hansen, J. L. S., and Timmermann, K. 2009. Local effects of blue mussels around turbine foundations in an ecosystem model of Nysted off-shore wind farm, Denmark. Journal of Sea Research, 62: 159-174.

Malczewski, J. 2006. Ordered weighted averaging with fuzzy quantifiers: GIS-based multicriteria evaluation for land-use suitability analysis. International Journal of Applied Earth Observation and Geoinformation, 8: 270-277.

Michler-Cieluch, T., Krause, G., and Buck, B. H. 2009. Reflections on integrating operation and maintenance activities of offshore wind farms and mariculture. Ocean & Coastal Management, 52: 57-68.

Neori, A., Troell, M., Chopin, T., Yarish, C., Critchley, A., and Buschmann, A. H. 2007. The need for a balanced ecosystem approach to blue revolution aquaculture. Environment, 49: Research Library Core, pg. 37.

Perera, A. H., Drew, C. A., and Johnson, C. J. 2011. Expert Knowledge and Its Application in Landscape Ecology. Springer Science & Business Media, 322 pp.

Perez, O. M., Telfer, T. C., and Ross, L. G. 2005. Geographical information systems-based models for offshore floating marine fish cage aquaculture site selection in Tenerife, Canary Islands. Aquaculture Research, 36: 946-961.

Pogoda, B., Buck, B. H., and Hagen, W. 2011. Growth performance and condition of oysters (Crassostrea gigas and Ostrea edulis) farmed in an offshore environment (North Sea, Germany). Aquaculture, 319: 484-492.

Rosenthal, H., Costa-Pierce, B., Krause, G., and Buck, B. 2012. Bremerhaven Declaration on the Future of Global Open Ocean Aquaculture - Part II: Recommendations on Subject Areas and Justifications. Aquaculture Forum on Open Ocean Aquaculture Development - From visions to reality: the future of offshore farming. Funded by: Investment in sustainable fisheries co-financed by the European Union (European Fisheries Fund - EFF), Ministry of Economics, Labour and Ports (Free Hanseatic City of Bremen), The Bremerhaven Economic Development Company Ltd., 8.

Soma, K., Ramos, J., Bergh, O., Schulze, T., van Oostenbrugge, H., van Duijn, A. P., Kopke, K., et al. 2013. The "mapping out" approach: effectiveness of marine spatial management options in European coastal waters. ICES Journal of Marine Science, 71: 2630-2642.

Stelzenmüller, V., Schulze, T., Fock, H. O., and Berkenhagen, J. 2011. Integrated modelling tools to support risk-based decision-making in marine spatial management. Marine Ecology Progress Series, 441: 197-212.

Stelzenmüller, V., Schulze, T., Gimpel, A., Bartelings, H., Bello, E., Bergh, O., Bolman, B., et al. 2013. Guidance on a Better Integration of Aquaculture, Fisheries, and other Activities in the Coastal Zone: From tools to practical examples. COEXIST project: 79pp.

Troell, M., Chopin, T., Angel, D., and Buck, B. H. in review. IMTA and offshore aquaculture development. Aquaculture Engineering.

Troell, M., Joyce, A., Chopin, T., Neori, A., Buschmann, A. H., and Fang, J.-G. 2009. Ecological engineering in aquaculture — Potential for integrated multi-trophic aquaculture (IMTA) in marine offshore systems. Aquaculture, 297: 1-9.

UNH 2014. University of New Hampshire. Atlantic marine aquaculture center., http://ooa.unh.edu/publications/publications_briefs.html.

Weisse, R., and Plüβ, A. 2005. Storm-related sea level variations along the North Sea coast as simulated by a high-resolution model 1958–2002. Ocean Dynamics, 56: 16-25.

Wever, L., Krause, G., and Buck, B. H. 2015. Lessons from stakeholder dialogues on marine aquaculture in offshore wind farms: Perceived potentials, constraints and research gaps. Marine Policy, 51: 251-259.

4. Manuscript 3: Quantitative environmental risk assessments in the context of marine spatial management: Current approaches and some perspectives

Vanessa Stelzenmüller[a], Heino O. Fock[a], Antje Gimpel[a], Henrike Rambo[a], Rabea Diekmann[b], Wolfgang N. Probst[a], Ulrich Callies[c], Frank Bockelmann[c], Hermann Neumann[d], Ingrid Kröncke[d]

[a] Thünen-Institute of Sea Fisheries, Palmaille 9, 22767 Hamburg, Germany

[b] Thünen-Institute of Fisheries Ecology, Palmaille 9, 22767 Hamburg, Germany

[c] Helmholtz-Zentrum Geesthacht, Centre for Materials and Coastal Research, Max-Planck-Straße 1, 21502 Geesthacht, Germany

[d] Senckenberg am Meer, Südstrand 40, 26382 Wilhelmshaven, Germany

ICES Journal of Marine Science (2015), 72(3), 1022–1042

4.1 Abstract

Marine spatial planning (MSP) requires spatially explicit environmental risk assessment (ERAs) frameworks with quantitative or probabilistic measures of risk, enabling an evaluation of spatial management scenarios. ERAs comprise the steps of risk identification, risk analysis and risk evaluation. A review of ERAs in in the context of spatial management revealed a synonymous use of the concepts of risk, vulnerability and impact, a need to account for uncertainty and a lack of a clear link between risk analysis and risk evaluation. In a case study we addressed some of the identified gaps and predicted the risk of changing the current state of benthic disturbance by bottom trawling due to future MSP measures in the German EEZ of the North Sea. We used a quantitative, dynamic and spatially explicit approach where we combined a Bayesian belief network (BN) with GIS to showcase the steps of risk characterisation, risk analysis and risk evaluation. We distinguished ten benthic communities and six international fishing fleets. The risk analysis produced spatially explicit estimates of benthic disturbance, which was computed as a ratio between relative local mortality by benthic trawling and the recovery potential after a trawl event. Results showed great differences in spatial patterns of benthic disturbance when accounting for different environmental impacts of the respective fleets. To illustrate a risk evaluation process, we simulated a spatial shift of the international effort of two beam trawl fleets, which are affected the most by future offshore wind development. The BN model was able to predict the proportion of the area where benthic disturbance likely increases. In conclusion, MSP processes should embed ERA frameworks which allow for the integration of multiple risk assessments and the quantification of related risks as well as uncertainties at a common spatial scale.

Keywords: Bayesian belief network, fishing frequency, GIS, marine spatial planning, review

4.2 Introduction

Place-based management tools such as marine spatial planning (MSP) are advocated worldwide to support the implementation of an ecosystem approach to marine management (Katsanevakis et al., 2011). In Europe, MSP is regarded as a means to solve inter-sectoral and cross-border conflicts over maritime space (Douvere and Ehler, 2010) and is promoted by the upcoming EU MSP Directive (Commission, 2014). The latter encourages blue growth and the sustainable use of marine resources (Qiu and Jones, 2013; Brennan et al., 2014). One of the future challenges for European regional Seas is the alignment of the sustainable use of the marine resources with the maintenance of ecosystem health and functioning, as demanded by the EU Marine Strategy Framework Directive (MSFD) (Commission, 2008). Hence, an ecosystem based MSP process should seek to manage human activities while balancing multiple ecological, economic and social objectives (Foley et al., 2013).

As a consequence, an ecosystem based MSP approach requires robust estimates of the risks of adverse effects of cumulative human pressures on the marine environment at meaningful ecological scales (Eastwood et al., 2007; Halpern et al., 2008a; Stelzenmüller et al., 2010; Fock et al., 2011). Environmental risk assessments (ERAs) (Hope, 2006) that link spatially explicit information on the vulnerability of ecosystem components with the occurrence and magnitude of pressures are fundamental for the successful implementation of an ecosystem based MSP approach. The fast growing number of MSP initiatives (Carneiro, 2013; Collie et al., 2013) highlights the increasing importance of spatially explicit ERAs and underpins the need for quantitative or probabilistic measures of risk.

In general, quantitative risk assessments rely on mathematical models to predict the response of the ecosystem component to changing pressures. Qualitative approaches, however, use ecosystem attributes combined with ecological receptors and stressors

(Astles et al., 2006). As for today, empirical studies on ERAs that provide, for example, spatially explicit quantifications of risk in relation to management options appear at a slower pace and take various risk assessment approaches (Stelzenmüller et al., 2010; Fock et al., 2011; Gimpel et al., 2013; Redfern et al., 2013). In the light of existing EU policies, in particular the MSFD and new MSP Directive, there is a growing need to align various spatially explicit ERAs to ongoing spatial management processes.

To account for this we adopted the risk assessment framework described in Cormier et al. (2013) to first, assess current ERA approaches and second, structure a case study on the risk of benthic disturbance in the German EEZ of the North Sea. The risk assessment framework comprises three steps. First, the risk identification specifies the pressure(s) of concern and the significant ecosystem components. Second, the risk analysis accounts for both, the probability and the magnitude of the pressure, its impacts on ecosystem components, and the degree of uncertainty involved. Third, the risk evaluation assesses the likely impacts on ecosystem components under alternative management measures.

We first reviewed empirical studies of spatially explicit and quantitative ERAs in the context of spatial management and assessed in detail the methods used for the risk identification, risk analysis and risk evaluation. To address some identified methodological gaps we defined a case study which describes the stepwise assessment of the risk when changing the current state of benthic disturbance by trawling due to future MSP measures in the German EEZ. Thus in the risk identification step we defined the offshore wind development and the related displacement of fishing effort as pressures. We identified ten benthic communities as described by Rachor and Nehmer (2003) as an example of significant ecosystem components since the good environmental status of seabed integrity reflects one of the goals of the MSFD. In the risk analysis step we computed spatial estimates of a benthic disturbance indicator (Fock, 2011a), which was defined as a ratio between relative local mortality by demersal trawling fleets and recovery potential of benthic communities (see Hiddink et al., 2006a). For the risk evaluation we used a spatially explicit probabilistic approach that allows a dynamic assessment of possible trade-offs of alternative spatial management scenarios. We coupled a Bayesian belief network (BN) with GIS and predicted occurrence probabilities of different states of benthic disturbance and % changes of the study area in relation to simulated spatial management objectives. BNs are acyclic graphs that represent causal dependencies among a set of random variables by means of directed links between them (McCann et al., 2006). Recently, they have been used in combination with GIS to conduct a spatially explicit assessment of the risk involved with spatial management options (Stelzenmüller et al., 2011; Johnson et al., 2012; Grêt-Regamey et al., 2013a; Grêt-Regamey et al., 2013b). In summary, here we identified some shortcomings of current spatially explicit ERA approaches, and showed some perspectives for assessing trade-offs of MSP scenarios in the German EEZ of the North Sea. Finally, we reflected on the challenges ahead when it comes to the integration of numerous assessment outputs in a multiple objectives spatial management context.

4.3 Material and methods

4.3.1 Risk assessment framework and review of current approaches

We adopted the standardised risk assessment framework defined by Cormier et al. (2013) to frame the steps of risk identification, risk analysis and risk evaluation in a spatial management context (Figure 1). We then analysed recent empirical studies of (semi-) quantitative environmental risk assessments in the context of marine spatial management with regard to these key steps. Here spatial management was rather broadly defined and encompassed studies concerned with MSP, sectoral management or marine conservation. With the help of multiple combinations of the key words: environmental risk assessment, risk analysis, quantitative, vulnerability, spatial management, marine spatial planning, and map(ping) we selected a total of 32 peer-reviewed papers. In the following we describe the three risk assessment steps in more detail and specify what information has been extracted from the reviewed literature.

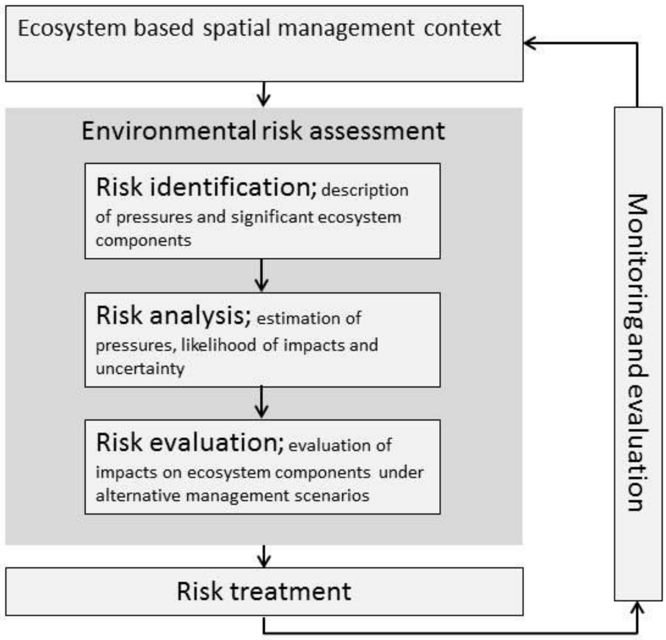


Figure 1: Simplified risk management process redrawn from (Cormier and al., 2013) in the context of marine spatial management such as MSP. Spatial management goals and operational objectives (Stelzenmüller et al., 2013) determine the contents of the environmental risk assessment. Risk assessment results enter the risk treatment phase which produces management options, based on cost-benefit analysis of implementation. Suggested management options will in turn feedback in to the spatial management process (development, implementation or evaluation process).

Risk identification - The risk identification comprises the definition of significant ecosystem components, stressors or pressures as well as the related environmental cause-effect pathways defined by the operational management objectives for a given area.

Operational objectives have specific, measureable, achievable, realistic and time limited (SMART) targets, such that management measures can be fitted and performance can be evaluated (Stelzenmüller et al., 2013). Stressors are single or multiple human pressures while cumulative impacts are described as the combined impact of multiple pressures over space and time (MacDonald, 2000). Here risk identification comprises also an estimate of the occurrence probability and magnitude of the pressure and the spatial quantification of the identified ecosystem components or state indicator. According to this definition, the assessed pressures and ecosystem components or state indicators together with the methods used to quantify their occurrence in the respective area were extracted from the reviewed empirical studies.

Risk analysis - This step addresses the quantification of impacts on ecosystem components that accounts for existing mitigation or management measures as well as the risk acceptance in society. The latter should be reflected in the operational management objectives. The impact is generally defined as a function of the vulnerability of ecosystem components and the occurrence likelihood and magnitude of a pressure (Stelzenmüller et al., 2010). De Lange et al. (2010) proposed to define vulnerability of an ecosystem component by means of exposure and sensitivity to a pressure as well as its recovery potential. The sensitivity to a pressure is due to structural properties, functions or trophic relations of the ecosystem component while recovery depends on population recovery, resilience, positive feedback loops and adaption (Tyler-Walters et al., 2001; Hope, 2006; Halpern et al., 2008b). We classified each case study according to the type of sensitivity measure used (expert knowledge, model output, empirical data) and the vulnerability assessment approach applied. Uncertainty should be recognised and constructively handled for any integrated risk assessment or models based decision support (Rotmans and van Asselt, 2001). For instance a recent review by Ferdous et al. (2013) assessed methods which allow recognising and evaluating the implications of uncertainty in a risk analysis.

Thus we reported further if any form of uncertainty analysis was undertaken and which methods have been used.

Risk evaluation - The result of a risk evaluation indicates whether or not new management actions need to be taken. Technically, this requires the evaluation of management scenarios, including the "the business as usual" scenario. More precisely, it entails a comprehensive assessment of the proposed management measures and scenarios with respect to the potential risks for relevant ecosystem components. Thus we investigated what kind of management scenarios, if at all, have been tested in the empirical studies.

4.3.2 Case study area and context

The here described risk assessment framework has been hardly applied to marine ecosystems in all aspects. We thus designed a case study assessing future MSP measures in the German EEZ and their likely implications for benthic communities using a quantitative, dynamic and spatially explicit approach. Since 2008 the maritime spatial plan is legally binding in the German EEZ and comprises designated preference areas for a number of sectors except fishing, including special areas of conservation designated under the Habitat Directive (92/43/EEC, 1992; Natura2000 sites) (BMVBS, 2009) (Fock, 2011b; Stelzenmüller et al., 2011; Gimpel et al., 2013). Further environmental objectives with potential spatial management measures are defined by the MSFD and require implementation by 2020. For illustration purposes we simplified this rather complex spatial management context and focused only on seabed integrity and defined the hypothetical operational management objective "*The relative benthic disturbance by trawling should not deteriorate with respect to current levels*". This operational objective defines the impact of trawling on benthic communities as the measure or indicator of concern and specifies the current level as the reference point. Therefore future MSP measures, which comprise the designation of offshore wind development sites within approx. 35 % of the study area, will be assessed against the here defined management objective. Future offshore wind development sites in the German EEZ show a clear spatial overlap with prevailing patterns of fishing (Stelzenmüller et al., 2011). Thus the potential area loss for fishing will most likely result in an effort displacement with as yet unknown environmental and economic consequences. In the following we describe the risk assessment steps for the current case study.

4.3.2.1 Risk identification - Offshore wind development, fisheries and benthic communities

We considered the currently designated offshore wind development sites as MSP measures as well as the submitted application areas. The development of this sector triggers a number of conflicts with other human uses through the competition for the same space (Gimpel et al., 2013). The highest conflict potential can be expected between the (international) fishing sector and the offshore wind development, since e.g. roughly 15 % of the total international large beam trawl effort takes place in areas where offshore wind development is envisaged. Thus we defined the average spatial and temporal activity of six different fishing fleets as pressures following Fock (2011) and Stelzenmüller et al. (2011) regarding to seabed integrity (as specified above). For this we combined German, Dutch and Danish VMS (vessel monitoring system) and logbook data from 2005 to 2008 to calculate the average bottom trawling effort (total hours fishing per year) per 3 x 3 nm grid cell (31 km²). We distinguished six different fleets, which are beam trawlers operating with 80 mm mesh size and an engine power > 221KW (Beam80lrg) and < 221 KW (Beam80sml), beam trawler with 16 to 31 mm mesh size and an engine power > 221 KW (Beam1631lrg) and < 221 KW (Beam1631sml), and otter trawlers with 80 mm mesh size and an engine power > 221 KW (Otter80lrg) and < 221 KW (Otter1631sml). For each grid cell we computed the frequency with which the seabed surface has been swept by the respective fleet (Ffr_{ik}) using the formula and parameters also presented in Fock (2011a) ($Ffr_{ik} = \frac{T_{ik} * V_k * w_k}{A_i}$; with T_{ik}=total hours fished (h), V_k = average fishing speed (km/h), w_k = net spread (km), and Ai= surface area in km^2). The ecosystem components of concern were ten benthic communities with a defined spatial distribution (Figure 2) and specific characteristics such as habitat preference or recovery frequency (Table 1) (Rachor and Nehmer, 2003; Pesch et al., 2008; Fock, 2011a). Thus with the help of GIS we allocated to each grid cell the most dominant benthic community with respective measures of recovery potential and mortality rates (see below) together with the average fishing frequency per fleet.

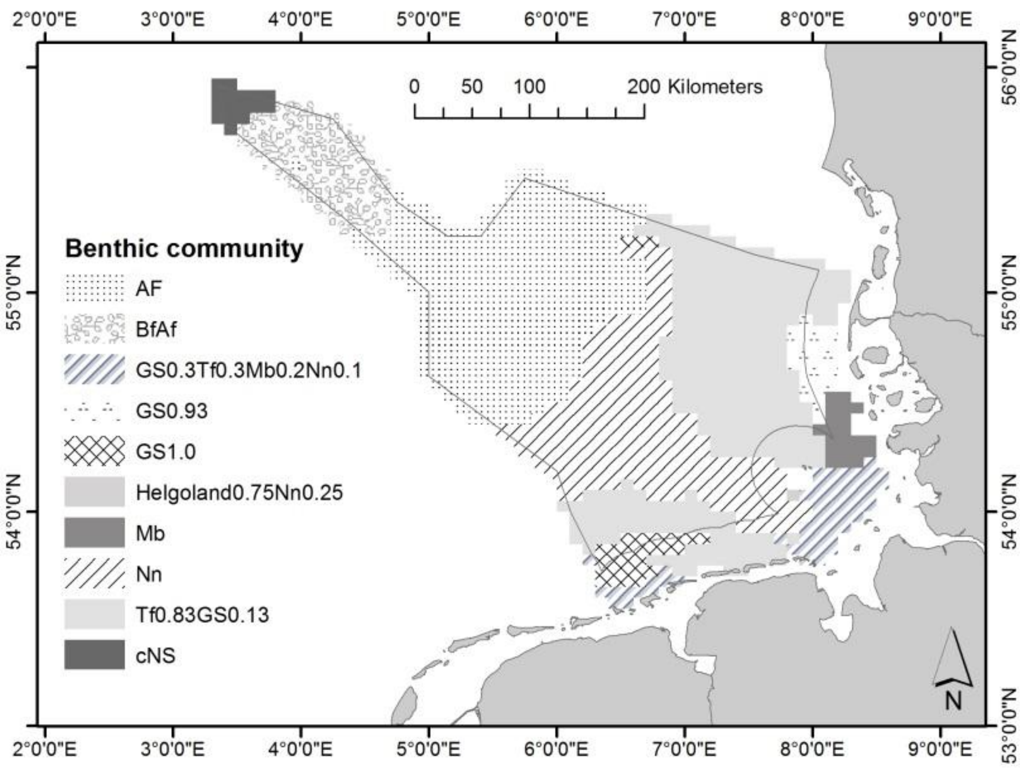


Figure 2: Predicted spatial distribution of the infaunal benthic community in the German EEZ of the North Sea and adjacent waters (redrawn after (Pesch et al., 2008).

4.3.2.2 Risk analysis – Measuring benthic disturbance

The next step required the definition of vulnerability of the ecosystem components to fishing pressures exerted by the different fleets. We built on a previous study (Fock, 2011a and references therein) and computed spatial estimates of the disturbance indicator (DI). DI_i reflects an overall relative local vulnerability of a benthic community to bottom trawling and is defined as the ratio between mortality and recovery (M_i/R_i). DI_i is a unitless relative ratio and $DI_i = 1$ indicates a balance between relative local mortality and recovery. $DI_i > 1$ indicates locally higher mortality rates than recovery potential, whereas $DI_i < 1$ indicates that the recovery potential exceeds local mortality rates by trawling.

The computation of this ratio requires relative estimates of recovery time and recovers frequency for each of the ten benthic communities (see Table 1). We used the proportion of typical sediment categories (mud, sand, muddy sand, and gravel) favoured by the respective benthic communities (Rachor and Nehmer, 2003) to construct combined relative measures of recovery time (y) ($RT_{BC} = \sum R_{Sediment} \cdot$ Proportion sediment) and recover frequency (y^{-1}) ($Rfr_{BC} = \sum Rfr_{Sediment} \cdot$ Proportion sediment), both in relation to one trawling event. With this we computed for each grid cell the relative recovery for each benthic community to 90 % of the abundance previous to trawling as a function of the recovery time and recover frequency $R_i = 1 - (1 - 0.9 \cdot RT_{BC})^{RfrBC}$ (Fock, 2011a). Hence, the here applied measure of sensitivity to benthic trawling is derived from model outputs presented in Hiddink et al. (2006a) and empirical results by Rachor and Nehmer (2003). In a next step, we computed for each grid cell the local mortality rate for each benthic community. For this we used the average percentage decline of abundance per sediment type (taken from Fock 2011a) to construct an average combined measure of mortality per benthic community ($MR_{BC} = \sum Decline_{Sediment} \cdot$Proportion sediment) (see Table 1). Accordingly, we computed for each grid cell the fleet specific mortality rate for the benthic community as $M_{ik} = 1 – (1 - MR_{BC})^{Ffrik}$. The overall local mortality rate is the sum of these mortality rates weighted by a respective impact sore (*is*); $M_i = \sum_{k=1}^{n} M_{ik} * is_k$ (modified after Fock, 2011a). This finally allowed us to compute the ratio between relative local mortality and recovery (M_i/R_i), and we refer to this as disturbance indicator (DI_i). We further explored the uncertainty within the estimates of benthic disturbance by accounting for fleet specific impacts on benthic communities. For that reason we calculated DI_i based on a local overall mortality rate (M_i) by assuming equal impacts of each fleet (i.e. impact score $is_k = 1$). Alternatively, we computed DI_{iw} with a local overall mortality rate weighted by different impact scores (adapted from Fock 2011a). Here highest weight is given to the beam trawlers operating with a mesh size of >80mm, which represent mainly the fishery targeting flatfish, and least weight is given to the small beam trawlers using mesh sizes of 16-31mm, representing the shrimp fishery ($is_{BEAM80lrg} = 1$; $is_{BEAM80sml} = 1$; $is_{BEAM1631lrg} = 0.1$; $is_{BEAM1631sml} = 0.1$; $is_{OTTER80lrg} = 0.15$; $is_{OTTER80sml} = 0.15$). We compiled for each grid cell the respective measures of recovery, mortality and benthic disturbance in ArcGIS 10.0 using the attribute table of the vector grid for subsequent mapping. Thus, DI and DI_w describe spatially disaggregated alternative assumptions of the relative state of benthic disturbance, based on the average bottom trawling effort from 2005-2008.

Table 1: Ten benthic communities as defined by Rachor and Nehmer (2003) comprising *Amphiura filiformis* 89% (AF); *Bathyporeia fabulina* 85%, *Amphiura filiformis* 10% (BtAf); central North Sea (cNS); *Tabulina fabula* (Tf) 83%, *Goniadella spisula* (GS) 12,5% (Tf0.83GS0.13); GS30%,Tf30%, *Macoma balthica* (Mb) 20%, *Nucula nitidosa* (Nn) 10% (GS0.3Tf0.3Mb0.2Nn0.1); GS 100% (GS1.0); GS 93% (GS0.93); Helgoland Depth 75%, Nn 25% (Helgoland0.75Nn0.25); Mb 100% (Mb); Nn 84% (Nn). For each community the relative distribution on four different sediment types, their sediment specific recovery time (R), recover frequency (Rfr) and decline after one trawling event (Decline) is given (after Fock 2011a; Hiddink et al. 2006a). Further, the community specific combined values are listed as relative combined recovery time (RT_{BC}), the relative combined recover frequency (Rfr_{BC}), the relative combined recovery rate (R_{BC}), and the relative combined abundance decline after one trawling event (MR_{BC}).

Benthic community	**AF**	**BtAf**	**cNS**	**Tf0.83 GS0.13**	**GS0.3Tf0.3Mb0.2 Nn0.1**	**GS1.0**	**GS0.93**	**Helgoland 0.75Nn0.25**	**Mb**	**Nn**
Prop mud^{+}					0.11			0.8		0.84
Prop muddy sand^{+}	1	0.15	0.5		0.28					0.16
Prop sand^{+}		0.85	0.5	0.93	0.44	0.5	0.6	0.15	0.50	
Prop gravel^{+}				0.07	0.16	0.5	0.4	0.05	0.50	
R_{Mud} (days)	25	25	25	25	25	25	25	25		
$R_{MuddySand}$ (days)	111	111	111	111	111	111	111	111	111	111
R_{Sand} (days)	193	193	193	193	193	193	193	193	193	193
Rfr_{Mud} (y^{-1})	14	14	14	14	14	14	14	14	14	14
$Rfr_{MuddySand}$ (y^{-1})	3	3	3	3	3	3	3	3	3	3
Rfr_{Sand}(y^{-1})	1.5	1.5	1.5	1.5	1.5	1.5	1.5	1.5	1.5	1.5
Rfr_{Gravel} (y^{-1})	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
$Decline_{Mud}$ (proportion)	0.345	0.345	0.345	0.345	0.345	0.345	0.345	0.345	0.345	0.345
$Decline_{MuddySand}$ (proportion)	0.675	0.675	0.675	0.675	0.675	0.675	0.675	0.675	0.675	0.675
$Decline_{Sand}$ (proportion)	0.535	0.535	0.535	0.535	0.535	0.535	0.535	0.535	0.535	0.535

Benthic community	AF	BtAf	cNS	Tf0.83 GS0.13	GS0.3Tf0.3Mb0.2 Nn0.1	GS1.0	GS0.93	Helgoland 0.75Nn0.25	Mb	Nn
$Decline_{Gravel}$ (proportion)	0.74	0.74	0.74	0.74	0.74	0.74	0.74	0.74	0.74	0.74
RT_{BC} (y)	0.3	0.5	0.42	0.49	0.33	0.26	0.32	0.13	0.26	0.11
Rfr_{BC} (y^{-1})	3	1.73	2.25	1.4	3.06	0.8	0.94	11.5	0.8	12.24
R_{BC}	0.62	0.64	0.65	0.56	0.65	0.2	0.27	0.77	0.2	0.71
MR_{BC} (proportion)	0.68	0.56	0.61	0.55	0.58	0.64	0.62	0.39	0.64	0.40

$^{+}$ The proportion of sediment per benthic community have been derived from (Fock, 2011a) based on a study from (Rachor and Nehmer, 2003)

4.3.2.3 Risk evaluation –Trade-off analysis of MSP measures

This final step corresponds to the evaluation of the risk of worsening the current state of benthic disturbance due to future MSP measures in the German EEZ. Our scenario applies to planned offshore wind development sites, where, in case of their realisation, extensive areas would be closed for fishery. As a rough estimate 15% of the large beam trawl effort and 3% of the small beam trawl effort would be affected. Effects on the fleets using otter boards are negligible. Thus, we defined the following spatial management scenario: "*Current and future offshore wind development cause a spatial shift of 15 % of the total fishing frequency of large beam trawlers (Beam80lrg) and 3% of the small beam trawlers (Beam1631sml)"*. We combined a Bayesian belief network (BN) with GIS to predict changing likelihoods of benthic disturbance states due to different trawling effort patterns. We used the Netica software system (www.norsys.com) (see details on the inference algorithm implemented in Netica in (Spiegelhalter and Dawid, 1993) to develop the BN model and used the attribute table compiled in the GIS to both built the prior probabilities for each variable (referred to as BN node) and to populate the conditional probability tables (CPTs) (see Table 2). The BN model contains the deterministic relationships described above and reflects the causal links of all parameters required to calculate the unweighted and weighted disturbance indicator (Figure 3). Benthic communities and the fishing frequencies of the six fleets are parent nodes and are considered to be independent from each other. Each parent node has discrete states (e.g. type of benthic community, category of fishing frequency) with an associated probability of occurrence. Fleet specific mortality rates are represented as functions of the respective fishing frequencies and the estimated decline rates for each benthic community. The overall mortality rate and weighted mortality rate are child nodes of the fleet specific mortality rates and are defined by their deterministic relationships with their parent nodes. Recover frequency, recovery time, and abundance decline are child nodes of the benthic communities. The likelihoods of the states of the disturbance indicator nodes are predicted as a function of the likelihood of the overall relative mortality rates (unweighted and weighted) and the predicted recovery by the benthic community.

We also assessed the sensitivity of the disturbance indictor node (DI) to the influence of the parent nodes by calculating the variance reduction. The performance or "goodness of fit" of the BN model was tested by computing the spherical payoff index (see Marcot et al., 2006). The latter describes how well the predictions of the BN match the actual cases and is defined as the mean probability value of a given state averaged over all cases.

Subsequently, we explored the effects of the planned offshore wind development sites on the two measures of benthic disturbance (DI and DI_w) with the help of the trained BN. We assumed that in 15 % of the area the likelihood of experiencing the lowest level of fishing pressures by large beam trawlers will increase (since 15 % of the area will be closed for this fisheries). Assuming that the fishing effort will relocate in areas with already high fishing intensity, the probability of a unit area experiencing the highest level of fishing pressures (or being in state 3) must increase. Thus we changed in the BN model the prior distribution for the Beam80lrg node, with now 47 % of the area having a value from 0 to 0.0025 and in 53 % of the area values range between 0.06 and 1.16. We inferred subsequently the changes of the probability distributions of the DI and DI_w nodes. Based on the same rational we have changed the prior distribution for the Beam1631sml node assuming that in 66 % the area no fishing is carried out by this fleet, while in 12 % of the area values range between >0 and 0.07, and in 22 % of the area values range between <0.07 and 1.17. It is worth mentioning that the here defined spatial shift in fishing effort reflects one out of many possible changes to the prior distributions of the parent nodes reflecting the fishing frequencies of the six fleets.

Table 2: Description of BN model nodes, discretisation method and states. Note: All model nodes reflect attributes from the 3 by 3 nm vector grid.

BN node	**States**	**Description**
Recover_frequency_BC	0 -1.4; >1.4 – 3; >3 – 12.24	Relative combined recover frequency for each benthic community (y^{-1}) (Table 1; $Rfr_{BC} = \sum Rfr_{Sediment} \cdot$ Proportion sediment) from benthic trawling.
Recovery_time_BC	0-0.26; >0.26 – 0.33; >0.33 – 0.5	Relative combined recovery time for each benthic community (y) (Table 1; $RT_{BC} = \sum R_{Sediment} \cdot$ Proportion sediment) from benthic trawling.
Abundance_decline_BC	0-0.5; >0.5-0.58; >0.58-0.68	Relative combined abundance decline after one trawling event for each benthic community (Table 1; $MR_{BC} = \sum Decline_{Sediment} \cdot$Proportion sediment)
Recovery	0-0.56; >0.56-0.62; >0.62-0.78	Relative local recovery rate for each benthic community (Table 1; $R_i = 1- (1 - 0.9 \cdot RT_{BC})^{RfrBC}$)
FrBeam80LR	0-0.0025; >0.0025-0.06; >0.06-1.16	Fleet specific mean (2005 to 2008) fishing frequency ($Ffr_{ik} = \frac{T_{ik} * V_k * w_k}{A_i}$; with T_{ik}=total hours fished (h), V_k = average fishing speed (km/h), w_k = net spread (km), and Ai= surface area in km^2) with which the surface area has been swept (Beam = beam trawlers, Otter = otter board trawlers, 80 = 80 mm mesh size, 1631 = 16 to 31 mm mesh size, LR = engine power > 221KW, SM = engine power < 221KW).
FrBeam80SM	0; >0-0.0004; >0.00004-0.076	
FrBeam1631LR	0; >0-0.00019; >0.000019-0.000347	
FrBeam1631SM	0; >0-0.07; >0.07-1.17	
FrOtter80LR	0; >0-0.000279; >0.000279-0.335	
FrOtter80SM	0-0.0007; >0.0007-0.012; >0.012-0.524	
M_Beam80LR	0-0.0021; >0.0021-0.05; >0.05-0.45	Fleet specific relative mean mortality rates of the prevailing benthic community as a function of the mean frequency of the respective fleet and the combined average abundance decline rate ($M_{ik} = 1 - (1 - MR_{BC})^{Ffrik}$ (Beam = beam trawlers, Otter = otter board trawlers, 80 = 80 mm mesh size, 1631 = 16 to 31 mm mesh size, LR = engine power > 221KW, SM = engine power < 221KW) (see Table 1).
M_Beam80SM	0; >0-0.0007; >0.0007-0.058.	
M_Beam1631LR	0; >0-0.000134; >0.000134-0.00039.	
M_Beam1631SM	0; >0-0.06; >0.06-0.64	
M_Otter80LR	0; >0-0.000313; >0.000313-0.31	
M_Otter80SM	0; >0-0.000313; >0.000313-0.31	
Mortality_rate	0-0.032; >0.032-0.14; >0.14-0.84	Overall mean local mortality rate expressed as the sum of the mean local mortality rates per fleet (from 2005 to 2008) weighted by equal impact scores (*is*): $M_i = \sum_{k=1}^{n} M_{ik} * is_k$; is_k=1

BN node	States	Description
Mortality_rate_W	0-0.032; >0.032-0.14; >0.14-0.84	Overall mean local mortality rate weighted by different impact scores (*is*): $is_{BEAM80lrg}$ = 1; $is_{BEAM80sml}$ = 1; $is_{BEAM1631lrg}$ = 0.1; $is_{BEAM1631sml}$ = 0.1; $is_{OTTER80lrg}$ = 0.15; $is_{OTTER80sml}$ = 0.15
Disturbance_indicator	0-0.3; >0.3-0.5; >0.5-1; >1-3	Estimated disturbance indicator (DI_i) as the ratio between mortality rate and recovery.
Disturbance_indicator_W	0-0.3; >0.3-0.5; >0.5-1 ;>1-3	Estimated disturbance indicator (DI_{iW}) as the ratio between the weighted mortality rate and recovery.
Benthic_communities	AF; BtAf; cNS; Tf0.83GS0.13; GS0.3Tf0.3Mb0.2Nn0.1; GS1.0; GS0.93; Helgoland0.75Nn0.25; Mb; Nn	Ten categories of benthic communities as defined by (Rachor and Nehmer, 2003) comprising *Amphiura filiformis* 89% (AF); *Bathyporeia fabulina* (85%), *Amphiura filiformis* (10%) (BfAf); central North Sea (cNS); *Tabulina fabula* (83%), *Goniadella spisula* (12,5%) (Tf0.83GS0.13); *Goniadella spisula* (30%), *Tabulina fabula* (30%), *Macoma balthica* (20%), *Nucula nitidosa* (10%) (GS0.3Tf0.3Mb0.2Nn0.1); *Goniadella spisula* (100%) (GS1.0); *Goniadella spisula* (93%) (GS0.93); *Helgoland Depth* 75%, *Nucula nitidosa* (25%) (Helgoland0.75Nn0.25); *Macoma balthica* (100%) (Mb); *Nucula nitidosa* (84%) (Nn)

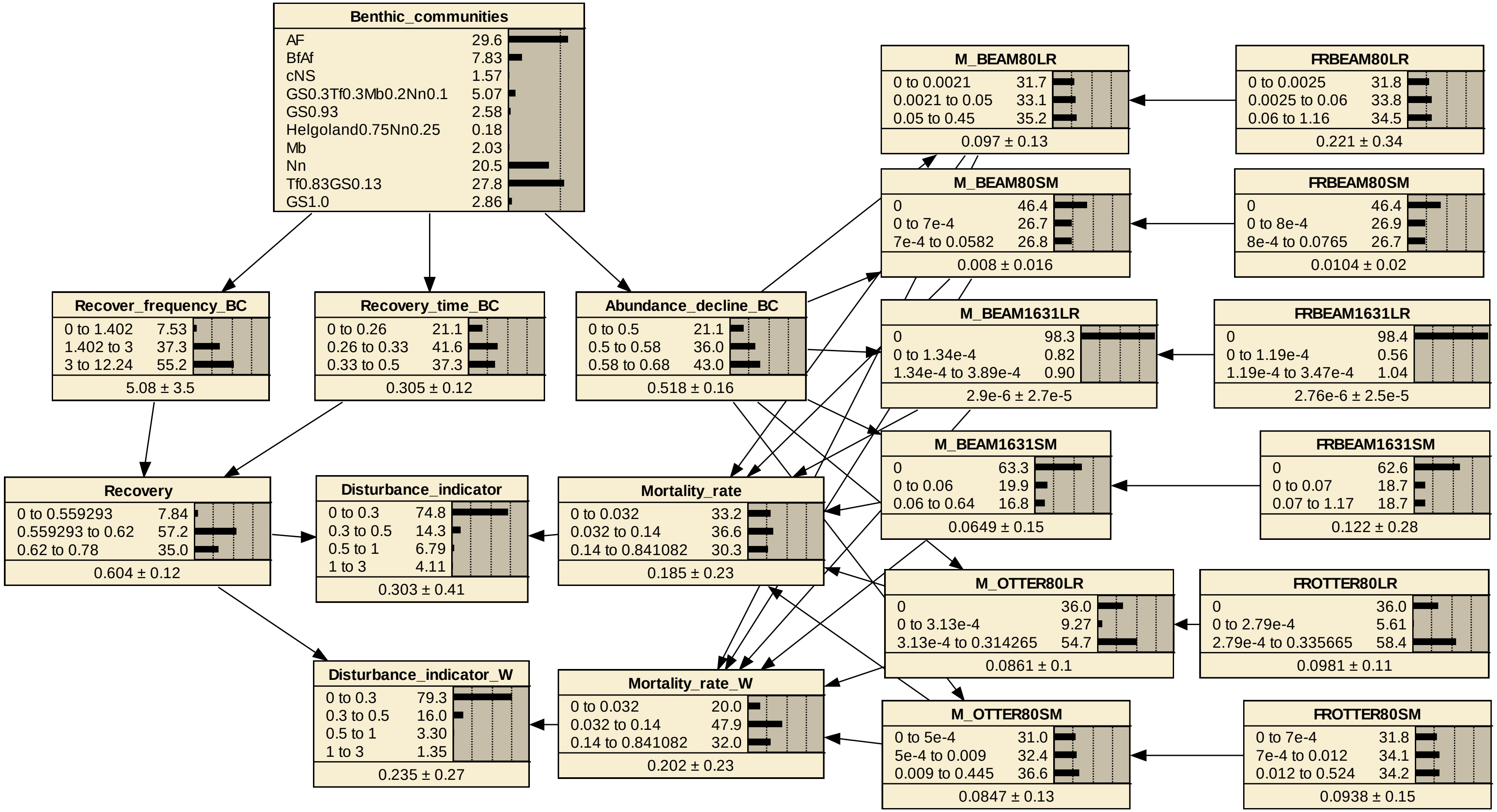


Figure 3: Structure of the Bayesian belief network for assessing future MSP measures in the German EEZ and their likely implications for benthic communities. Values for categorical probabilities (%) of each node are given for the baseline scenario (referred to as "business as usual scenario") (node definitions in Table 2).

4.4 Results

4.4.1 Review of current approaches

The results of the structured literature review of 32 papers are summarised in Table 3. Most studies focused on one or two stressors with a clear emphasis on fisheries; other activities included aggregate mining and marine traffic. Cumulative pressures were analysed in a quarter of all examined studies, mostly assuming additive effects. We observed that the measure of sensitivity of ecosystem components or indicators was mostly related to a metric derived from a model output which based either on empirical data or expert knowledge. In contrast, a quarter of the reviewed studies were based on expert knowledge and three studies being based exclusively on empirical data. Another important result was that the terminology of risk, vulnerability and impact varied greatly across the studies and has been used synonymously. Despite this variation in terminology the components to calculate a measure of vulnerability or impact have been similar across all cases. All studies defined vulnerability or impact as a function of a measure of ecosystem sensitivity and the occurrence probability and magnitude of a stressor or pressure. However, the concepts of resistance and resilience of ecosystem components were only considered in a few studies. The dominating type of assessment outputs (13 studies) have been maps with 'semi-quantitative measures per unit areas' (from 250 m² to 90 km²), followed by 'quantitative measures per unit area' (from 400 m to 100 km²) in 12 studies, only a small proportion of the assessment outputs related to quantitative (2 studies) or semi-quantitative (5 studies) measures for given management units (thus one value for a case study area). More than half of the reviewed studies carried out a risk evaluation and tested a broad range of scenarios including simulated pressure-effect scenarios, mostly related to the future license areas of wind farms or fisheries management measures. Cumulative effect scenarios have been tested by weighting for instance the relationship between indicators and pressures. It is relevant to allude to the fact that about one third of the studies did not account for uncertainty. Some studies assessed uncertainty quantitatively based on model uncertainty. Other studies addressed uncertainty in a qualitative way, mainly by a discussion about the issue of uncertainty and/or proposed methods for further analysis.

Table 3: List of 32 recent empirical studies of (semi-) quantitative environmental risk assessments in the context of the development, implementation or evaluation of marine spatial management. Studies were reviewed according to the spatial scale and the methods with regard to the three steps of a risk assessment: risk identification, risk analysis and risk evaluation.

Scale and location	**Risk identification and characterisation**		**Risk analysis**			**Risk evaluation**	**References**
small <500.000km² meso >=500.000-106 km², large >=106km²	***Stressor(driver)/pressure indicators used***	***Ecosystem components/ state indicators***	***Measure of sensitivity of state indicators***	***Measure and approach used of vulnerability/risk/impact of ecosystem/ area***	***Assessment output type***	***Management scenario analysis (assessed: yes/no)***	
Small (ca. 270.000km²); Great Barrier Reef MPAs, Australia	Pollution	Mulitple habitats (coral reefs and seagrass beds)	Model output	Frequency of plume occurrence with spatially distributed loads; final maps of exposure (E) = annual frequency of plume occurrence grid (F)*sum of spatially distributed TSS and DIN loads grid [for all rivers (P)]	Quantitative measures per unit area; mapping out approach of frequencies	No	(Alvarez-Romero et al., 2013)
Meso (ca. 500.000km²); Canada's EEZ, Pacific coast	Cumulative pressure (additive) from human stressors	Multiple habitats	Expert knowledge	Cumulative impact = ∑(intensity*habitat*vulnerability (vulnerability score for activity i and habitat j, by expert judgement), MPA restrictions included	Quantitative measure per unit area (400m grid); cumulative impact score matrices	Yes, three scenarios were used:1) include each fishery separately, 2) summarize fisheries by type of impact, 3) include only one layer for commercial and one for recreational fisheries	(Ban et al., 2010)
Small; 3 Italian MPA, Mediterranean Sea	Multiple human and environmental stressors	Multiple habitats	Empirical data, expert knowledge	Environmental diagnostic = ∑ scores of individual habitat per cell for degradation and risk [level]; weighted vulnerability [vulnerability of habitat*number of cells where the habitat is present]; environmental quality [∑naturalistic*economic*aesthetic*rarity of the habitat]; susceptibility to human use [number of habitats*importance]	Semi-quantitative measure per unit area; mapping out approach of environmental quality, susceptibility to use, weighted vulnerability	No	(Bianchi et al., 2012)
Large; Australasia	Cumulative pressure (additive, antagonistic, synergistic) from global (climate change) and local (nutrient input) stressors	Habitat (seagrass)	Empirical data	Additive effects model (effect size*stressor values) to test for interactions between pressures (no, antagonistic and synergistic interactions)	Quantitative measure per unit area (100km²); interactive impact maps (local and global stressors)	Yes, the management effect of each pressure has been assessed	(Brown et al., 2013)
Small (ca. 70km²); Ebro Delta, NW Mediterranean Sea	Offshore windfarms	Multiple species (sea birds)	Empirical data, model output	Potential risk = spatial overlap between aggregative patterns of seabirds [coupling Taylor's power law (TPL) with linear mixed effect models] and offshore wind farm placement	Semi-quantitative measure per unit area (12.5km²); mapping out approach risk	Yes, future offshore wind farm areas have been considered	(Christel et al., 2013)

Scale and location	Risk identification and characterisation		Risk analysis			Risk evaluation	References
Small; South Florida coastal ecosystem, Gulf of Mexico	Multiple global (e.g. climate change) and local (e.g. fishing) stressors	Multiple species, multiple habitats	Expert knowledge	Impact = matrix-based analyses of pressures to states and services, scored by expert opinion	Quantitative measures for given management unit; relative impact matrices	No	(Cook et al., 2013)
Small (28.500km^2); German EEZ, North Sea	Fisheries	Habitat (benthic)	Model output	Risk = proportion of the ecosystem component*∑(proportion of the cell*gain function per cell (∑recovery potential over mortality potential for all impacts))	Quantitative measure per unit area; distribution of cumulative risk by area and benthic distribution	Yes, four scenarios evaluated against goals from European maritime policies (MSFD, CFP, HD)	(Fock et al., 2011)
Small (28.500km^2); German EEZ, North Sea	Fisheries, aggregate extraction	Multiple species (benthic, mammals, sea birds)	Model output	Loss and exposure = mortality (M) / recovery (R)	Quantitative measure per unit area (3*3nm/6*6nm); risk scores by area and ecosystem function	No	(Fock, 2011a)
Small (256.500km^2 and 40km^2); UK (English and Welsh) waters	Cumulative pressures (additive, antagonistic, synergistic) from fisheries and aggregate extraction	Multiple habitats (benthic)	Expert knowledge, model output	Cumulative impact = degree of disturbance from type of fishing gear, fishing intensity, habitat sensitivity and recovery rates	Quantitative measure per unit area (1km^2); cumulative impact scenario output	Yes, four cumulative effects scenarios (greatest, additive, antagonistic and synergistic) to estimate overall recovery times	(Foden et al., 2010)
Small (256.500km^2); UK (English and Welsh) waters	Cumulative pressures (greatest, additive, antagonistic, synergistic) from human stressors	Multiple habitats (benthic)	Expert knowledge, model output	Cumulative impact = degree of disturbance from type of pressure, pressure intensity, habitat sensitivity and recovery rates	Quantitative measure per unit area (1km^2); cumulative impact scenario output	Yes, four cumulative effects scenarios (greatest, additive, antagonistic and synergistic) to estimate overall recovery times	(Foden et al., 2011)
Small (ca. 55.500km^2); Northern-Central Adriatic, Mediterranean Sea	Fisheries	Multiple species (functional groups)	Model output	Biomass and catch changes = amount of total biomass, commercial species biomass, predator species biomass, fish biomass, invertebrates (except plankton) biomass, total catch, demersal catch, pelagic catch) assessed using spatial–temporal food web model Ecospace	Quantitative measure per unit area (25km^2); scenario output tables	Yes, scenarios regarding spatial management (MPAs), three temporal simulations of temporary closures and overall reduction of fishing effort (Ecospace)	(Fouzai et al., 2012)
Small, Scottish waters	Cumulative pressures (additive) from human stressors	Multiple species (sea birds)	Expert knowledge, model output	Disturbance risk = (ship and helicopter traffic, habitat specialisation)*conservation importance	Semi-quantitative measure for given management unit; ranked species concern scores	No	(Furness and Tasker, 2000)

Scale and location	Risk identification and characterisation		Risk analysis			Risk evaluation	References
Small (28.500km²); German EEZ, North Sea	Cumulative pressure (additive) from human stressors	Single species (fish)	Expert knowledge, model output	Risk = pressure to state vulnerability [severity and duration of (negative) effects (due to human pressure) + the sensitivity of species (resiliency, reversibility, sensitivity etc.)]	Semi-quantitative measure per unit area (5km²); mapping out approach and scenario output	Yes, multiple risk scenarios based on the identification of potential conflict areas between drivers and between pressures and nursery grounds	(Gimpel et al., 2013)
Small; coastal zone of the Great Australian Bight, South Australia	Fisheries	Multiple species (mammals)	Expert knowledge, model output	Risk of extinction = population viability analysis based on time and probability of terminal extinction and quasi extinction by subpopulation, region and marine fishing areas with the greatest bycatch risk	Semi-quantitative measure per unit area (10*10 km nodes); risk scenario output, bycatch rates	Yes, three scenarios of increasing, stable and decreasing population trajectories	(Goldsworthy and Page, 2007)
Small (ca. 26.000km²); Great Barrier Reef, Australia	Cumulative pressure (additive) from human stressors	Habitat (seagrass)	Expert knowledge	Cumulative impact = vulnerability [frequency, functional impact, resistance, recovery time (years) and certainty]	Semi-quantitative measure per unit area (2km²); cumulative impact score mapping	No	(Grech et al., 2011)
Small; Barcelona Habour, Spain	Pollution	Habitat quality (water)	Model output	Risk index = probability, exposure and vulnerability; branch-decision scheme to evaluate the cost of each decision as a function of vulnerability, proximity and toxicity of potential contaminants	Semi-quantitative measures per unit area; spatial distribution of risk	Yes, decision branch model based on cost/utility	(Grifoll et al., 2010)
Small, (125.000km^2); North Sea	Fisheries	Multiple species (benthic)	Model output	Relative ecological impacts of disturbance = degree to which production and biomass in habitats respond to trawling disturbance; sensitivity = recovery time	Semi-quantitative measures per unit area (9 km²); impact maps	Yes, five management scenarios based on modelled reduction in biomass and production	(Hiddink et al., 2007)
Small (ca. 80.000km²); Baltic Sea	Cumulative pressure (additive) from human stressors	Multiple habitats (benthic)	Expert knowledge	Cumulative impact = weighting of pressures to habitat specific impacts [statistical approach, thresholds based on mean ± sd of cumulative impact within habitat type] using HELCOM weighting factors	Semi-quantitative measure per unit area (71289m²); cumulative impact scores	No	(Korpinen et al., 2013)
Small (1km²); Spanish coast, local beaches, Mediterranean Sea	Multiple human and environmental stressors	Habitat quality, multiple species, ecosystem function and services	Empirical data, expert knowledge	Risk = ∑hazard intensity*ecosystem service values [habitat, disturbance regulation, water supply, recreational and aesthetic services, spiritual and historic values]	Semi-quantitative measure for given management unit; risk valuation and prioritization	No	(Lozoya et al., 2011)
Small (20.000km²); Brazilian coast (continental shelf area), Atlantic	Marine traffic, hydrocarbon exploration	Single species (mammals)	Empirical data, expert knowledge	Risk = humpback whale density category + anthropogenic impact category	Semi-quantitative measure per unit area (ca. 50km radius); risk mapping and conservation prioritization	No	(Martins et al., 2013)

Scale and location	Risk identification and characterisation		Risk analysis			Risk evaluation	References
Small (30km²); Archipelago of La Maddalena (Sardinia, Italy), Mediterranean Sea	Pollution	Habitat quality (beaches)	Empirical data, expert knowledge	Risk = hazard index [normalised oil particle concentration derived from models]*vulnerability [geomorphology and environmental protection]	Semi-quantitative measure per unit area (90km²); mapping out of hazard index	No	(Olita et al., 2012)
Small (4km²); Ligurian Sea MPA (Italy), Mediterranean Sea	Multiple human stressors	Multiple habitats	Expert knowledge, model output	Marine territory score (impact) = relationship between pressure intensities and ecosystem status (spatially resolved [distance of habitats from reference/unperturbed conditions (4 habitat indices)] and average over territory)	Semi-quantitative per unit area (250m²): mapping of change in marine territory status (impact)	Yes, management scenarios based on experts judgment of changes in pressure intensities used in the model	(Parravicini et al., 2012)
Small (10.000km²); Bay of BiscaySpanish EEZ at the Basque Coast, Atlantic	Fisheries	Multiple species (trophic levels)	Empirical data, expert knowledge	Total fishing pressure (TFP) = cumulative fishing intensity; fishing pressure per commercially relevant species; fishing pressure by trophic level	Semi-quantitative measure per unit area (1km²); TFP maps	No	(Pascual et al., 2013)
Small (1km²); San Foca tourist harbour (Italy), Mediterranean Sea	Pollution	Habitat quality	Expert knowledge, model output	Risk = likelihood of negative environmental changes resulting from human activities (subjective and objective expert opinions)	Semi-quantitative measure for management unit; mapping of spatially explicit risk values	No	(Irene et al., 2010)
Small (ca. 25.000km²); South California, USA	Marine traffic	Multiple species (mammals)	Model output	Ship-strike risk = shipping routes [route-use overlay] in combination with whale distribution model [generalised additive model (GAM)]	Quantitative measures per unit area (4km²); risk scores for different shipping scenarios	Yes, spatial scenarios for (alternative) ship traffic and military use, fishing and conservation (MPAs)	(Redfern et al., 2013)
Small (ca 10.000km²); Pudget Sound, USA	Multiple human stressors	Multiple species (fish)	Empirical data	Risk = direct impacts of pressures [mortality] and resilience [fecundity, behavioural/physiological response, life-history traits]; spatial overlaps between pressure and states of various ecosystem components	Semi-quantitative measure for given management units; risk maps and risk scores	No	(Samhouri and Levin, 2012)
Meso (500.000km²); UK southern, eastern and western coastal waters	Aggregate extraction	Multiple species	Empirical data, expert knowledge	Risk = vulnerability [spatial overlap and statistical test]; sensitivity index [recovery potential (e.g. ability to switch diet and reproductive strategy)]	Quantitative measure per unit area (2*2nm); overlay map as vulnerability	Yes, current and future license areas have been considered	(Stelzenmüller et al., 2010)

Scale and location	Risk identification and characterisation		Risk analysis			Risk evaluation	References
Small (28.500km^2); German EEZ, North Sea	Fisheries	Single species (fish)	Empirical data	Risk = ratio of species abundance [environmental parameters (temperature, salinity and depth)] and catch in commercial fisheries using BN model	Quantitative measures per unit area (3*3 degrees); BN model output, vulnerability states	Yes, the impact of no-takes zones due to establishment of wind parks have been considered (changes in fishing effort distribution and temperature)	(Stelzenmüller et al., 2011)
Small (150.000km^2); Gulf of Finland	Nutrient loads	Habitat quality (water body)	Model output	Risk = phosphorus loads (t/year), nitrogen loads (t/year)	Quantitative measure for given management unit; mapping out approach of predicted concentrations	Yes, coupled model output using multiple scenarios	(Vanhatalo et al., 2013)
Large; Western and Central Pacific Ocean	Fisheries	Multiple species (sea birds)	Empirical data, expert knowledge	Risk = productivity (P) / susceptibility (S) [P = Fecundity Factors index; S = product of fishing effort and normalised species distributions weighted with vulnerability of species to longline fishing gear; vulnerability = number of kills reported]; PSA Analysis	Semi-quantitative measure per unit area (5*5 degrees); mapping out approach, summing up over all species, season and flag	No	(Waugh et al., 2012)
Large; Australian waters	Fisheries	Multiple habitats	Empirical data, expert knowledge	Impact = PSA (Productivity Susceptibility Analysis [productivity = level of natural disturbance, regeneration of fauna; susceptibility = availability, encounterability, selectivity])	Semi-quantitative measure for given management unit (30 or 60nm); risk category per habitat	No	(Williams et al., 2011)
Small (3800km^2); Rhode Island	Offshore wind farms	Multiple species	Empirical data, expert knowledge	Impact = concern index [sensitivity to displacement, weighting of species by species] to predict areas with high conservation priority in relation to their distribution (surface area)	Quantitative measure per unit area (2km^2); scenario output, mapping of vulnerability	Yes, Zonation software (Moilanen, 2013)	(Winiarski et al., 2014)

4.4.2 Case study

Fleet specific trawling frequencies show clear spatial patterns, and as an example we illustrated the spatial distribution of the mean trawling frequency of the international beam trawl fleet with 80 mm mesh size and > 221 KW overlaid with the current (2013) offshore wind development (OWD) application areas in Figure 4. The mean overall local mortality rate assuming an equal impact of all fishing fleets is displayed in Figure 5 (top), where high values can be found in the North-East of the study area and along a coastal strip.

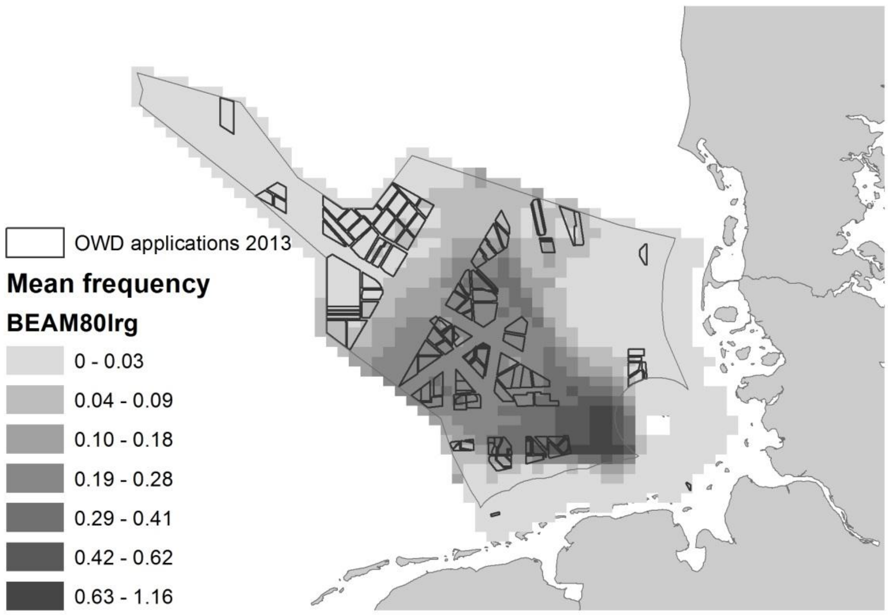


Figure 4: Mean (2005 – 2008) frequency of a unit area (3 x 3 nm) being reworked by the international beam trawl fleet with a mesh size of 80 mm and > 221KW derived from VMS data (Beam80lrg) and additionally overlaid with the current (2013) offshore wind development (OWD) application areas.

The relative combined recovery rates of the benthic communities are fishery independent and therefore patterns resembled the benthic communities (Figure 5; bottom). Spatial predictions of DI revealed that 5.3 % of the total area showed values >1, indicating a higher rate of mortality than recover, whereas 0.74 was the maximum value estimated for the weighted disturbance indicator (DI_w). High values of the unweighted and weighted disturbance indicator were found in different places (Figure 6). This is due to the fact that in the case of DI_w the beam trawl fleets using nets with >800mm mesh size (Beam80lrg and Beam80sml) were given by far the highest impact weights.

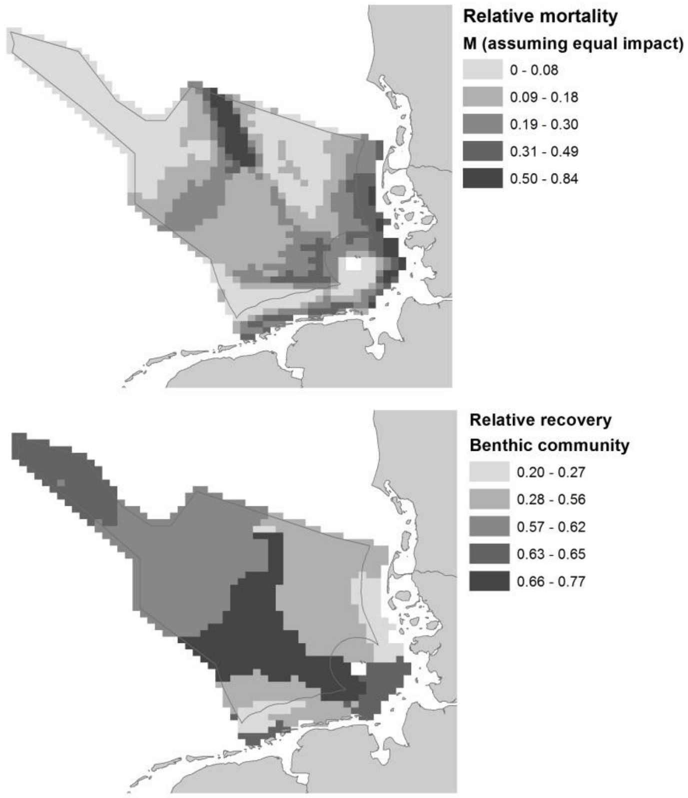


Figure 5: top: Relative overall local mortality rate (M) ($is_k = 1$) based on the distribution of the mean fishing frequency by the respective fleets; bottom: distribution of the estimated relative recovery rates derived from the combined recovery time and recover frequency of the prevailing benthic communities (see Table 1).

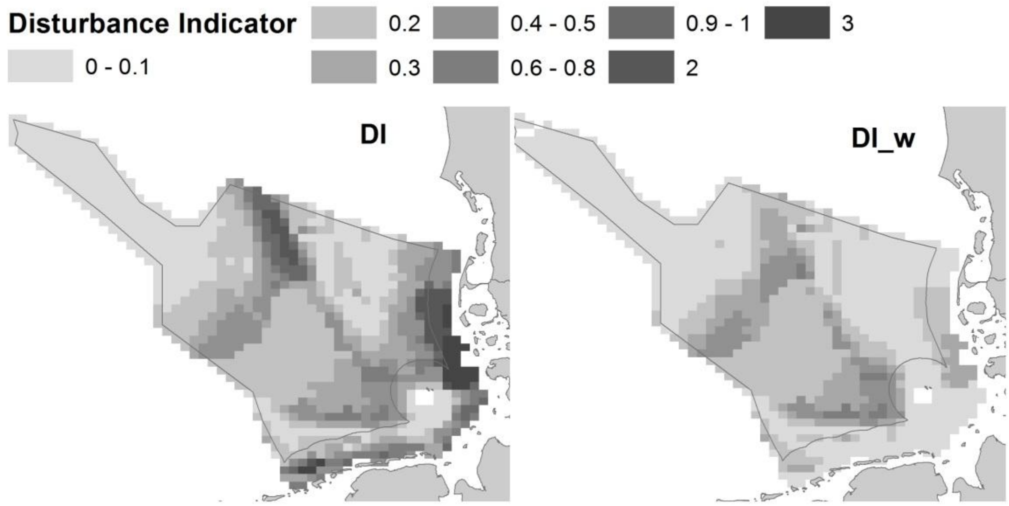


Figure 6: left: Estimated values of the disturbance indicator (DI) based on an overall local mortality rate with equal weight for the impact scores of the six fishing fleets; right: Estimated values of the disturbance indicator (DI_w) based on an overall local mortality rate with different weights for the impact scores of the six fishing fleets ($is_{\mathrm{BEAM80lrg}} = 1$; $is_{\mathrm{BEAM80sml}} = 1$; $is_{\mathrm{BEAM1631lrg}} = 0.1$; $is_{\mathrm{BEAM1631sml}} = 0.1$; $is_{\mathrm{OTTER80lrg}} = 0.15$; $is_{\mathrm{OTTER80sml}} = 0.15$).

For each BN node that represents a continuous variable the weighed mean (the mean value weighted by the probability of occurrence) with its Gaussian standard deviation is shown on the bottom of each node (Figure 3). For instance the weighted mean state value for large beam trawl frequencies is 0.221 +/- 0.34 indicating a high level of variance. The trained BN displays the "business as usual scenario" using the fishing effort patterns from 2005-2008, from which it was derived that 34.5 % of the total area showed the highest level of trawling frequencies (state 3: 0.06 and 1.16, Figure 3). An alternative interpretation of the probabilities associated to the respective node states is that there is a 34.5% chance to find a value between 0.06 and 1.16 within any given unit area (vector grid cell). The baseline BN showed further that there is a 4.12% chance to find values of DI >1 within any given unit area. In contrast, there is only a 1.35% chance to find values of DI_w >1 within any given unit area. The sensitivity analysis of the disturbance indicator node (DI) showed that the latter was most influenced by the findings for mortality (node M; variance reduction = 22.5%), recovery (node R; variance reduction = 13.8%), combined recover frequency (variance reduction = 10.9%), and type of benthic community (variance reduction = 10.3%), while all other nodes resulted in a variance reduction < 2%. The classification success rate (spherical payoff) which ranges from 0 to 1, with 1 being the best model performance, indicated a relative accuracy of the BN model for predicting the disturbance indicator (DI) with a value of 0.87 and a value of 0.95 for predicting DI_w, respectively.

The effects of the planned offshore wind development sites on the two measures of benthic disturbance (DI and DI_w) were explored stepwise (Figure 7a and b). Figure 7a showed that the new prior distribution of the Beam80lrg node (corresponding to the spatial relocation of 15% of the fishing activities) resulted in an average likely value of 0.31 for DI along with a standard error of 0.42. Compared to the "business as usual" scenario the predicted probabilities of the DI states only altered around 1 %. In contrast, using the same scenario the average likely value of DI_w increased from 0.235 (+/-0.27) to 0.261 (+/- 0.29). However, this increase was not significant due to the great variance in estimates. The additional modification of the prior distribution of the Beam1631sml node and the predicted probabilities of benthic disturbance states are displayed in Figure 7b. The model predicted an average likely value of 0.309 for DI (+/- 0.42), while the average likely value for DI_W remained the same. However, for this case study, where the BN is populated with spatial data, the likely values of the disturbance indicator averaged over the entire study area of minor importance (as indicated by the high standard error). Here, the predicted likelihood of an area proportion having a certain value is much more relevant to evaluate trade-offs of spatial management scenarios. Whereas the assumed redistribution scenario of both fleets showed no significant effect on the four DI states, overall changes were predicted in relation to the probability distributions of $DI_{w.}$ states.

(a)

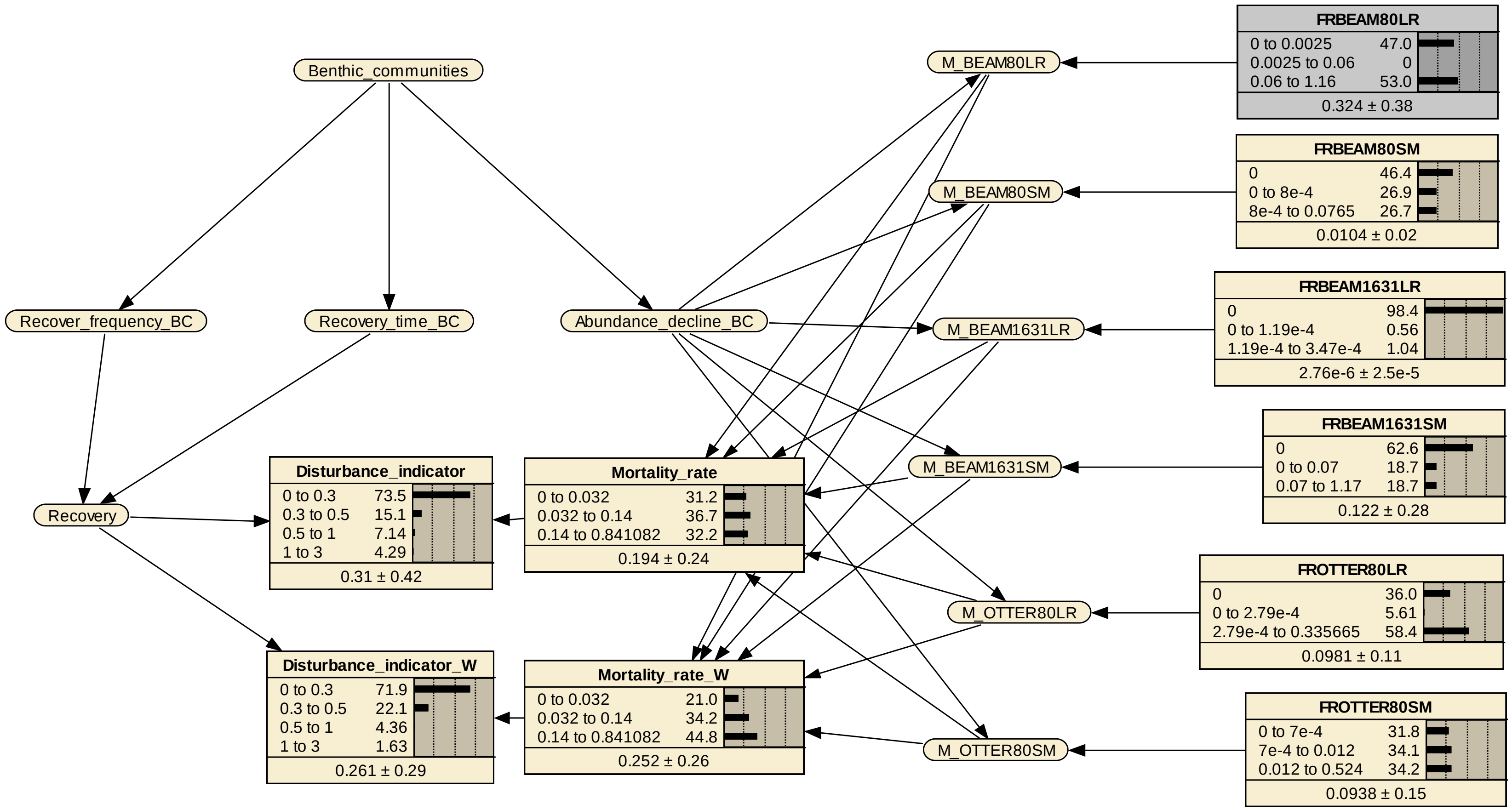
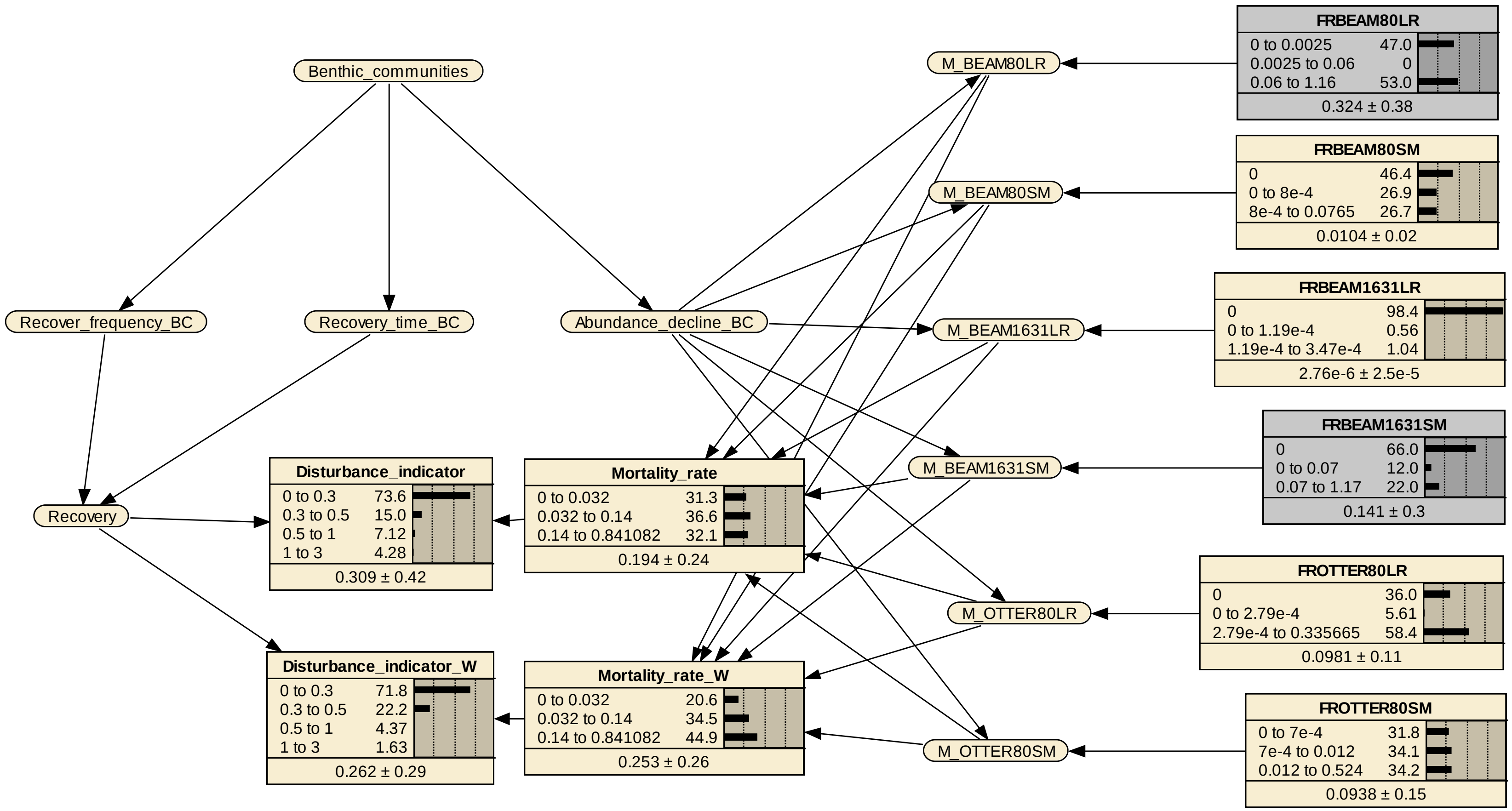


Figure 7a,b: Results of the inference the Bayesian belief network model applying the spatial management scenario “What are the likely impacts of spatial shifts of 15 % of the total fishing frequency of large beam trawlers (Beam80lrg) and 3% of the small beam trawlers (Beam1631sml) on local disturbance rates (assuming equal and weighted impacts of the different fishing fleets)”. Predicted probabilities (%) are shown for all states of the relevant model nodes.

(b)



Benthic_communities
Recover_frequency_BC
Recovery_time_BC
Abundance_decline_BC
Recovery
Disturbance_indicator
0 to 0.3 73.6
0.3 to 0.5 15.0
0.5 to 1 7.12
1 to 3 4.28
0.309 ± 0.42
Disturbance_indicator_W
0 to 0.3 71.8
0.3 to 0.5 22.2
0.5 to 1 4.37
1 to 3 1.63
0.262 ± 0.29
Mortality_rate
0 to 0.032 31.3
0.032 to 0.14 36.6
0.14 to 0.841082 32.1
0.194 ± 0.24
Mortality_rate_W
0 to 0.032 20.6
0.032 to 0.14 34.5
0.14 to 0.841082 44.9
0.253 ± 0.26
M_BEAM80LR
M_BEAM80SM
M_BEAM1631LR
M_BEAM1631SM
M_OTTER80LR
M_OTTER80SM
FRBEAM80LR
0 to 0.0025 47.0
0.0025 to 0.06 0
0.06 to 1.16 53.0
0.324 ± 0.38
FRBEAM80SM
0 46.4
0 to 8e-4 26.9
8e-4 to 0.0765 26.7
0.0104 ± 0.02
FRBEAM1631LR
0 98.4
0 to 1.19e-4 0.56
1.19e-4 to 3.47e-4 1.04
2.76e-6 ± 2.5e-5
FRBEAM1631SM
0 66.0
0 to 0.07 12.0
0.07 to 1.17 22.0
0.141 ± 0.3
FROTTER80LR
0 36.0
0 to 2.79e-4 5.61
2.79e-4 to 0.335665 58.4
0.0981 ± 0.11
FROTTER80SM
0 to 7e-4 31.8
7e-4 to 0.012 34.1
0.012 to 0.524 34.2
0.0938 ± 0.15

The estimated probabilities of DI_w values > 1 ranged between 1.35% (business as usual scenario) and 1.63 % (full displacement scenario). This means that 1.63 % of the study area (or 1.63% of all vector grid cells) will experience DI_w values > 1 using the current fishing effort displacement scenario. More relevant changes to the predicted probabilities were observed for the DI_w states 1 and 2. Compared to the baseline scenario the predicted probabilities of the DI_w state 1 decreased around 8 % (from 79.3 % to 71.9 %), while the probabilities of DI_w state 2 increased about 6 % (from 16 % to 22.1 %). This means that 8 % of the area (8 % of the vector grid cells) will likely face a worsening of DI_w values compared to the current state. This is consequently related with an increased probability (by 6%) for any given unit area to have a DI_w value ranging from 0.3 to 0.5. Thus the here considered MSP measures and the related fishing effort displacement scenario would not fulfil the defined overall operational management objective ("*The average relative vulnerability of benthic communities to fishing should not deteriorate with respect to current levels")*, since the predicted probability distributions of the DI_w values showed deteriorating values compared to the current state.

4.5 Discussion

4.5.1 Current ERA approaches and gaps in a spatial management context

We used the steps of a risk assessment framework described by Cormier and al. (2013) to frame the assessment of a fair number of spatially explicit and quantitative ERAs concerned with spatial management questions. There are, of course, other established risk assessment frameworks such as a Productivity–Susceptibility Analysis (PSA) a semi-quantitative ERA methodology (Waugh et al., 2012) or the conceptual DPSIR (Driver-Pressure-State-Impact-Response) framework which illustrates cause-effect pathways (Elliott, 2002). Further bow tie diagrams describe and analyse risk events by visualising relevant pathways from causes to consequences (Ferdous et al., 2013). The bow tie diagram focuses on so-called barriers representing existing control or mitigation measures that are placed between the causes and the risk, and the risk and consequences. These diagrams can also be adapted to the DIPSR framework. Recently, BNs have been used in combination with bow tie diagrams to overcome their purely depictive capabilities by adding probabilities and conditional dependencies between components (Badreddine and Amor, 2013; Khakzad et al., 2013).

The here identified methodological shortcomings were based on a structured, but not exhaustive selection of studies. Nevertheless, this selection was a result of a literature database search (Scopus) using defined key-words, context and expected type of output. Review results showed that independently from the investigated ecosystem components, computing quantitative measures of sensitivity is still challenging and could hardly be derived from empirical data alone. Often a combination of model outputs and expert knowledge seemed to deliver the preferred metric (e.g. Foden et al., 2011). Thus our findings emphasised the lack of empirical studies to support extrapolation of measures of sensitivity to system scale questions (see discussion in Crain et al., 2008). Another identified weakness was the lack of an explicit assessment of uncertainty, especially in cases where expert judgements were used. Uncertainty cannot be eliminated from any integrated assessment or model-based decision support, however it should be recognised and constructively handled (Astles et al., 2006; Rotmans and van Asselt, 2001). Thus the assessment of uncertainty is an important prerequisite of the herein described steps of risk analysis and subsequent risk evaluation. For instance fuzzy sets and advice theory allow for characterisation of uncertainty associated with expert knowledge (Ferdous et al., 2013). Also Walker-type and pedigree matrices were utilised to assess both the sources and respective relative levels of uncertainty related to an assessment process which integrates numerous sources of information and data qualities (Stelzenmüller et al., 2015).

Despite the great variation of terminology across studies the minimum measure of vulnerability involved in all cases was a combination of a measure of sensitivity of an ecosystem component and the probability and magnitude of a stressor occurring. However, only a few studies computed vulnerability according to the best practices defined in De Lange et al. (2010), which require the consideration of resistance and resilience when defining sensitivity and vulnerability, respectively. This depicts a future need to root spatially explicit quantitative ERAs more in ecological theory with regard to system function and processes (e.g. Fock et al., 2011).

Scenario evaluation is deemed as an important step in the risk assessment framework and which has been carried out in roughly half of the reviewed studies. Those who did simulate management scenarios generally used spatially explicit tools and approaches such as Ecospace (Fouzai et al., 2012), Zonation (Moilanen, 2013; Winiarski et al., 2014) or a combination of GIS and BN models (Stelzenmüller et al., 2011) to allow for a non-static assessment of cause-effect pathways.

Surprisingly, only one of the studies, included in this review, exploited a process-based numerical model to predict ecosystem responses to natural or human pressures (Vanhatalo et al., 2013). Process-based models represent physical processes and typically include forcing by waves and/or currents, a response in terms of sediment transport and a morphology-updating module. Routinely used for reconstructions of past conditions or to forecast possible future trends, such models are useful in the context of risk assessments (Weisse et al., 2009), in particularly, when the simulations cover a wide range of natural variability. Building on hydrodynamic drift simulations, Chrastansky and Callies (2011) have demonstrated how such model data can be turned into spatially explicit information on the risk posed by hypothetical oil spills in the North Sea. Their approach based on a BN, which makes the essential information of the model available without the need to access the memory-intensive, original data sets. In that way, detailed information on key natural drivers and their causal relationships with existing pressures can easily be considered in a wider GIS-coupled risk assessment framework. Until now, this is rarely the case in ERAs making it difficult (if not impossible) to *separate the effects of natural disturbance, for example* by waves, from that caused by human activities such as bottom trawling (Diesing et al., 2013). According to ecological theory (Pickett and White, 1985), disturbance regime is, however, an important spatial process which should be accounted for when assessing the risks of spatial management scenarios.

4.5.2 Perspectives for assessing the trade-offs of MSP measures in the German EEZ of the North Sea

The aim of the case study was to address some of the methodological shortcomings identified in the current literature on spatially explicit and quantitative ERAs and to provide some perspectives for assessing the trade-offs of on MSP measures in the German EEZ of the North Sea.

We built on a study by Fock (2011a) for calculating measures of fishing frequency, mortality rates and the disturbance indicators. The overall measures of recovery and mortality have been computed for ten benthic communities (Pesch et al., 2008). For this we converted existing model outputs on recovery and mortality rates by sediment type to respective rates by benthic community. This has been done by weighting sediment specific parameters with likely species habitat preferences given in Rachor and Nehmer (2003).

As a consequence, those benthic community specific estimates on mortality and recovery rates reflect rather rough estimates of those parameters. A promising alternative source for recovery rates (days) by phyla and habitat type provides a meta-analysis of trawl impact studies carried out by Kaiser et al. (2006). In future studies, those results could be used to redefine for instance fleet specific impact scores (is_{fleet}) of the weighted mortality rates. Further, benthic disturbance was only calculated for infaunal benthic communities, while epifaunal species may be more vulnerable to fishing disturbance (Piet et al., 2000). Empirical data for instance revealed longer recovery times of benthic epifaunal communities (7 - 8 years) compared to infauna communities (2 - 5 years) in the German Bight (at least after the impact of cold winters) (Neumann and Kröncke, 2011). As a result, future steps to improve mortality and recovery rates of benthic communities would embrace the combination of infaunal and epifaunal recovery and decline rates.

In our case study we did not explicitly map or consider a measure of natural disturbance, however we can assume that natural disturbance, e.g. by tidal and wave stress as well as daily and seasonal temperature variability, is highest in shallow coastal areas (Becker et al., 1992; Neumann et al., 2013). Here, benthic communities will show greater resilience to fishing disturbance than in zones with larger water depths (e.g. Hiddink et al., 2006b). Further Elliott and Quintino (2007) argued that communities in stressed environments are well adapted to natural stress and will probably never show a recovery to “undisturbed” communities. Thus taking interactions between fishing and natural disturbances into account would very likely result in different patterns of the disturbance indicator. Nevertheless, Fock et al. (2011) suggested that observed recovery rates incorporate indirectly local effects of natural disturbance. Addressing a similar topic Diesing et al. (2013) investigated the impact of demersal fishing on sea-floor integrity in the greater North Sea and proposed a method to incorporate natural and fishing disturbance in a spatially explicit study. They defined trawling impact as significant when it exceeds natural disturbance (by waves and tides). The resulting indicator was expressed as a probability on a 12x12nm grid and could as such be rescaled and incorporated into our risk assessment approach.

The observed differences in spatial pattern of the two disturbance indicators were clearly a result of the weighting of the impact of the different fishing fleets. Hence DI and DI_w describe a range of likely outcomes of disturbance modelling with DI_w as lower and DI as upper bound. In this sense it reflects a transparent assessment of uncertainty.

To enable a dynamic link of risk analysis and risk evaluation, hence scenario evaluation, we combined GIS with a BN model to conduct a quantitative spatially explicit risk assessment. For the integration of BNs and GIS we followed in general the good practice described in Johnson et al. (2012). BNs indeed are advantageous, especially when considering the input from various data types (Aguilera et al., 2011), but model construction often is challenging and nontrivial (Kjræulff and Madsen, 2012). BNs represent multi-dimensional distributions and can conveniently be applied for updating probability distributions of all variables given observations for just a subset of them. Information available will propagate across the whole network regardless of the orientation of edges (see e.g. Kjræulff and Madsen, 2012). This analysis of joint probabilities based on incomplete observations must be distinguished, however, from predicting the results of external interventions (e.g. scenario assessment). For the latter purpose a BN must be formulated in line with causal relationships (see Pearl, 2000). According to Kjræulff and Madsen, (2012) a BN is a probabilistic network for reasoning under uncertainty, whereas an influence diagram is a probabilistic network for reasoning about decision making under uncertainty. Thus an influence diagram represents parameters actively controlled by rational decision-makers as non-random decision nodes. They rate system configurations that result from management decisions based on value or utility nodes (Pearl, 1988; Bedford and Cooke, 2001). In our example we did not construct an influence diagram with decision nodes. Further multistage decision networks allow even for considering a sequence of decisions at future points in time when certain types of information will become available. Such repeated decision making is an essential part of an adaptive management process (Vugteveen et al., 2014). A representation of such practically relevant concepts in a probabilistic framework such as the one illustrated here, however, is scientifically challenging and requires future development.

Our spatial management scenario simulated a general spatial shift of fishing effort from medium fished areas to low and highly fished areas due to the development of offshore renewables in areas where 15% and 3 % of the total average beam trawl effort took place. This was based on the assumption that vessels conducting demersal mixed or crustacean fishery reallocate their effort in areas of potential large catch or previous knowledge and experience (Bastardie et al., 2013a). Results showed that the assumed shift in fishing frequencies did not result in significant changes of the average likely value of the disturbance indicator. However, disturbance indicators (assuming unequal impact) still worsen in approximately 8 % of the study area. This information is much more meaningful when evaluating the trade-offs of spatial management options. Once, more realistic fishing effort displacement scenarios become available, the combined GIS and BN approach can be used to predict likely local values of e.g. the disturbance indicator. For instance individual based models, predicting fishing fleet behaviour under changing economic or ecological conditions (Bastardie et al., 2013b), would allow entering specific findings for prior distributions of fishing frequencies of specific fleets.

4.6 Conclusion

Currently, quantitative ERA studies in a spatial management context reflect a wide range of assessment approaches, with varying interpretations of the terms risk, vulnerability or impact. Especially the different definitions of vulnerability suggest that future spatially explicit quantitative ERAs should be more rooted in ecological theory with regard to system function and processes. Spatially explicit risk assessments yet to come should also consider the inclusion of numerical models for instance describing natural disturbance, since this is an important component in ecological disturbance theory. We identified a transparent assessment of uncertainty as clear shortcoming of many current approaches and conclude that the application of Bayesian belief networks are a promising approach to address this. Also future research is needed on how to build meaningful influence diagrams, with parameters actively controlled by rational decision-makers (decision nodes), in the course of quantitative ERAs. Independently from the concepts and methods applied to predict a measure of risk, we strongly recommend putting caution on the type of output produced and its potential uptake in an actual spatial management process. The latter often refers to complex multiple objectives settings, where the impacts of numerous human activities need to be jointly assessed. In conclusion, marine spatial management or MSP processes should embed ERA frameworks which allow for the integration of multiple risk assessments and the quantification of related uncertainties at a common spatial scale.

4.7 Acknowledgements

Contributions of RD and HN were funded by the German Federal Ministry of Education and Research under project NOAH (03F0669A, North Sea – Observation and Assessment of Habitats). HR was funded by a grant of the "Deutsche Bundesstiftung Umwelt". The effort data used in the BN model were collated during the plaice evaluation project (PBOX funded by DG Mare), and for this data for the Danish and Dutch fleets were kindly provided by Josefine Egekvist (DTU Aqua), and Niels Hintzen (IMARES). Further many thanks to Torsten Schulze for processing the original VMS and logbook data.

4.8 References

92/43/EEC, C. D. 1992. Council Directive 92/43/EEC of 21 May 1992 on the conservation of natural habitats and of wild fauna and flora.

Aguilera, P. A., Fernández, A., Fernández, R., Rumí, R., and Salmerón, A. 2011. Bayesian networks in environmental modelling. Environmental Modelling and Software, 26: 1376-1388.

Alvarez-Romero, J. G., Devlin, M., Teixeira da Silva, E., Petus, C., Ban, N. C., Pressey, R. L., Kool, J., et al. 2013. A novel approach to model exposure of coastal-marine ecosystems to riverine flood plumes based on remote sensing techniques. Journal of Environmental Management, 119: 194-207.

Astles, K. L., Holloway, M. G., Steffe, A., Green, M., Ganassin, C., and Gibbs, P. J. 2006. An ecological method for qualitative risk assessment and its use in the management of fisheries in New South Wales, Australia. Fisheries Research, 82: 290-303.

Badreddine, A., and Amor, N. B. 2013. A Bayesian approach to construct bow tie diagrams for risk evaluation. Process Safety and Environmental Protection, 91: 159-171.

Ban, N. C., Alidina, H. M., and Ardron, J. A. 2010. Cumulative impact mapping: Advances, relevance and limitations to marine management and conservation, using Canada's Pacific waters as a case study. Marine Policy, 34: 876-886.

Bastardie, F., Nielsen, J. R., Andersen, B. S., and Eigaard, O. R. 2013a. Integrating individual trip planning in energy efficiency - Building decision tree models for Danish fisheries. Fisheries Research, 143: 119-130.

Bastardie, F., Nielsen, J. R., and Miethe, T. 2013b. DISPLACE: a dynamic, individualbased model for spatial fishing planning and effort displacement — integrating underlying fish population models. Canadian Journal of Fisheries and Aquatic Sciences: 1-21.

Becker, G. A., Dick, S., and Dippner, J. W. 1992. Hydrography of the German Bight. Marine Ecology Progress Series, 91: 9-18.

Bedford, T., and Cooke, R. 2001. Probabilistic risk analysis: foundations and methods, Cambridge University Press.

Bianchi, C. N., Parravicini, V., Montefalcone, M., Rovere, A., and Morri, C. 2012. The challenge of managing marine biodiversity: A practical toolkit for a cartographic, territorial approach. Diversity, 4: 419-452.

BMVBS 2009. Spatial Plan for the German Exclusive Economic Zone in the North Sea. http://www.bsh.de/en/Marine_uses/Spatial_Planning_in_the_German_EEZ/index.jsp.

Brennan, J., Fitzsimmons, C., Gray, T., and Raggatt, L. 2014. EU marine strategy framework directive (MSFD) and marine spatial planning (MSP): Which is the more dominant and practicable contributor to maritime policy in the UK? Marine Policy, 43: 359-366.

Brown, C. J., Saunders, M. I., Possingham, H. P., and Richardson, A. J. 2013. Interactions between global and local stressors of ecosystems determine management effectiveness in cumulative impact mapping. Diversity and Distributions.

Carneiro, G. 2013. Evaluation of marine spatial planning. Marine Policy, 37: 214-229.

Chrastansky, A., and Callies, U. 2011. Using a Bayesian Network to Summarize Variability in Numerical Long-Term Simulations of a Meteorological-Marine System: Drift Climatology of Assumed Oil Spills in the North Sea. Environmental Modeling and Assessment, 16: 1-14.

Christel, I., Certain, G., Cama, A., Vieites, D. R., and Ferrer, X. 2013. Seabird aggregative patterns: A new tool for offshore wind energy risk assessment. Marine Pollution Bulletin, 66: 84-91.

Collie, J. S., Vic Adamowicz, W. L., Beck, M. W., Craig, B., Essington, T. E., Fluharty, D., Rice, J., et al. 2013. Marine spatial planning in practice. Estuarine, Coastal and Shelf Science, 117: 1-11.

Commission, E. 2008. Directive 2008/56/EC of the European Parliament and of the Council of 17 June 2008, establishing a framework for community action in the field of marine environmental policy (Marine Strategy Framework Directive). Official Journal of the European Union, L164: 19-40.

Commission, E. 2014. Directive 2014/89/EU of the European Parliament and of the Council of 23 July 2014 establishing a framework for maritime spatial planning. *In* /89/EU, pp. 135-145. Ed. by E. P. a. t. C. o. t. E. Union.

Cook, G. S., Fletcher, P. J., and Kelble, C. R. 2013. Towards marine ecosystem based management in South Florida: Investigating the connections among ecosystem pressures, states, and services in a complex coastal system. Ecological Indicators.

Cormier, R., and al., e. 2013. Marine and coastal ecosystem-based risk management handbook. 317. 60 pp.

Crain, C. M., Kroeker, K., and Halpern, B. S. 2008. Interactive and cumulative effects of multiple human stressors in marine systems. Ecology Letters, 11: 1304-1315.

De Lange, H. J., Sala, S., Vighi, M., and Faber, J. H. 2010. Ecological vulnerability in risk assessment - A review and perspectives. Science of the Total Environment, 408: 3871-3879.

Diesing, M., Stephens, D., and Aldridge, J. 2013. A proposed method for assessing the extent of the seabed significantly affected by demersal fishing in the Greater North Sea. ICES Journal of Marine Science, 70: 1085-1096.

Douvere, F., and Ehler, C. N. 2010. The importance of monitoring and evaluation in adaptive maritime spatial planning Journal of Coast Conservation: Online First.

Eastwood, P. D., Mills, C. M., Aldridge, J. N., Houghton, C. A., and Rogers, S. I. 2007. Human activities in UK offshore waters: an assessment of direct, physical pressure on the seabed. ICES Journal of Marine Science, 64: 453-463.

Elliott, M. 2002. The role of the DPSIR approach and conceptual models in marine environmental management: An example for offshore wind power. Marine Pollution Bulletin, 44: iii-vii.

Elliott, M., and Quintino, V. 2007. The Estuarine Quality Paradox, Environmental Homeostasis and the difficulty of detecting anthropogenic stress in naturally stressed areas. Marine Pollution Bulletin, 54: 640-645.

Ferdous, R., Khan, F., Sadiq, R., Amyotte, P., and Veitch, B. 2013. Analyzing system safety and risks under uncertainty using a bow-tie diagram: An innovative approach. Process Safety and Environmental Protection, 91: 1-18.

Fock, H. 2011a. Integrating multiple pressures at different spatial and temporal scales: A concept for relative ecological risk assessment in the european marine environment. Human and Ecological Risk Assessment, 17: 187-211.

Fock, H. O. 2011b. Natura 2000 and the European Common Fisheries Policy. Marine Policy, 35: 181-188.

Fock, H. O., Kloppmann, M., and Stelzenmüller, V. 2011. Linking marine fisheries to environmental objectives: A case study on seafloor integrity under European maritime policies. Environmental Science and Policy, 14: 289-300.

Foden, J., Rogers, S. I., and Jones, A. P. 2010. Recovery of UK seabed habitats from benthic fishing and aggregate extraction-Towards a cumulative impact assessment. Marine Ecology Progress Series, 411: 259-270.

Foden, J., Rogers, S. I., and Jones, A. P. 2011. Human pressures on UK seabed habitats: A cumulative impact assessment. Marine Ecology Progress Series, 428: 33-47.

Foley, M. M., Armsby, M. H., Prahler, E. E., Caldwell, M. R., Erickson, A. L., Kittinger, J. N., Crowder, L. B., et al. 2013. Improving ocean management through the use of ecological principles and integrated ecosystem assessments. BioScience, 63: 619-631.

Fouzai, N., Coll, M., Palomera, I., Santojanni, A., Arneri, E., and Christensen, V. 2012. Fishing management scenarios to rebuild exploited resources and ecosystems of the Northern-Central Adriatic (Mediterranean Sea). Journal of Marine Systems, 102-104: 39-51.

Furness, R. W., and Tasker, M. L. 2000. Seabird-fishery interactions: Quantifying the sensitivity of seabirds to reductions in sandeel abundance, and identification of key areas for sensitive seabirds in the North Sea. Marine Ecology Progress Series, 202: 253-264.

Gimpel, A., Stelzenmüller, V., Cormier, R., Floeter, J., and Temming, A. 2013. A spatially explicit risk approach to support marine spatial planning in the German EEZ. Marine Environmental Research, 86: 56-69.

Goldsworthy, S. D., and Page, B. 2007. A risk-assessment approach to evaluating the significance of seal bycatch in two Australian fisheries. Biological Conservation, 139: 269-285.

Grech, A., Coles, R., and Marsh, H. 2011. A broad-scale assessment of the risk to coastal seagrasses from cumulative threats. Marine Policy, 35: 560-567.

Grêt-Regamey, A., Brunner, S. H., Altwegg, J., and Bebi, P. 2013a. Facing uncertainty in ecosystem services-based resource management. Journal of Environmental Management, 127: S145-S154.

Grêt-Regamey, A., Brunner, S. H., Altwegg, J., Christen, M., and Bebi, P. 2013b. Integrating expert knowledge into mapping ecosystem services tradeoffs for sustainable forest management. Ecology and Society, 18.

Grifoll, M., Jordà, G., Borja, Á., and Espino, M. 2010. A new risk assessment method for water quality degradation in harbour domains, using hydrodynamic models. Marine Pollution Bulletin, 60: 69-78.

Halpern, B. S., McLeod, K. L., Rosenberg, A. A., and Crowder, L. B. 2008a. Managing for cumulative impacts in ecosystem-based management through ocean zoning. Ocean and Coastal Management, 51: 203-211.

Halpern, B. S., Walbridge, S., Selkoe, K. A., Kappel, C. V., Micheli, F., D'Agrosa, C., Bruno, J. F., et al. 2008b. A global map of human impact on marine ecosystems. Science, 319: 948-952.

Hiddink, J. G., Jennings, S., and Kaiser, M. J. 2006a. Indicators of the ecological impact of bottom-trawl disturbance on seabed communities. Ecosystems, 9: 1190-1199.

Hiddink, J. G., Jennings, S., and Kaiser, M. J. 2007. Assessing and predicting the relative ecological impacts of disturbance on habitats with different sensitivities. Journal of Applied Ecology, 44: 405-413.

Hiddink, J. G., Jennings, S., Kaiser, M. J., Queirós, A. M., Duplisea, D. E., and Piet, G. J. 2006b. Cumulative impacts of seabed trawl disturbance on benthic biomass, production, and species richness in different habitats. Canadian Journal of Fisheries and Aquatic Sciences, 63: 721-736.

Hope, B. K. 2006. An examination of ecological risk assessment and management practices. Environment International, 32: 983-995.

Irene, P., Paolo, V., Donatella, V., Alberto, M. J., Mauro, F., and Giovanni, Z. 2010. Mapping the environmental risk of a tourist harbor in order to foster environmental security: Objective vs. subjective assessments. Marine Pollution Bulletin, 60: 1051-1058.

Johnson, S., Low-Choy, S., and Mengersen, K. 2012. Integrating bayesian networks and geographic information systems: Good practice examples. Integrated Environmental Assessment and Management, 8: 473-479.

Kaiser, M. J., Clarke, K. R., Hinz, H., Austen, M. C. V., Somerfield, P. J., and Karakassis, I. 2006. Global analysis of response and recovery of benthic biota to fishing. Marine Ecology Progress Series, 311: 1-14.

Katsanevakis, S., Stelzenmüller, V., South, A., Sørensen, T. K., Jones, P. J. S., Kerr, S., Badalamenti, F., et al. 2011. Ecosystem-based marine spatial management: Review of concepts, policies, tools, and critical issues. Ocean and Coastal Management, 54: 807-820.

Khakzad, N., Khan, F., and Amyotte, P. 2013. Dynamic safety analysis of process systems by mapping bow-tie into Bayesian network. Process Safety and Environmental Protection, 91: 46-53.

Kjræulff, U. B., and Madsen, A. L. 2012. Bayesian Networks and Influence Diagrams: A Guide to Construction and Analysis: A Guide to Construction and Analysis, Springer.

Korpinen, S., Meidinger, M., and Laamanen, M. 2013. Cumulative impacts on seabed habitats: An indicator for assessments of good environmental status. Marine Pollution Bulletin, 74: 311-319.

Lozoya, J. P., Sardá, R., and Jiménez, J. A. 2011. A methodological framework for multi-hazard risk assessment in beaches. Environmental Science & Policy, 14: 685-696.

MacDonald, L. 2000. Evaluating and managing cumulative effects: process and constraints. Environmental Management, 26: 299-315.

Marcot, B. G., Steventon, J. D., Sutherland, G. D., and McCann, R. K. 2006. Guidelines for developing and updating Bayesian belief networks applied to ecological modeling and conservation. Canadian Journal of Forest Research-Revue Canadienne De Recherche Forestiere, 36: 3063-3074.

Martins, C. C. A., Andriolo, A., Engel, M. H., Kinas, P. G., and Saito, C. H. 2013. Identifying priority areas for humpback whale conservation at Eastern Brazilian Coast. Ocean and Coastal Management, 75: 63-71.

McCann, R. K., Marcot, B. G., and Ellis, R. 2006. Bayesian belief networks: Applications in ecology and natural resource management. Canadian Journal of Forest Research, 36: 3053-3062.

Moilanen, A. 2013. Planning impact avoidance and biodiversity offsetting using software for spatial conservation prioritisation. Wildlife Research, 40: 153-162.

Neumann, H., and Kröncke, I. 2011. The effect of temperature variability on ecological functioning of epifauna in the German Bight. Marine Ecology, 32: 49-57.

Neumann, H., Reiss, H., Ehrich, S., Sell, A., Panten, K., Kloppmann, M., Wilhelms, I., et al. 2013. Benthos and demersal fish habitats in the German Exclusive Economic Zone (EEZ) of the North Sea. Helgoland Marine Research, 67: 445-459.

Olita, A., Cucco, A., Simeone, S., Ribotti, A., Fazioli, L., Sorgente, B., and Sorgente, R. 2012. Oil spill hazard and risk assessment for the shorelines of a Mediterranean coastal archipelago. Ocean & Coastal Management, 57: 44-52.

Parravicini, V., Rovere, A., Vassallo, P., Micheli, F., Montefalcone, M., Morri, C., Paoli, C., et al. 2012. Understanding relationships between conflicting human uses and coastal ecosystems status: A geospatial modeling approach. Ecological Indicators, 19: 253-263.

Pascual, M., Borja, A., Galparsoro, I., Ruiz, J., Mugerza, E., Quincoces, I., Murillas, A., et al. 2013. Total fishing pressure produced by artisanal fisheries, from a Marine Spatial Planning perspective: A case study from the Basque Country (Bay of Biscay). Fisheries Research, 147: 240-252.

Pearl, J. 1988. Probabilistic reasoning in intelligent systems: networks of plausible inference, Morgan Kaufmann.

Pearl, J. 2000. Causality: models, reasoning and inference, Cambridge Univ Press.

Pesch, R., Pehlke, H., Jerosch, K., Schröder, W., and Schlüter, M. 2008. Using decision trees to predict benthic communities within and near the German Exclusive Economic Zone (EEZ) of the North Sea. Environmental monitoring and assessment, 136: 313-325.

Pickett, S. T. A., and White, P. S. 1985. Patch dynamics: a synthesis. *In* The ecology of natural disturbance and patch dynamics, pp. 371-384. Ed. by S. T. A. P. a. P. S. White. Academic Press, New York.

Piet, G. J., Rijnsdorp, A. D., Bergman, M. J. N., Van Santbrink, J. W., Craeymeersch, J., and Buijs, J. 2000. A quantitative evaluation of the impact of beam trawling on benthic fauna in the southern North Sea. ICES Journal of Marine Science, 57: 1332-1339.

Qiu, W., and Jones, P. J. S. 2013. The emerging policy landscape for marine spatial planning in Europe. Marine Policy, 39: 182-190.

Rachor, E., and Nehmer, P. 2003. Erfassung und Bewertung ökologisch wertvoller Lebensräume in der Nordsee. Alfred Wegener Institute for Polar and Marine Research, Bremerhaven.

Redfern, J. V., McKenna, M. F., Moore, T. J., Calambokidis, J., Deangelis, M. L., Becker, E. A., Barlow, J., et al. 2013. Assessing the Risk of Ships Striking Large Whales in Marine Spatial Planning. Conservation Biology, 27: 292-302.

Rotmans, J., and van Asselt, M. B. 2001. Uncertainty management in integrated assessment modeling: towards a pluralistic approach. Environmental monitoring and assessment, 69: 101-130.

Samhouri, J. F., and Levin, P. S. 2012. Linking land- and sea-based activities to risk in coastal ecosystems. Biological Conservation, 145: 118-129.

Spiegelhalter, D. J., and Dawid, P. 1993. Bayesian analysis in expert systems. Statistical Science, 8: 219-283.

Stelzenmüller, V., Breen, P., Stamford, T., Thomsen, F., Badalamenti, F., Borja, T., Buhl-Mortensen, L., et al. 2013. Monitoring and evaluation of spatially managed areas: A generic framework for implementation of ecosystem based marine management and its application. Marine Policy, 37: 149-164.

Stelzenmüller, V., Ellis, J. R., and Rogers, S. I. 2010. Towards a spatially explicit risk assessment for marine management: Assessing the vulnerability of fish to aggregate extraction. Biological Conservation, 143: 230-238.

Stelzenmüller, V., Schulze, T., Fock, H. O., and Berkenhagen, J. 2011. Integrated modelling tools to support risk-based decision-making in marine spatial management. Marine Ecology Progress Series, 441: 197-212.

Stelzenmüller, V., Vega Fernández, T., Cronin, K., Röckmann, C., Pantazi, M., Vanaverbeke, J., Stamford, T., et al. 2015. Assessing uncertainty associated with the monitoring and evaluation of spatially managed areas. Marine Policy, 51: 151-162.

Tyler-Walters, H., Hiscock, K., Lear, D., and Jackson, A. 2001. Identifying species and ecosystem sensitivities.

Vanhatalo, J. P., Tuomi, L. M., Inkala, A. T., Helle, S. I., and Pitkänen, J. H. 2013. Probabilistic ecosystem model for predicting the nutrient concentrations in the gulf of finland under diverse management actions. Environmental Science and Technology, 47: 334-341.

Vugteveen, P., van Katwijk, M. M., Rouwette, E., and Hanssen, L. 2014. How to structure and prioritize information needs in support of monitoring design for Integrated Coastal Management. Journal of Sea Research, 86: 23-33.

Waugh, S. M., Filippi, D. P., Kirby, D. S., Abraham, E., and Walker, N. 2012. Ecological Risk Assessment for seabird interactions in Western and Central Pacific longline fisheries. Marine Policy, 36: 933-946.

Weisse, R., von Storch, H., Callies, U., Chrastansky, A., Feser, F., Grabemann, I., Günther, H., et al. 2009. Regional meteorological - Marine reanalyses and climate change projections. Bulletin of the American Meteorological Society, 90: 849-860.

Williams, A., Dowdney, J., Smith, A. D. M., Hobday, A. J., and Fuller, M. 2011. Evaluating impacts of fishing on benthic habitats: A risk assessment framework applied to Australian fisheries. Fisheries Research, 112: 154-167.

Winiarski, K. J., Miller, D. L., Paton, P. W. C., and McWilliams, S. R. 2014. A spatial conservation prioritization approach for protecting marine birds given proposed offshore wind energy development. Biological Conservation, 169: 79-88.

5. Manuscript 4: Ecosystem service trade-off assessment to support marine spatial planning

Antje Gimpel[a], Vanessa Stelzenmüller[a], Jens Floeter [b], Axel Temming [b],

[a] Johann Heinrich von Thünen Institute (TI), Federal Research Institute for Rural Areas, Forestry and Fisheries, Institute of Sea Fisheries, Palmaille 9, 22767 Hamburg, Germany

[b] Institute for Hydrobiology and Fisheries Science, University of Hamburg, Olbersweg 24, 22767 Hamburg, Germany

To be submitted to Ecosystem Services.

5.1 Abstract

Facing the revision process of the German Maritime Spatial Plan for the Exclusive Economic Zone (EEZ) of the North Sea, green energy, sustainable food production and accruing competition for maritime space are currently a matter of debate. Aiming to reduce impacts across sectors and avoid conflicting uses of coastal resources, the concept of ecosystem services (ES) attracts attention to support efficient marine spatial planning (MSP). It facilitates the comparison of linked costs (risks) and services (economic returns), namely ES trade-offs, and thus the detection of efficient management objectives. Within this study, the state of the art of spatially explicit ES modelling is investigated. Further, a spatially explicit trade-off analysis of ES is applied using the German Bight of the North Sea as a case study area. Accordingly, spatial management scenarios are developed focusing on multiple segments such as fisheries, aquaculture or wind energy. The value of a bundle of ES was forecasted to provide key information for decision-makers seeking critical areas in the delivery of ES in a case study in the North Sea.

Surprisingly, the majority of studies examined used GIS-based models instead of off-the-shelf tools to support ES evaluation. As a matter of fact, simple GIS-based mapping proved to be useful during the case study. It facilitated the identification of productive and essential areas (e.g. habitats) and the analysis of trade-offs between habitat (conservation) features and/or (socio) economic ones. Assessing the effects of management measures quantitatively in terms of ES helps to provide a common language across all disciplines (e.g. stakeholders, planners, decision makers). Thus, ES trade-off analyses inform an ecosystem-based approach to MSP in future.

Keywords: Ecosystem services, Geographic information system (GIS), German Bight, Marine spatial planning (MSP), Trade-off analysis

5.2 Introduction

In the EU, 88 % of fish stocks are overexploited or significantly depleted (EC, 2014c). The Good Environmental Status (GES), promoted by the EU Marine Strategy Framework Directive (MSFD), requires ecologically diverse and dynamic oceans and seas which are clean, healthy and productive. Such requirements need to be achieved by the member states in 2020 (EC, 2014a). In the course of Integrated Maritime Policy (IMP) and the Europe 2020 strategy tall orders are placed with the European countries (EC, 2012). Among others, an optimization of fisheries and aquaculture contributions to food security was raised (EC, 2015). All of these policies, directives and strategies identify Marine Spatial Planning (MSP) as a cross-cutting policy tool. MSP shall contribute to the implementation of Blue growth, a long term strategy promoted by the MSP directive to support sustainable growth in the marine environment by 2020 while also benefitting GES (EC, 2014b). Place-based marine management tools such as MSP are geared to organize human activities in space and time (Stelzenmüller et al., 2014; Ehler and Douvere, 2009; Katsanevakis et al., 2011; Douvere, 2008). MSP integrates ecological, social, and economic interests, interactions between human activities, regardless of whether cross-border or inter-sectoral nature, whether conflict or synergy (Stelzenmüller et al., 2014; Halpern et al., 2008; Ehler and Douvere, 2009; Gimpel et al., 2013; Foley et al., 2010). Its process is characterized as dynamic and evolving, integrating permanent revisions (Ehler and Douvere, 2009).

5.2.1 Multi-objective setting as a catalyst for MSP in Germany

In the case of Germany, multiple objectives originating from a range of policies are pursued. Accordingly, marine management resulted in a bundle of spatial management measures such as designated areas for renewable energy and nature conservation (Stelzenmüller et al., 2014; Gimpel et al., 2013). Beyond, potential areas in the German Exclusive Economic Zone (EEZ) of the North Sea for the co-utilisation of Offshore Wind Farms (OWF) and Integrated Multi-Trophic Aquaculture (IMTA) are examined within the interdisciplinary project Offshore Site Selection (OSS) (Gimpel et al., 2015).

However, such management measures are drivers of change, having effects on the health of the ecosystem and therefore on human wellbeing (economic, social and personal well-being), which is based on benefits derived from the ecosystem (Burkhard et al., 2012). According to the report of the European Environment Agency (EEA), the state of European Seas is recently threatening human wellbeing due to anthropogenic impacts such as climate change or other human induced pressures. While being productive, the seas are neither healthy nor clean. Commercial fish stocks are overexploited and the EU is increasingly depending on the import of aquatic products (EEA, 2015). The effects of management strategies driving environmental alteration need to be clearly recognized and received by decision makers in a transparent form. As the German MSP is recently under revision, controversial subjects can be reconsidered. Issues such as e.g. the expansion of OWF development, which have just been revoked by the Federal Maritime and Hydrographic Agency (Bundesamt für Seeschifffahrt und Hydrographie, BSH) can be revisited. Further, the elimination of mobile bottom contact fishery gears and passive fishery gears within the selected nature protection areas Natura 2000, recently brought into action by an environmental association of NGOs, can be further examined (BUND et al., 2015).

Previous scientific studies focused on current and future management strategies regarding their potential of conflict in between human uses (Gimpel et al., 2013), potential synergies between sectors (Gimpel et al., 2015), risk of impact on essential habitats (Stelzenmüller et al., 2014; Gimpel et al., 2013), or appeal to stakeholders (Ramos et al., 2014). Nevertheless, none of those studies weighed the risks and returns of cross (or multi) sector management achievable for the German Bight. To reconcile GES with Blue Growth means to work towards the most efficient management strategies (Polasky et al., 2008; White et al., 2012). This needs to be evaluated by using a transparent approach assessing the trade-off between costs (risks) and benefits (economic return). Subsequently, management strategies can be weighed to finally balance sustainable use against ecosystem health (Halpern et al., 2012).

5.2.2 The concept of Ecosystem Services

In order to define the status of marine ecosystems and the goods and services they provide, the concept of Ecosystem Services (ES) can be applied. ES can be defined and categorized as being provisioning (e.g. food), regulative (e.g. clean water), supporting (e.g. habitats) or cultural (e.g. aesthetics) (MA, 2005). The provision of ES depends on biophysical conditions and changes over space and time due to human induced land cover, land use and climatic changes (Burkhard et al., 2012). An ES framework provides indicators to assess the potential environmental and economic costs and benefits of management strategies (Guerry et al., 2012). ES are valued in economic and other terms (e.g. biomass) at a common unit, linked to human wellbeing. Therefore, such a framework provides the ability to look at trade-offs in service provision that emerge from alternative uses of marine and coastal environments. While the monetizing of ES is widely discussed, it enables scientists to communicate results of ecological production functions on a common ground to the public. Besides, working together with stakeholders, managers and policy makers, such a common language is of great importance ensuring flawless communication across all disciplines (Ramos et al., 2014). Combining an ES framework with MSP increases the potential to ensure the sustainability of natural resources. Recently, a number of example case studies is provided (Guerry et al., 2012; Douvere, 2008; White et al., 2012). Consequently, the most efficient strategy managing the direct drivers of change towards the 2020 requirements can be identified and incorporated into decision making.

Albeit full ES trade-off analyses have rarely been used in MSP efforts to date, a range of (marine) ecosystem modelling concepts and tools is available providing decision support.

Those can in most of the times integrate political, economic and social criteria beyond physical, chemical and biological ones. For instance ARIES (Artificial Intelligence on Ecosystem Services), InVEST (Integrated Valuation of Ecosystem Services and Tradeoffs) and MIMES (Multiscale Integrated Model of Ecosystem Services) can be utilized to map and model benefits as well as service flows (Vigerstol and Aukema, 2011). ARIES, Atlantis (Atlantis Ecosystem Model) and InVEST further allow the valuing of ES and trade-off assessments (Guerry et al., 2012; Vigerstol and Aukema, 2011). A comparison of ecological outcomes and an assessment of potential trade-offs or different spatial arrangements through optimization approaches is also provided applying Marxan, Marxan with Zones, Marine Map or MIMES (Nackoney and Williams, 2013; Ban et al., 2013).

In the course of this study, recent publications focussing on ES (trade-off) assessments are examined in a literature review. They are selected accounting for their background in (marine) ecosystem modelling, i.e. priority mapping of valuable ES. The tools applied are analysed and compared by reference to standardised criteria (e.g. practical use for MSP), examining the state of the art of ES frameworks. Using the German Bight of the North Sea as a case study area, the application of an ES framework is exemplified. In order to assess the effectiveness and efficiency of future management strategies, spatially explicit ES indicators are selected and transferred in a spatial analysis. Subsequently, management scenarios linked to certain drivers such as Blue Growth and GES are defined. In order to evaluate the efficiency of those scenarios, a trade-off assessment of sector values and ES under multiple spatially explicit management scenarios is conducted. The overarching goal is to examine the concept of ES regarding its potential for decision support. Besides, it is tested if competing needs might be balanced in management for both sustainable use and ecosystem health. Here, the procedure as well as the main findings are summarised to identify the value of the ES concept for the evaluation of management scenarios towards an ecosystem-based approach to MSP.

5.3 Material and Methods

5.3.1 Research strategy

Aiming to present a first overview about spatial explicit tools applicable to assess ES, a literature review is conducted. Next, the concept of ES is transferred to evaluate the efficiency of management strategies using the German Bight of the North Sea as a case study area. Finally, the concept is examined regarding its potential for decision support, balancing sustainable use and ecosystem health in future.

5.3.2 Literature review on tools assessing Ecosystem Services

In order to assess the state of the art of spatial explicit ES models, a literature research is conducted. Using a combination of the key words “decision support tool”, “trade off”, “(marine) spatial planning”, “GIS” and “ecosystem service”, a total of 31 peer reviewed papers were selected. The approaches of evaluation encompassed current distribution assessments, explicit future trends or (environmental and socio-economic) impact analysis, where the ES provision has been linked with ES demands. In a first step, each study was categorised with regard to the aim of the study, the methods applied, the analytical process adapted to approach the aim of the study, the data needed to run the analysis and its strengths and weaknesses. Further, comparisons were made regarding the development and evaluation of spatial management scenarios. Scientific uncertainty related to input data, model parameters or model prediction was also addressed. Finally, the focus was put on the question whether a practical application had been included and how it was related to MSP or its broader context.

5.3.3 Case study

Facing its state of the art, the added value of the ES concept for decision support is tested in a case study. The logical flow below is setting the stages to perform an ES trade-off assessment in support of efficient marine management decisions.

i. Identify management objectives
ii. Develop management scenarios
iii. Identify ES indicators
iv. Assess ES trade-offs
v. Evaluate management options

First, management objectives and alternative management scenarios helpful to achieve the objectives need to be identified. Next, the level of ES produced in each scenario is estimated. The outputs are evaluated in terms of trade-offs, supporting the ultimate goal: to identify the most efficient management strategy (assessed) towards MSP.

5.3.3.1 Case study specifications

The study area comprised the German EEZ of the North Sea with a surface area of 28,539 km² (Fig. 1). The main human activities regulated by the German MSP are safety and efficiency of navigation, oil and gas exploitation, cables and pipelines, renewable energy development, and aggregate extraction as well as other uses (Buck et al., 2004; BSH, 2009). The allocation of fishing activities is not spatially managed by the MSP (Gimpel et al., 2013; Fock, 2011; Stelzenmüller et al., 2011). Currently, marine aquaculture is only taking place nearshore in terms of mussel and oyster cultures within the Wadden Sea National Park. Offshore cultivation is currently conducted in various pilot studies, but not yet conducted at a commercial scale (Buck et al., 2004; Buck and Krause, 2012; Gimpel et al., 2015). Considering the conservation perspective, marine protected areas (MPA) under the Natura 2000 protocol were implemented to protect both habitats and species by eliminating destructive mobile bottom contact gears or passive gears, representing a risk for marine mammals. The respective study area was subdivided into a set of grid cells. Accounting for the spatial resolution of available data and computation time at the scale of the study area revealed a grid size on C-square resolution (3x1.5nm or 15.43 km², respectively).

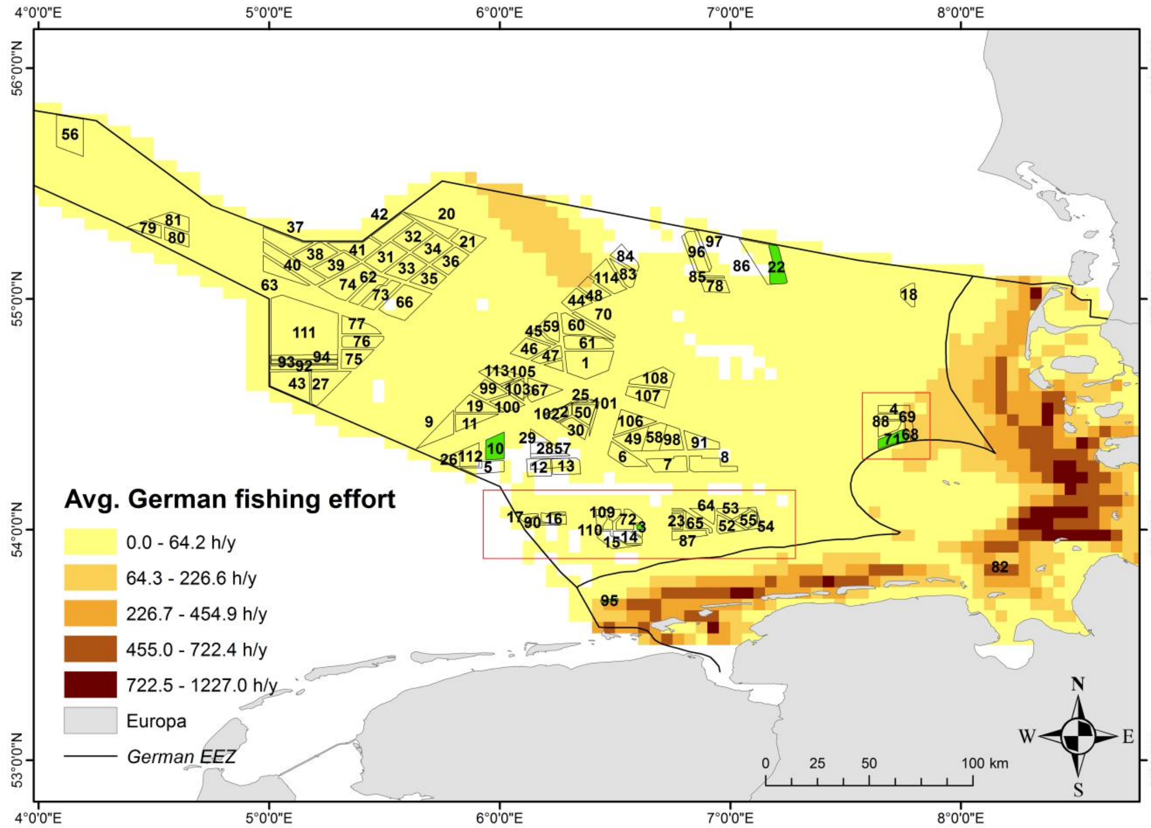


Figure 1: The German Bight including the EEZ of the North Sea, averaged German fishing activities (2008-2011) and the Offshore Wind Farms (OWF) planned until 2025 (effective from May 2015, BSH). OWFs already 'at work' are highlighted in green, suitable sites for co-locations of OWFs with aquaculture Gimpel et al. (2015) considered during this study are framed in red.

5.3.3.2 Management objectives and derived management scenarios

Working towards an ecosystem-based approach to MSP, the strategic management goal identified reads as follows: “To maintain an ecosystem in a healthy, productive and resilient condition so that it can provide the ecosystem services humans want and need” (McLeod et al., 2005). In order to achieve both, ‘Blue Growth’ and ‘GES’, the general objective is to ensure the sustainable provision of ecosystem services (EEA, 2015). As a simplification, the study is restricted to a small choice of operational objectives: “Maintain supporting services” and “Maintain provisioning services”. Hence, representative biotic and abiotic ES indicators were chosen (Tab. 1).

Table 1: Selection of management objectives driving decision making and the indicators providing information about the state of the ES and therefore the achievement of objectives.

Strategic goal	**General objective**	**Operational objective**	**Indicators**
To maintain an ecosystem in a healthy, productive and resilient condition so that it can provide the ecosystem services humans want and need	Ensure the sustainable provision of ecosystem services		
		Maintain supporting services:	*Habitats*
		Maintain provisioning services:	*Food provided by fisheries* *Food provided by aquaculture* *Renewable energy*

Further, management scenarios (Sc.) including management options linked to the management objectives (Tab. 1) were identified as follows and illustrated in detail below:

(Sc. 1) the utilisation of ES provided by the marine ecosystem of the German Bight of the North Sea on a daily business,

(Sc. 2) the realisation of the OWF development and spatial closures of Natura 2000 sites as a future perspective, causing a spatial shift of fisheries and an increase of renewable energy production, and

(Sc. 3) the realisation of co-location as a management option in MSP.

In order to assess ES under business as usual (Sc. 1), their current performances are assessed per C-square. Performance indicators are demersal trawling fishery revenues, recent OWF revenues (‘at work’, Fig. 1) and benthic habitats for essential species, redrawn from (Coull et al., 1998). The species were selected because of their dependence on the benthic health.

For the purpose of supporting the achievement of future management goals, the impact of spatial closures due to OWFs and Natura 2000 areas is assessed (Sc. 2). Performance indicators are the wind energy revenues of all OWFs, the Natura 2000 sites closed for demersal fisheries, the fishery revenues, and the benthic habitat values. Finally, the realization of co-locations of OWFs and offshore aquaculture as examined during the national project Open Ocean Multi-Use (OOMU) (Buck et al., 2012) and the national project Offshore Site Selection (OSS) (Gimpel et al., 2015) are assessed as shown in Figure 1 (Sc. 3). Consequently, performance indicators are the potential aquaculture revenues, the wind energy revenues, the fishery revenues, and the benthic habitat value per C-square.

5.3.3.3 Compilation of data

Food from fisheries

German Vessel Monitoring System (VMS) and logbook data from 2008 to 2011 were combined to calculate the average bottom trawling effort (total hours fishing per year) per C-square as described in Stelzenmüller et al. (2014) for plaice (*Pleuronectes platessa*), sole (*Solea solea),* sandeel (*Ammodytidae)* and brown shrimp (*Crangon crangon)* fisheries (Fig. 1). In order to assess benthic/demersal fishing intensity, the data are restricted to mobile bottom contact (MBC) gears (beam trawler, otter board trawler), accounting for gear width and fishing speed. To get information about the fishery revenues, an average market price for each target species (plaice, sole, sandeel and brown shrimp) is assessed for each year and multiplied with the catch per C-Square to estimate the annual fishery revenues.

Food from aquaculture

Aiming to assess potential aquaculture revenues, suitable aquaculture candidates were taken from Gimpel et al. (2015). As such, European cod (*Gadus morhua)* due to a number of reasons seems to be the most suitable: Cod grow faster than the other fish species modelled, prove to be profitable with 2.5 € per kg (Buck et al., 2012) and reveal the highest suitability year-round (Gimpel et al., 2015). As *G. morhua* showed especially high suitability along coastal areas while at the same time requiring a high degree of care (feeding, clearing of cages etc.), a cultivation approach in OWFs situated closer to the coast is preferred due to logistic constraints (Fig. 1). Here, one free-standing cage per OWF is assumed, exhibiting a cage size of 8960m³. When determining a stocking density of 25 kg per m³, a harvest of 224,000 kg per year can be assumed (Buck et al., 2012). Those results are projected onto the C-squares overlaying the most suitable OWFs to estimate the annual aquaculture harvest.

Marine renewable energy

OWFs designated by the BSH are mapped (Fig. 1) and annotated with their potential power (in kWh) by accounting for the number of turbines and the OWF area (km²). In Germany, 1 kWh is charged with approximately 14.3 cents (Hobohm et al., 2013). That information is summed up per C-square to estimate the annual wind energy harvest.

Habitats

It is assumed that there are no economic returns generated from habitats or areas closed for conservation. As a simplification of the complex ecosystem covering the entire German Bight, habitats such as sandbanks and reefs, protected by the Natura 2000 sites, are mapped. Further, spawning and nursery grounds modelled and redrawn from Coull et al. (1998) were mapped for *P. platessa*, *S. solea*, and *Ammodytidae* as shown in Fig. 2.

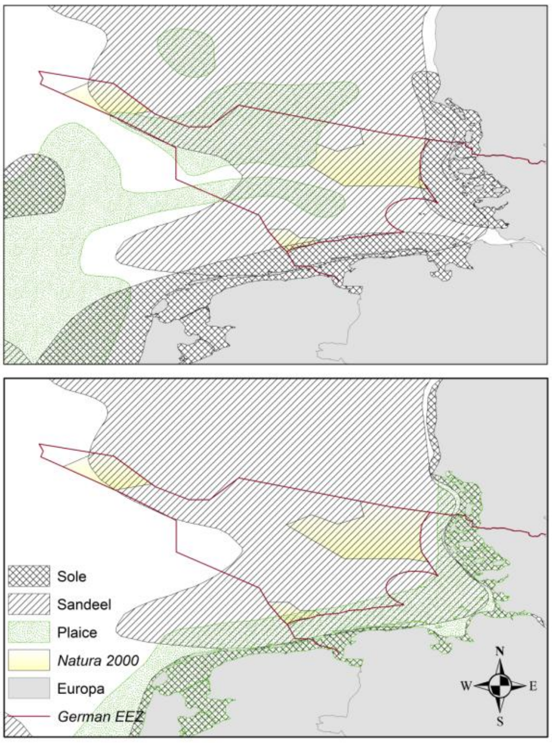


Figure 2: Spawning (top) and nursery grounds (below), taken from Coull et al. (1998).

The drivers of human activities affecting those habitats the most were identified in Gimpel et al. (2013) for plaice nursery grounds. Assuming the same pressures and sensitivity of the ecosystem to those pressures, the outcomes of Gimpel et al. (2013) were adopted for *S. solea*, and *Ammodytidae*: A high influence on the habitats was excerted by the pressures abrasion and extraction through demersal fisheries as well as obstruction and smothering through OWF development.

Thus the average spatial and temporal activity of those drivers was identified. The fishing effort was summarized for each C-Square with information on the frequency the seabed surface had been swept with. Assuming an average gear width of 18m and an average fishing speed of 3.5kn, the duration of trawling (D_t) 100 % of a C-Square was assessed to be 132.29h. The OWFs were considered based on their spatial extent (m^2).

Aiming to extrapolate from benthic habitats to the physical condition of a fish stock, a relation between habitat size, the respective Spawning Stock Biomass (SSB) and recruitment are adduced. This gained importance if an adapted behaviour of the ecosystem needed to be taken into consideration. Taking into account stock-recruitment relationships, the effects of habitat proportions lost on SSB can be expressed. Here, the stock assessments for 2015 of the ICES Advisory Committee for the Subarea IV (North Sea) were consolidated (ICES, 2014a; ICES, 2014b; ICES, 2014c). It was assumed that SSB falls under threshold B_{lim} (SSB limit reference point), if the proportion of habitat affected exceeds the proportion of total SSB to B_{lim}. Finally, all C-Squares were summed up to assess a relative measure of risk for the whole habitat.

According to the report of the ICES Advisory Committee (ICES, 2014a), the B_{lim} of plaice was assessed to be 150’ at a SSB of 675’ (weights in ‘000 tonnes). Consequently, the threshold per C-Square supporting recruitment was specified as a proportion of 22 %, whereas a C-Square being affected by > 78 % was assumed to be at risk. For sole (ICES, 2014c), a B_{lim}

of 25’ at a SSB of 50’ (weights in ‘000 tonnes) was reported. Consequently, the threshold per C-Square supporting recruitment was specified as a proportion of 50 %, whereas a C-Square being affected by > 50 % was assumed to be at risk. According to (ICES, 2014b), a B_{lim} of 75’ at a SSB of 100’ (weights in ‘000 tonnes) was reported for sandeel in the Southeastern North Sea (Sandeel Area 2). Consequently, the threshold per C-Square supporting recruitment was specified as a proportion of 75 %, whereas a C-Square being affected by > 25 % was assumed to be at risk. Consequently, a C-Square is getting unsuitable being swept with > 103.19 h/y for plaice, 66.15 h/y for sole and > 33.07 h/y for sandeel.

5.3.3.4 Trade-off assessment

The ES indicators were valued per C-Square (Tab. 2) as described in section 5.3.3.3 for each of the scenarios generated in section 5.3.3.2. Next, the indicator values (normalised to 1) wee plotted in relation to current conditions (Sc. 1), representing the risks and economic returns each management strategy brings about.

The indicators were translated into ES bundles. Accordingly, nursery or spawning grounds were converted in ‘supporting services’, wind energy into ‘provisioning services’ etc. (Tab. 2). Next, ES were aggregated to ES bundles and plotted against each other to approximate the most efficient management scenario.

Table 2: Valuation of Ecosystem Services (ES). Inputs include spatial explicit information about the indicators identified. ES outputs are expressed in biophysical or monetized units.

Services	Indicator	Definition	Value
Supporting services	Spawning grounds	Suitable reproduction habitat	km²/C-square
	Nursery grounds	Suitable habitat	km²/C-square
	Natura 2000	Suitable living space for wild animals	km²/C-square
Provisioning services	Food provided by fisheries	Conversion of solar energy into commercially harvested species	€/C-square
	Food provided by aquaculture	Conversion of solar energy into commercially harvested species	€/C-square
	Energy conversion	Providing suitable medium for energy conversion	€/Csquare

5.4 Results

5.4.1 Literature review

The literature research applying the key words specified above yielded 50 publications. 31 of the studies implemented the concept of ES spatially explicit (Tab. 3). The study aims ranged from simple distribution probability analysis (Koschke et al., 2012; Roces-Díaz et al., 2014; Sherrouse et al., 2011; De Meyer et al., 2013) over explicit trends in the provision of ES (Geneletti, 2013; Haines-Young et al., 2012) to environmental and socio-economic impact analyses (Koschke et al., 2012; Depellegrin et al., 2014; Hoyer and Chang, 2014; Jackson et al., 2013; Sanon et al., 2012; Klug and Jenewein, 2010). All of those studies aimed to link human wellbeing to ES values and vice versa, regardless of originating from an agriculture, forestry or marine background. A range of studies intended for instance a link of urban designs with ES provision (Grêt-Regamey et al., 2013; Neuenschwander et al., 2014; Lauf et al., 2014).

The tools and methods applied depended on the ES values to be assessed. An evaluation of ES from a socio-economic perspective is facilitated by Public participation GIS (Van Riper and

Kyle, 2014), SoiVes (Sherrouse et al., 2011; Sherrouse et al., 2014; Van Riper and Kyle, 2014) or Participatory mapping (Palomo et al., 2013; Klain and Chan, 2012). An evaluation of ES from an environmental perspective leads to a choice of tools such as InVest (Geneletti, 2013; Hoyer and Chang, 2014), Marxan (Ban et al., 2013) or MaxEnt (Van Riper and Kyle, 2014; Sherrouse et al., 2014; Geneletti, 2013). When the valuation of ES is determined by multiple, diverse factors, the application of MCAs (Koschke et al., 2012; Sacchelli et al., 2013) and Bayesian Belief Networks (Van der Biest et al., 2014; Grêt-Regamey et al., 2012) was observed (Fig. 3).

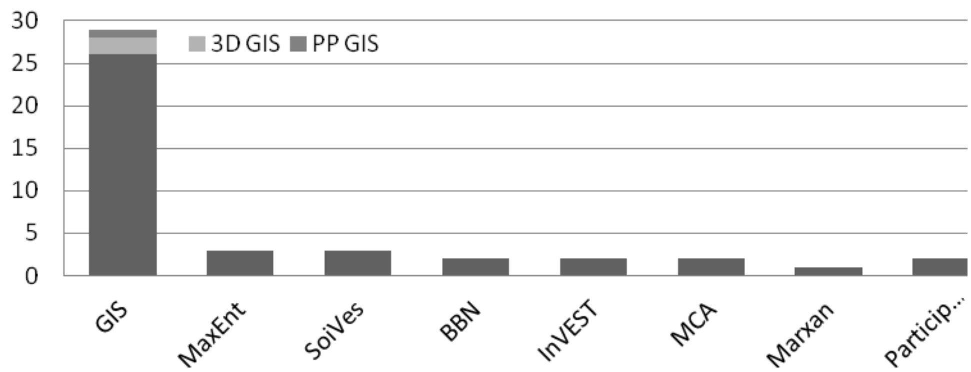


Figure 3: Tools identified to evaluate ecosystem services and/or trade-offs and number of peer-reviewed publications with corresponding applications.

The analytical approaches underlying the study aims were mostly made up from linking ES based on trade-offs or synergies (Geneletti, 2013; Hermann et al., 2014; Van der Biest et al., 2014; Castro et al., 2014; Haines-Young et al., 2012; Lauf et al., 2014; Sanon et al., 2012; Grêt-Regamey et al., 2013; Kovács et al., 2014). In general but especially in the publications cited in this context, the wordings were highly diverse: While some authors made use of ES budget concepts (Castro et al., 2014; Burkhard et al., 2012; Palomo et al., 2013), where the focus is on ES supplies and demands, Palomo et al. (2013) focussed on the areas featuring such attributes, called service provision hotspots (SPHs) and service benefiting areas (SBAs).

Furthermore, the terms labelling ecosystem services and functions were used slightly different across the studies analyzed:

While Depellegrin et al. (2014) named the ES “environmental and socio-economic assets”, Sacchelli et al. (2013) titled the ES “forest functions”. Further, the terms “landscape services” (Hermann et al., 2014; Klug and Jenewein, 2010), “value” (being of social, ecological or economic nature) (Labiosa et al., 2013; Ban et al., 2013; Van Riper and Kyle, 2014; Hilde and Paterson, 2014; Maes et al., 2012) or “benefits” of ES were applied (Koschke et al., 2012; Wainger et al., 2010; Van der Biest et al., 2014; Hilde and Paterson, 2014; Burkhard et al., 2012; Palomo et al., 2013; Klain and Chan, 2012).

ES model outputs

All of the ES identified have in common that they were mapped out using GIS. Like the aim of the studies, also their outputs appeared to be highly diverse. ES have been mapped in most of the studies (Sherrouse et al., 2011; Koschke et al., 2012; Roces-Díaz et al., 2014; Labiosa et al., 2013; Geneletti, 2013; Hermann et al., 2014; Hoyer and Chang, 2014; Castro et al., 2014; Grêt-Regamey et al., 2013; Haines-Young et al., 2012; Neuenschwander et al., 2014; Hilde and Paterson, 2014; Lauf et al., 2014; Maes et al., 2012; Burkhard et al., 2012; Palomo et al., 2013; Klain and Chan, 2012; Ban et al., 2013; Grêt-Regamey et al., 2012), whereas other studies provided maps of potential ES use (Van Riper and Kyle, 2014; Van der Biest et al., 2014; Haines-Young et al., 2012; Hilde and Paterson, 2014; Burkhard et al., 2012; Swetnam et al., 2011) or even maps of unsustainable use (Mayer et al., 2013; Lauf et al., 2014; Jackson et al., 2013; Kovács et al., 2014). Going one step further, the spatially explicit vulnerability of, the risk to and the impact on ES was assessed (Depellegrin et al., 2014; Hoyer and Chang, 2014; Sacchelli et al., 2013; Palomo et al., 2013; Klain and Chan, 2012; Jackson et al., 2013; Sanon et al., 2012; Grêt-Regamey et al., 2012; Labiosa et al., 2013).

Table 3: List of 31 recent empirical studies spatially modelling Ecosystem Services (ES).

Aim	Methods/ Model	General requirements	Analytical approach	Services modelled	Scientific uncertainty	Case study	
Assessment of human and ecologic values	Marxan	GIS, Marxan, ecological data sets (habitats)	Expert survey; mapping of known marine ES and human uses, areas of conservation value and human use value	Commercial fishing, sport fishing, ocean energy, tenures, shipping and transport, tourism, recreation	yes, model parameter (sensitivity)	British Columbia, Canada	(Ban et al., 2013)
Analysis of ES supply, demand and budgets	GIS-based modelling	GIS, CORINE land cover maps, spatial and statistical data of energy supply and demand	Empirical modelling, Mapping of ES supplies, demands and budgets	Solar energy, wind energy, energy crops, lignite	no	Leipzig-Halle, Germany	(Burkhard et al., 2012)
Spatial analysis of ES trade-offs across different landscape units	GIS-based modelling	GIS, Hierarchical classification of ES (CICES), Soil Organic Carbon (SOC) model, APLIS model, Universal Soil Loss Equation (USLE) model, Biodiversity Combined Index (BCI) model	Face-to-face, questionnaire-based surveys, Mapping of biophysical values, socio-cultural and economic values, Trade-off analysis, mismatch analysis (Chi-square test and ANOVA)	Provisioning services (cultivate crops), regulating service (climate regulation, water flow maintenance, control of erosion, maintaining habitats)	no	Iberian Peninsula, Andalusia	(Castro et al., 2014)
Linking land units to predefined ES-attribute values and vice versa	On-Site Multi-criteria Optimisation for Spatial Evaluation (OSMOSE) framework ft. BoLa (DSS to support land use planning, with focus on soil protection, generated by OSMOSE)	GIS, PostgreSQL software, MapWindow Open Source GIS software, Land use type data, Corine Land Cover datasets and soil association	Implementation of BoLa software, Performance analysis, Mapping of ES sufficiency	ES attributes soil productivity, buffering capacity, soil organic carbon stock (SOC); Susceptibility to soil compaction, soil loss due to water- and tillage erosion and soil loss due to wind erosion	no	Flanders, Belgium	(De Meyer et al., 2013))
Visual impact assessment of sea based infrastructure on the coastline and coastal hinterland	Visual impact assessment models	GIS, integrated visual impact assessment model (Vsens)	Cumulative viewshed (CV) analysis (Sea Land and Land Sea Visibility Model), environmental and socio-economic impact analysis, Distance significance definition	Cadastral value, landscape diversity, management areas, urban aggregation, recreational value, population density	no	Lithuania	(Depellegrin et al., 2014)
Linking land-use zoning policies to future ES provision	Land-Use Change (LUC) model	GIS, Land Change Modeler (IDRISI Taiga), Integrated Valuation of Ecosystem Services and Tradeoffs (InVEST), MaxEnt	Empirical modelling, Analysis of zoning policies, Illustration of explicit trends in the provision of and trade-offs between ES	Water purification, soil conservation, habitat for species, carbon sequestration, timber production	yes, model parameter (sensitivity of thresholds)	Araucanía, Chile	(Geneletti, 2013)
ES valuation across different landscape units	Bayesian Belief Network (BBN)	GIS, BN, Landscape data (forest type, elevation, etc.)	Expert knowledge, Spatially explicit uncertainty quantification related to the outcomes (probabilistic approaches, monetary risks), Traditional ES valuation	Carbon sequestration, wood production, avalanche protection	yes, model parameter and prediction (uncertainty)	Davos, Swiss Alps	(Grêt-Regamey et al., 2012)
Interactive procedural ES modeling for sustainable urban planning	GIS-based 3D visualisation (Esri CityEngine), interactive	3D GIS, Computer Graphics Architecture (CGA) rule shape grammar, Landscape elements	Literature research on urban ES, Link of parametric shape grammars for the design of generative urban patterns and the reporting of urban ES; 3d modelling, pattern valuation with interactive rulers	Regulating services (Micro-climate, water), habitat services (connectivity, habitat for flagship species), cultural services (landscape aesthetics, recreational activities etc.)	no	Abu Dhabi, Masdar City, United Arab Emirates	(Grêt-Regamey et al., 2013)

Aim	Methods/ Model	General requirements	Analytical approach	Services modelled	Scientific uncertainty	Case study	
Mapping marginal changes in capacity and ES trade-offs at different scales	Expert-and literature-driven modelling methods	Land cover data, Hierarchical classification of ES (CICES), mapping tools	Importance binary links (0/1 lookup table) based on expert knowledge and literature, (Empirical) multi criteria mapping of ES, Quantitative and qualitative map evaluation, Trade-off cluster analysis	Crop-based production, wildlife products, habitat diversity, recreation	no	EU-25 plus Switzerland and Norway	(Haines-Young et al., 2012)
Evaluation of landscape services at different spatial scales	GIS-based modelling	GIS, Corine land cover maps, tourism data	Expert knowledge, field survey, Trade-off assessment, Spatial scales: Landform approach (Corine land cover) broader habitat approach (expert driven capacity matrix)	Regulation, habitat, provision, information, carrier	no	Cross-border region of Austria and Hungary	(Hermann et al., 2014)
Assessment of future benefits of public street trees	ES valuation model (i-Tree) integrated in scenario planning software (Envision Tomorrow)	GIS, i-Tree, Envision tomorrow, Climate zone maps, annual per-tree estimated benefits by species data	Literature research, adjusting of Envision Tomorrow, scenario modelling	Energy, CO2, air quality, storm water, property values	no	City of Hutto, Central Texas	(Hilde and Paterson, 2014)
Impact assessment of climate change and land cover change on freshwater ES	Integrated Valuation of Ecosystem Services and Tradeoffs (InVEST)	GIS, InVEST, land cover and climate data	ES mapping, Sensitivity analysis, Tradeoffs between provisioning and regulating ES	Water yield, water purification (nitrogen, phosphorus), sediment retention	yes, input data and model prediction (sensitivity analysis)	Tualatin and Yamhill basins, Oregon	(Hoyer and Chang, 2014)
ES valuation on landscape scale	Polyscape - GIS mapping framework	GIS, Polyscape input data (land use data, soil map, impact of flood risk, habitat connectivity, erosion and associated sediment delivery to receptors, carbon sequestration and agricultural productivity	Spatially explicit synergy and trade-off analysis amongst ES	Flood risk, habitat connectivity, erosion and associated sediment delivery to receptors, carbon sequestration, agricultural productivity	no	Pontbren catchment, Wales	(Jackson et al., 2013)
Participatory mapping of monetarised ES	Participatory mapping techniques	GIS, semi-structured interview protocol, nautical maps	Interviews, georeferencing of nautical maps, participatory mapping of ES categories, calculation of bivariate correlations (Spearman's rank)	Economic activity (e.g. commercial fishing), tangible non-monetary benefit (Biodiversity/wildlife, recreation, scientific study site etc.), intangible non-monetary benefit (education, sense of place/home etc.), threat activity (e.g. salmon aquaculture)	yes, model prediction (uncertainty between outputs)	Vancouver Island, Canada	(Klain and Chan, 2012)
Mapping agrarian subsidy payments for evaluating changes of ES	Ground rent model framework	GIS, data on labour force, water quality, land use, subsidy payments	Quote of expected change of ES due to subsidy cash flows	Green (environmental and landscape services), blue (socio-economic services), yellow (water resources service)	no	Mondsee catchment, Austria	(Klug and Jenewein, 2010)
Real time impact assessment of land cover pattern on the provision of ES	Pimp Your Landscape (PYL)	PYL, MCA, land cover classes, benefit transfer, expert judgement	Purely expert driven approach	Supporting services (ecological integrity), cultural services (aesthetic value), provisioning services (provision of fresh water and air, human health and well-being, timber, food, and fibres,	no	Saxony, Germany	(Koschke et al., 2012)

Aim	Methods/ Model	General requirements	Analytical approach	Services modelled	Scientific uncertainty	Case study	
				regional economy), regulating services (mitigation of climate change impact)			
Linking ES trade-offs to land use conflicts in protected areas	Participatory conflict analysis	Expert judgement, Background documents, reports, notes and transcripts on qualitative ES analysis	Semi-structured interviews and focus groups, Trade-off assessments between ES perceived as important by different stakeholder groups	Provisioning (crop, fodder, habitat), regulating (flood protection), cultural services (tourism, education, research, recreation, sense of place)	no	Great Hungarian Plain, Hungary	(Kovács et al., 2014)
Characterisation of changes in important land cover related ES	Ecosystem Portfolio Model (multi-criteria scenario evaluation web tool for participatory land-use planning in urbanized areas)	GIS, Ecosystem Portfolio Model	Spatially-explicit land-use/land-cover change-sensitive modelling, ecosystem valuing related to ES and functions, land parcel prices, and community quality-of-life (QoL) metrics, Trade-off assessment	Biodiversity potential, threatened and endangered species, rare and unique habitats, landscape pattern and fragmentation index, water quality buffer potential, Ecological restoration potential	no	South Florida, USA	(Labiosa et al., 2013)
Analysis of demographic and socio-economic shifts (land-use changes) on ES flows and linkages	Land-Use Change (LUC) model with ES Assessment (ESA) model	GIS, Corine Land Cover data, LUC	Literature research, Multi-criteria ES assessment matrix for regional linkages (synergies and trade-offs etc.)	Provisioning services (energy supply, food supply), regulating services (net carbon storage, thermal emission, bioclimatic comfort), cultural service (provision of recreational green area)	yes, model prediction (sensitivity analysis)	Berlin, Germany	(Lauf et al., 2014)
Assessment of ES at EU scale	ES cascade model	Pan-European statistical model (GREEN)	Mapping of water purification services	Water purification service	no	Adour-Garonne, France	(Maes et al., 2012)
Linking urban green space typologies to ES provision	GIS-based 3D visualisation (Esri CityEngine)	3D GIS, Computer Generated Architecture (CGA), Urban green space types	Literature research, pattern designing with a form-based code, integrated into parametric modeling and visualization chain of Esri CityEngine	Microclimate regulation and air purification, water flow regulation and runoff mitigation, recreation, food and wood production, habitat, place attachment, community cohesion	no	Altstetten, Zurich, Switzerland	(Neuenschwander et al., 2014)
Assessment of benefits derived from protected areas	Participatory mapping techniques	GIS	Stakeholder survey on ES and drivers, participatory mapping of service provision hotspots (SPHs), degraded SPHs and service benefiting areas (SBAs)	Provisioning (food, water, renewable energy etc.), regulating (climate, air purification, water, Habitat), cultural (scientific knowledge, environmental education etc.)	no	Donana and Sierra Nevada Nationalparks, Andalusia	(Palomo et al., 2013)
Spatial patterns of ES at different scales	GIS-based modelling, Concept of lacunarity	GIS, SAS software	Binary maps: distribution probability, greyscale maps: quantification of ES; concept of lacunarity	Provision of food, materials and energy, Flow regulation services, Abiotic regulation services	no	Galicia, Spain	(Roces-Díaz et al., 2014)
Impact assessment of biomass removal on forest multifunctionality at different scales	GIS-based modelling, Compromise programming (CP) methodology	GIS, Hierarchical classification of ES (CICES), Corine Land Cover map, Vegetation classes, bare soil, humid classes, soil maps, fire risks etc.	Impact analysis, trade-offs between forest functions (Multifunctionality trade-off), MCA, Multi scale analysis, Compromise programming (CP) methodology	Soil and water protection, biodiversity and habitat conservation, fire risk prevention, tourist and recreational function, economic evaluation related to timber, bioenergy processing	yes, input data (sensitivity of biomass price)	Trento, Tuscany region, Italy	(Sacchelli et al., 2013)
ES trade-off	Multi Criteria Decision	Mulino decision support tool	Stakeholder and decision maker preference	Aquatic habitats, terrestrial	yes, model	Lobau	(Sanon et al.,

Aim	Methods/ Model	General requirements	Analytical approach	Services modelled	Scientific uncertainty	Case study	
assessment to support wetland restoration	Analysis (MCDA)	(mDSS4), TOPSIS, input data (fishing licenses, farmable land etc.), management options	survey, ES potential analysis, Trade-offs between ES related to management options (MCDA), stakeholder interests (mDSS4, TOPSIS)	habitats, recreation, fishery, agriculture, drinking water production	parameter (sensitivity analysis)	floodplain (urban), Danube River, Vienna, Austria	2012)
Assessment of social ES values perceived by public	Social Values for Ecosystem Services (SolVES)	GIS, SoIVES	Non-monetary Value Index from responses to a public attitude and preference survey	Aesthetic, biodiversity, future life sustaining, recreation, therapeutic	no	Pike and San Isabel National Forest, Colorado, USA	(Sherrouse et al., 2011)
Assessment of nonmarket ES values perceived by stakeholders	Social Values for Ecosystem Services (SolVES 2.0), Maximum entropy modeling software (MaxEnt)	GIS, SoIVES, MaxEnt	Stakeholder preference survey, SoIVES 2.0 compared to frequency analysis and ANOVA, discriminant function and correlation analyses	Aesthetic, Biodiversity, Cultural, Economic, Future, Historic, Intrinsic, Learning, Life Sustaining, Recreation, Spiritual, Subsistence, Therapeutic	no	3 National Forests in Colorado and Wyoming, USA	(Sherrouse et al., 2014)
Qualitative interpretation of ES using spatially explicit socio-economic scenarios	Carbon storage model	GIS, Land cover data, empirical knowledge (qualitative to quantitative)	Mapping of changed spatial distribution of carbon storage	Carbon storage	no	Eastern Arc Mountains, Tanzania	(Swetnam et al., 2011)
Linking land use planning to ES bundles	Bayesian Belief Network (BBN), Ecosystem Bundle Index (EBI)	GIS, BBN coupled with EBI, land cover data, expert judgement	Literature research, BBN and EBI analysis, Trade-off analysis betw. ES, opportunity and land use shift mapping, face validity test	Provisioning services (food production, wood production), regulating service (climate regulation)	yes, input data (uncertainty)	Grote Nete basin, Belgium	(Van der Biest et al., 2014)
Spatial analysis of terrestrial and aquatic ecosystem values perceived by public	Public participation GIS (PPGIS)	PPGIS, SoiVes, MaxEnt, New Ecological Paradigm (NEP) scales, Kmeans cluster analysis (SPSS version 21.0)	Interviews, mapping, measure of worldview using NEP scales, Kmeans cluster analysis, Mapping of value patterns, Analysis of value patterns	Aesthetic, biological diversity, cultural, economic, future, intrinsic, learning, life sustaining, spiritual, recreation, therapeutic, scientific	no	Santa Cruz Island, Channel Islands National Park, USA	(Van Riper and Kyle, 2014)
Support the targeting of (cheatgrass) restoration funds to maximize benefits of ES	Cost-effectiveness analysis (CEA) framework and optimization model	GIS, RISKOptimizer v. 1.0, Restoration funding data	Literature research on ES benefits from cheatgrass, Mapping of funding, cost-effectiveness analysis (Benefits of restoration, likelihood of successful restoration, costs of restoration), optimisation model application	Habitat/existence, property protection, grazing, hunting/recreation	yes, model prediction (uncertainty in optimisation)	Twin Falls District, Southern Idaho	(Wainger et al., 2010)

Uncertainty in the spatial explicit prediction of ES was integrated by expert opinion (Grêt-Regamey et al., 2012) and face validity maps (Van der Biest et al., 2014). Only Kovács et al. (2014), Wainger et al. (2010), Klug and Jenewein (2010) and Sanon et al. (2012) did not map the ES.

Next to ES maps, simple trend curves are shown to present ES linkages (Geneletti, 2013; Klug and Jenewein, 2010). Beyond, explicit stakeholder perspectives are given. These include preferences of ES which manifest their values as well as threats which could debase those values (Sherrouse et al., 2014; Klain and Chan, 2012; Sanon et al., 2012). Klain and Chan (2012) assessed correlations between ES categories, other authors illustrated synergies and trade-offs (Geneletti, 2013; Hermann et al., 2014; Van der Biest et al., 2014; Castro et al., 2014; Haines-Young et al., 2012; Lauf et al., 2014; Sanon et al., 2012; Grêt-Regamey et al., 2013; Kovács et al., 2014).

Strengths and weaknesses of the tools applied

Facing the wide choice of tools to represent ES, strengths and weaknesses were queried during literature research. *Public Participation GIS* and *SoIVES* were judged as interdisciplinary, distinct tools which explicitly quantify and illustrate the connections between social values, the attitudes and preferences that manifest these values, and the environmental characteristics, locations, and associated ES that elicit such values (Sherrouse et al., 2011; Sherrouse et al., 2014; Van Riper and Kyle, 2014). Weaknesses lie in their requirement of diverse input data (e.g. social surveys or environmental data).

The same is mentioned about *Participatory mapping* (Palomo et al., 2013; Klain and Chan, 2012), which can inform concrete policy proposals, but only get advantageous when being aware of all the ES specific ecosystems provide. Klain and Chan (2012) mention further limitations in gaining insight in the spatial extent of non-monetary values, which might be reasoned by difficulties in localizing them.

InVest is assessed as showing a great suite of models for the spatial representation of services and consequently decision support. Nevertheless, it requires a high amount of input data, a challenging calibration, and therefore uncertainty in final estimates of ES(Geneletti, 2013; Hoyer and Chang, 2014). Ban et al. (2013) report comparable experiences with *Marxan*. Working with Marxan requires calibration and target setting (e.g. on biodiversity) which can be a tough job. Working with environmental data requires huge effort as distinctions have to be made regarding temporal and spatial variations, data being of qualitative or quantitative nature.

Facing the same problem, a *MCA* approach was appraised as helpful to aggregate several ES at different scales, facilitating expert-based assessments of ES useful to predict the influence of several (weighted) variables on service provision at different scales of analysis and enabling trade-off assessments (by different weightings) (Koschke et al., 2012; Sacchelli et al., 2013).

In order to capture inherent ecological complexity and uncertainty in ES modelling, a *Bayesian Belief Network (BBN)* analysis is rated as being highly applicable (Van der Biest et al., 2014; Grêt-Regamey et al., 2012). It facilitates the combination of quantitative and qualitative data, empirical results and expert judgments and is judged as being transparent, adaptive and flexible in updating. Again, the implementation of spatial and temporal interactions between ES remains a challenge. Furthermore, a BN integrates both, data and modelling uncertainties, in form of conditional probability tables. This requires system knowledge, not only to understand the influence factors have on the ecosystem, but also the degree of uncertainty which is related to each variable.

Scenario evaluation

During literature research, further comparisons were made regarding the development and evaluation of spatial management scenarios. Some authors assessed the effects of land use change on ES on different time horizons (Labiosa et al., 2013; Haines-Young et al., 2012; Lauf et al., 2014; Swetnam et al., 2011; Jackson et al., 2013). All the other studies which implemented scenario assessments mention similar modifications. Those could have been induced by OWF development (Depellegrin et al., 2014), future zoning policies (Geneletti, 2013), urbanization and climate change (Hoyer and Chang, 2014), urban design (Grêt-Regamey et al., 2013; Hilde and Paterson, 2014), preferences given to ES (Sacchelli et al., 2013; Sanon et al., 2012), restoration treatment (Wainger et al., 2010), changing political strategies and incentive payments (Klug and Jenewein, 2010) or just an optimal use of ES (Van der Biest et al., 2014; Ban et al., 2013). Grêt-Regamey et al. (2012) tested a future (2050) land use scenario to forecast the values of ES under business as usual. The deviation of methods accomplished to evaluate those scenarios was more or less proportional (Fig. 4).

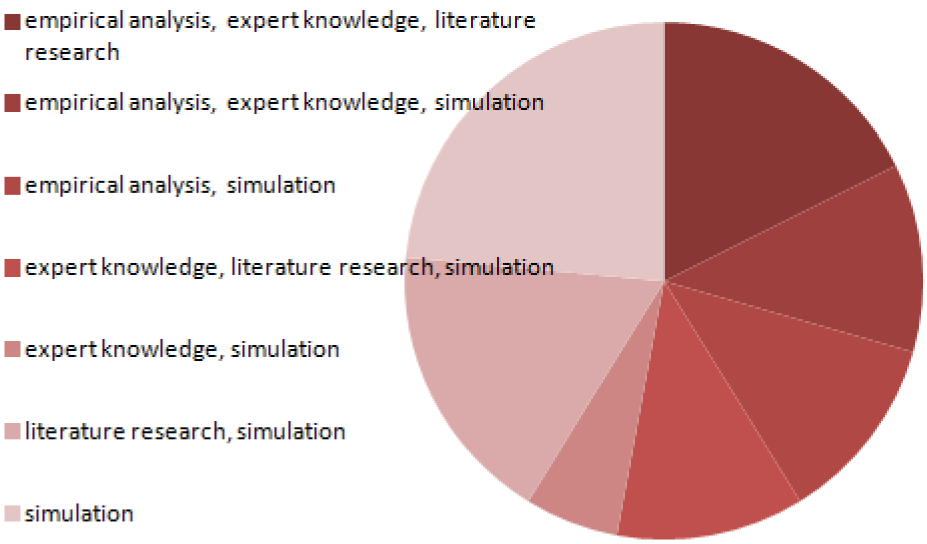


Figure 4: Proportional application of methods accomplished to evaluate future scenarios shown in Table 2.

However, the only study related to spatial planning in practice was published by Ban et al. (2013), who aimed to set the stage for MSP at Canada's Pacific coast. The study identifies areas with conservation potential to provide resource managers, scientists, decision-makers, and stakeholders with a new set of resources to inform coast-wide integrated marine planning and management initiatives.

5.4.2 Case study

The highest risk appeared to be a decrease in habitat quality for nursery and spawning grounds in Sc. 2 and 3, caused by demersal trawling and OWF development. The area of habitats decreased from 71.15 to 58.33 x 10^3 km², despite the fact, that Natura 2000 sites exhibiting an area of 7639.48 km² are closed for fisheries in Sc. 2 and 3. When splitting those results up, the spawning and nursery grounds are facing irreversible alteration. Scenario 1, representing the recent "business as usual", already leads to a degradation of 96.8 % and 50.6 % for plaice spawning and nursery grounds, 35.9 % and 49.4 % for sole spawning and nursery grounds, and 73.14 % for spawning and nursery grounds of sandeel. Similar effects are evident in scenarios 2 and 3: Plaice spawning and nursery grounds experience a loss of 56 % or 0.6 % respectively. The spawning and nursery grounds of sandeel are facing both a degradation of 29 %. In contrast, sole spawning grounds win 3.8 % due to an overlap with Natura 2000 sites, while nursery grounds are not affected at all (compare Fig. 2 and 5).

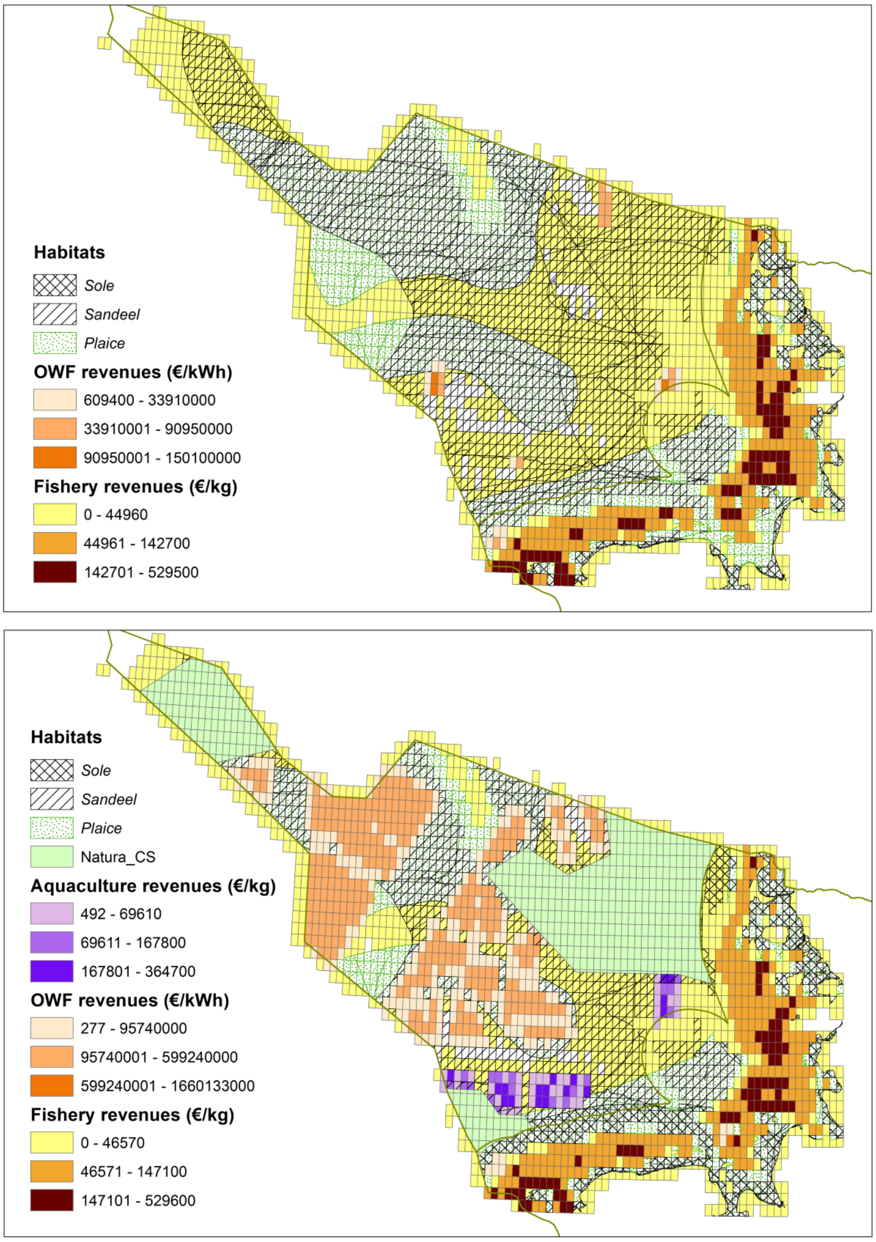


Figure 5: Scenario 1 for the German Bight "business as usual". Shown are the fish spawning and nursery grounds per species, the fishery revenues and the Offshore Wind Farm (OWF) revenues at the current state (top). Scenario 3 implements all OWFs planned, a closure of the Natura 2000 sites and co-locations of OWFs and aquaculture near the coast (below).

As a consequence of Sc. 2 and 3, an increase in renewable energy revenues from 1,433 € to 69,365 € (in '000000 €) occurs. Due to the following spatial closures, the fishery revenues decline. Showing up to € 42m in Sc. 1, the revenues decrease to round about € 40m in Sc.

2 and 3. In contrast, the aquaculture revenues increase in Sc. 3, where revenues are expected to reach nearly € 13m without having effects on the ecological indicators habitat quality and Natura 2000 (Fig. 6).

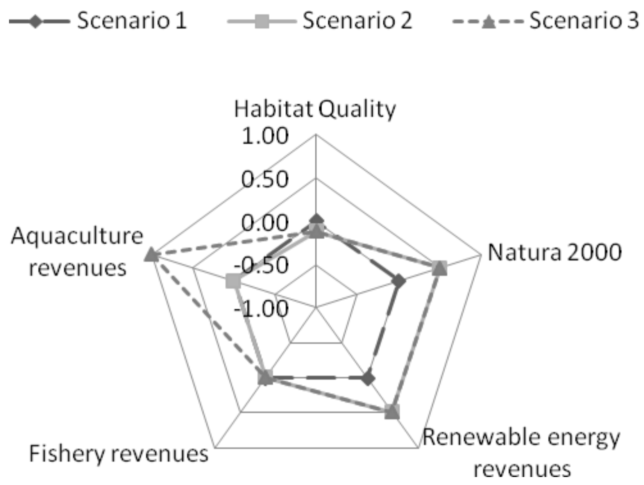


Figure 6: Trade-off between indicator values. Overall change in ES indicators relative to current condition (scenario 1, broken black line) under three alternative management scenarios for the German Bight. Expansions of the shape toward the exterior represent returns relative to the baseline and contractions represent risks. The values are normalised to 1, based on ES indicators given in Tab. 1 for supporting services (km²) and provisioning services (Euro).

The overall trend in ES production generated by management measures gets visible when aggregating the ES to supporting and provisioning services (Fig. 7). While Sc. 1 represents the maximum ecological score in supporting 71.15 km², the score decreases down to 58.33 km² in Sc. 2 and 3, respectively ('000 km²). Sc. 2 and 3 represent different estimates for the economic scores. While Sc. 2 with 69.05m (€ 1.48bn) represents a higher marine resource utilization than Sc. 1 with 15.27m (€ 69.41bn), the maximal utilization is reached with 74.20m (€ 69.42bn) in Sc. 3. Balancing sustainable use and ecosystem health, the scenario promoting the highest degree of supporting services and likewise provisioning services can be identified. Consequently, the most efficient scenario tested in the case study can be deployed towards the operational and general objectives identified in Table 1.

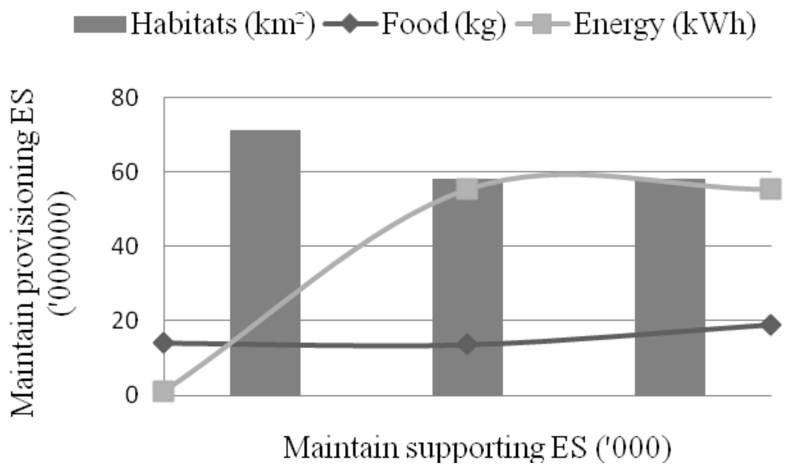


Figure 7: ES trade-offs. Overall trend in ES production relative to current conditions (scenario 1 to the left) under three alternative management scenarios for the German Bight. The area of habitats expected to be sustained is shown on the vertical axis. The values generated per scenario are given in km² for the supporting services and biomass or kWh respectively for the provisioning services.

5.5 Discussion

In this paper the concept of ES is examined based on a literature review and a case study. The focus was on its potential for decision support. Besides, it is tested if competing needs might be balanced in management for both, sustainable use and ecosystem health.

According to the literature sighted, the strength of the ES concept is in its flexibility in harmonizing and integrating multiple interdisciplinary dimensions of knowledge (economic, ecological or socio-economic), which can be visualized to provide basis for planning decisions. It is transparent, has prioritizing features, identifies risks and returns, and facilitates the communication of complex issues. Even when utilized with different wordings, the concept of ES is increasingly advanced to inform decision making related to agricultural, forest or marine management. It has been applied to estimate the impact of changed funding strategies at EU level to support farmers identifying the best economic income sources on parcel and farm level or vice versa – to extract concrete policy proposals. Nevertheless, being not expert-driven or based on empirical modelling, huge efforts have to be expended. Otherwise, the modelling approach is suffering from lacking knowledge, coarse data sets, temporal and spatial scale mismatches or a challenging model validation. Based on the concept of ES, trade-off assessments are frequently used as shown during literature research. The tools and methods applied depend on the issue of concern, being of socio-cultural (Public participation GIS, SoiVes etc.), economic or ecological nature (InVest, MaxEnt). Surprisingly, the majority of studies examined used GIS-based models instead of off-the-shelf tools to support ES evaluation. Not for no reason. While being advantageous in quantifying and illustrating ES trade-offs explicitly, off-the-shelf tools still require system knowledge, skills to edit the input data and operate the model and consequently a high level of expertise.

In the course of the case study, first aspects gained during literature review were confirmed. As it was assumed that there are no economic returns generated on habitats or areas closed for conservation, pure monetary valuations as preferred by many authors were not achievable (Sacchelli et al., 2013; Swetnam et al., 2011; Klain and Chan, 2012; Grêt-Regamey et al., 2012). Consequently, the indicator values were normalised when illustrating the trade-offs. In order to incorporate biophysical metrics as well as real ES trade-offs, the intrinsically services were clustered to ES bundles as shown in Figure 7.

It was found that competing needs could be balanced in careful spatial marine management for both, sustainable use and ecosystem health. Accordingly, the highest risk was identified to be induced by OWF areas activated in Sc. 2 and 3, resulting in habitat decreases. The Natura 2000 sites did not compensate such negative effects on the spawning and nursery grounds as they hardly overlap. Furthermore, those habitats are already facing irreversible alteration in scenario 1. If the area studied in the German Bight would represent 100 % of those essential habitats, the stocks had to face a rocky future. To get the big picture, cross-border modelling would be needed, enabling a transfer of habitat degradation to stock recruitment relationships. The OWF development resulted further in decreasing areas open for fisheries. Both activities, fisheries and renewable energy, exert pressures on the seafloor, altering the benthic habitat structure and the state of benthic communities. Due to OWF development, fisheries will switch to areas not closed for fishing. In turn, the remaining benthic habitats are facing additional pressure as well. As a consequence thereof, slow growth rates of juvenile fish could affect the recruitment and the SSB later on. Moreover, food from fisheries could decline on a long term as not being supported by the ecosystem anymore. In Sc. 3, the ES indicator ‘aquaculture’ is gaining from the spatial synergy with OWFs, causing no additional charge at the cost of the habitats. Here, additional economic or ecological returns caused by IMTA techniques are not included yet. From a conservational perspective Sc. 1 represents the maximum ecological score in supporting services. From an economic perspective, Sc. 3 represents the highest marine resource utilization. Further, the scenario promotes the highest degree of supporting services and likewise provisioning services, balancing sustainable use and ecosystem health.

As a future perspective, an adaption of the fishery sector to spatial closures needs to be integrated. In Stelzenmüller et al. (2014), an increased disturbance in 8 % of the remaining area due to a shift of MBC fisheries, caused by OWF development was assessed. In order to predict its real extent and dynamics, one has to be roughly aware about the direction the fisheries will shift and how the fishery fleets will aggregate around areas closed for fisheries. Further, the case study consists only of a worst case scenario. Positive effects e.g. due to wind farm development, such as MPAs in between turbines, need to be considered in future. Integrating further scenarios would counteract on uncertainties related to human environment interactions, based on parameterization or model outputs, which are quite common. Unfortunately, the illustration of uncertainties when mapping ES or visualizing trade-offs still constitutes a challenge. However, ignoring them can modify decisions and lead to overlooking important management possibilities (Grêt-Regamey et al., 2012). To address uncertainty in the spatial explicit prediction of ES, expert opinion (Grêt-Regamey et al., 2012) and face validity maps (Van der Biest et al., 2014) can be integrated.

Aiming to empirically prioritize alternative management scenarios at a final stage, stakeholder preferences given to ES can be incorporated (Sacchelli et al., 2013; Sanon et al., 2012). These include preferences of ES which manifest their values as well as threats which could debase those values (Sherrouse et al., 2014; Klain and Chan, 2012; Sanon et al., 2012). Moreover, management strategies can be analysed being ranked by stakeholders. A nice example apposite to this study is given in Ramos et al. (2014), where management objectives were among others ranked (in the same order) as followed: Preserve GES, reduce benthic damaging, enhance friendly energy, competitively of aquaculture. Bringing those in line with final management decisions facilitates the communication and consequently the implementation of strategic plans.

Ecosystem-based MSP shall not only implement multi-sector planning but also multi-objective planning when contributing to the MSFD objective GES. Nevertheless, in contrast to the MSFD, no indicators describing the effectiveness of management measures are given. Accordingly, the best compromise has to be found, having in mind the Green Growth objective. Consequently, guidance towards spatially explicit optimization measures should reflect management efficiency. During case study, as a simplification of reality, only a choice of human activities and habitats was integrated when conducting the trade-off assessment. Nevertheless, conflicts got obvious in all scenarios as they already got the norm for the German Bight.

When analysing the costs and benefits of alternative management strategies in order to support decision making processes, the incorporation of best scientific knowledge is self-evident. The application of the ES concept towards MSP requires the integration of services being highly interdisciplinary (cultural values etc.). The mapping of those constitutes a challenge. Nevertheless, the overall results showed that spatial (GIS) data and ecosystem understanding are mostly sufficient to cluster risk and returns towards a balance in between sustainable use and ecosystem health. Vigerstol and Aukema (2011) rated ES off-the-shelf tools such as InVest even as to be accessible to non-experts, which want to get a general picture of the ES existing. Facing the outcomes of the literature analysis and the effort conducted to set up the case study, opinions are deeply divided on this issue. According to the authors opinion, except some web-based visualization tools the most of the ES models are not yet applicable for decision makers.

Finally, the study proved that the German Bight requires an integrated assessment process. Such an approach has to consider all risk and returns of the direct, indirect and combined effects of human drivers. In regard to the fishery sector a comprehensive analysis defining principal areas for all vessels operating in a given planning area has to be included. Such an integrated assessment is also promoted by the IMP, the European 2020 Strategy, MSFD obliging EU member states to achieve GES by 2020 and the MSP directive to achieve Blue Growth. In turn, this requires member states to conduct an initial assessment of the current state of the marine environment and to develop a strategy for 'Green Growth' as the combined alignment of MSP and GES management strategies should be considered in future planning processes. In addition, a coherent planning and cross-border assessment should be considered.

According to the EEA Executive Director Hans Bruyninckx, "*we need to respect the ecological boundaries of Europe's seas if we want to continue enjoying the benefits we receive. This requires aligning our policy ambitions for economic growth with our policy targets of securing healthy, clean and productive seas. Ultimately, this will entail making fundamental changes in the way we meet our societal needs*" (EEA, 2015).

5.6 Conclusion

Aiming to reduce impacts across sectors and avoid conflicting uses of coastal resources, the concept of ES attracts attention. Its strength is in its flexibility in harmonizing and integrating multiple interdisciplinary dimensions of knowledge (economic, ecological or socio-economic), supporting efficient MSP. The tools presented have the power to aid real life decision making, being problem-focused and needs-driven. Their use increase transparency of decisions made by end users, and increase the commitment of the stakeholders – provided that decisions are made based on environmental needs. Such an integrated assessment is also promoted by the IMP, the European 2020 Strategy, MSFD and the MSP directive.

5.7 Acknowledgements

Many thanks to Torsten Schulze for combining the original VMS and logbook data and Jörg Berkenhagen for providing helpful comments in calculating the market prices of the fishery products. Further many thanks to Michael Ebeling for providing information on the calculation of aquaculture production and Matthias Schaber for giving valuable and constructive comments.

5.8 References

Ban, N. C., Bodtker, K. M., Nicolson, D., Robb, C. K., Royle, K., and Short, C. 2013. Setting the stage for marine spatial planning: Ecological and social data collation and analyses in Canada's Pacific waters. Marine Policy, 39: 11-20.

BSH 2009. The Federal Agency for Shipping and Hydrography (Bundesamt für Seeschifffahrt und Hydrographie, BSH). Map North Sea, http://www.bsh.de/en/Marine_uses/Spatial_Planning_in_the_German_EEZ/documents2/MSP_DE_NorthSea.pdf.

Buck, B. H., Dubois, J., Ebeling, M., Franz, B., Goseberg, N., Hundt, M., Schaumann, P., et al. 2012. Schlussbericht. Mulitple Nutzung und Co-Management von Offshore-Strukturen: Marine Aquakultur und Offshore-Windparks. OOMU-Projekt, Förderkennzeichen 0325206.

Buck, B. H., and Krause, G. 2012. Integration of Aquaculture and Renewable Energy Systems. Encyclopaedia of Sustainability Science and Technology, 1: 511-533.

Buck, B. H., Krause, G., and Rosenthal, H. 2004. Extensive open ocean aquaculture development within wind farms in Germany: the prospect of offshore co-management and legal constraints. Ocean & Coastal Management, 47: 95-122.

BUND, DUH, Greenpeace, NABU, WDC, and WWF 2015. Hintergrund zur Verbändeklage - Schutz den Meeresschutzgebieten in Nord- und Ostsee! https://www.greenpeace.de/sites/www.greenpeace.de/files/publications/verbaendeklage-meeresschutzgebiete-hintergrund-20150127.pdf.

Burkhard, B., Kroll, F., Nedkov, S., and Müller, F. 2012. Mapping ecosystem service supply, demand and budgets. Ecological Indicators, 21: 17-29.

Castro, A. J., Verburg, P. H., Martín-López, B., Garcia-Llorente, M., Cabello, J., Vaughn, C. C., and López, E. 2014. Ecosystem service trade-offs from supply to social demand: A landscape-scale spatial analysis. Landscape and Urban Planning, 132: 102-110.

Coull, K. A., Johnstone, R., and Rogers, S. I. 1998. Fisheries Sensitivity Maps in British Waters. Published and distributed by UKOOA Ltd.

De Meyer, A., Estrella, R., Jacxsens, P., Deckers, J., Van Rompaey, A., and Van Orshoven, J. 2013. A conceptual framework and its software implementation to generate spatial decision support systems for land use planning. Land Use Policy, 35: 271-282.

Depellegrin, D., Blažauskas, N., and Egarter-Vigl, L. 2014. An integrated visual impact assessment model for offshore windfarm development. Ocean & Coastal Management, 98: 95-110.

Douvere, F. 2008. The importance of marine spatial planning in advancing ecosystem-based sea use management. Marine Policy, 32: 762-771.

EC 2012. European Commission. Progress of the EU's Integrated Maritime Policy – Report from the commission to the European Parliament, the council, the European Economic and social committee and the committee of the Regions. Luxembourg: Publications Office of the European Union: 11 pp.

EC 2014a. European Commission. Environment. Marine directive. Good Environmental Status., http://ec.europa.eu/environment/marine/good-environmental-status/index_en.htm.

EC 2014b. European Commission. Maritime Affairs. Policy. Blue growth., http://ec.europa.eu/maritimeaffairs/policy/blue_growth/.

EC 2014c. European Commission. Environment. Nature & Biodiversity. EU Biodiversity Strategy to 2020 - towards implementation, http://ec.europa.eu/environment/nature/biodiversity/comm2006/2020.htm.

EC 2015. European Commission. Aquatic Ressources, http://ec.europa.eu/programmes/horizon2020/en/area/aquatic-resources.

EEA 2015. European Environment Agency, State of Europes Seas. Luxembourg: Publications Office of the European Union.

Ehler, C., and Douvere, F. 2009. Marine Spatial Planning. *In* A Step-by-Step Approach toward Ecosystem-based Management, p. 99. Ed. by I. O. Commission. Intergovernmental Oceanographic Commission.

Fock, H. O. 2011. Natura 2000 and the European Common Fisheries Policy. Marine Policy, 35: 181-188.

Foley, M. M., Halpern, B. S., Micheli, F., Armsby, M. H., Caldwell, M. R., Crain, C. M., Prahler, E., et al. 2010. Guiding ecological principles for marine spatial planning. Marine Policy, 34: 955-966.

Geneletti, D. 2013. Assessing the impact of alternative land-use zoning policies on future ecosystem services. Environmental Impact Assessment Review, 40: 25-35.

Gimpel, A., Stelzenmuller, V., Cormier, R., Floeter, J., and Temming, A. 2013. A spatially explicit risk approach to support marine spatial planning in the German EEZ. Marine environmental research, 86: 56-69.

Gimpel, A., Stelzenmüller, V., Grote, B., Buck, B. H., Floeter, J., Núñez-Riboni, I., Pogoda, B., et al. 2015. A GIS modelling framework to evaluate marine spatial planning scenarios: Co-location of offshore wind farms and aquaculture in the German EEZ. Marine Policy, 55: 102-115.

Grêt-Regamey, A., Brunner, S. H., Altwegg, J., and Bebi, P. 2012. Facing uncertainty in ecosystem services-based resource management. International Journal of Biodiversity Science, Ecosystem Services & Management, 1–15.

Grêt-Regamey, A., Celio, E., Klein, T. M., and Wissen Hayek, U. 2013. Understanding ecosystem services trade-offs with interactive procedural modeling for sustainable urban planning. Landscape and Urban Planning, 109: 107-116.

Guerry, A. D., Ruckelshaus, M. H., Arkema, K. K., Bernhardt, J. R., Guannel, G., Kim, C.-K., Marsik, M., et al. 2012. Modeling benefits from nature: using ecosystem services to inform coastal and marine spatial planning. International Journal of Biodiversity Science, Ecosystem Services & Management, 8: 107-121.

Haines-Young, R., Potschin, M., and Kienast, F. 2012. Indicators of ecosystem service potential at European scales: Mapping marginal changes and trade-offs. Ecological Indicators, 21: 39-53.

Halpern, B. S., Diamond, J., Gaines, S., Gelcich, S., Gleason, M., Jennings, S., Lester, S., et al. 2012. Near-term priorities for the science, policy and practice of Coastal and Marine Spatial Planning (CMSP). Marine Policy, 36: 198-205.

Halpern, B. S., Walbridge, S., Selkoe, K. A., Kappel, C. V., Micheli, F., D'Agrosa, C., Bruno, J. F., et al. 2008. A global map of human impact on marine ecosystems. Science, 319: 948-952.

Hermann, A., Kuttner, M., Hainz-Renetzeder, C., Konkoly-Gyuró, É., Tirászi, Á., Brandenburg, C., Allex, B., et al. 2014. Assessment framework for landscape services in European cultural landscapes: An Austrian Hungarian case study. Ecological Indicators, 37: 229-240.

Hilde, T., and Paterson, R. 2014. Integrating ecosystem services analysis into scenario planning practice: accounting for street tree benefits with i-Tree valuation in Central Texas. J Environ Manage, 146: 524-534.

Hobohm, J., Krampe, L., Frank, P., Gerken, A., Heinrich, P., and Richter, M. 2013. Prognos & Fichtner Studie. Kostensenkungspotenziale der Offshore-Windenergie in Deutschland. Langfassung. Stiftung OFFSHORE-WINDENERGIE und Partner, 118 pp.: Retrieved: July 18th 2015, 2023:2000.

Hoyer, R., and Chang, H. 2014. Assessment of freshwater ecosystem services in the Tualatin and Yamhill basins under climate change and urbanization. Applied Geography, 53: 402-416.

ICES 2014a. International Council for the Exploration of the Sea. Plaice in Subarea IV (North Sea) (updated). Advice Novemer 2014, http://www.ices.dk/sites/pub/Publication%20Reports/Advice/2014/2014/ple-nsea.pdf.

ICES 2014b. International Council for the Exploration of the Sea. Sandeel in Division IIIa and Subarea IV. Advice February 2014, http://www.ices.dk/sites/pub/Publication%20Reports/Advice/2014/2014/san_34.pdf.

ICES 2014c. International Council on the Exploration of the Sea. Sole in Subarea IV (North Sea) (updated). http://www.ices.dk/sites/pub/Publication%20Reports/Advice/2014/2014/sol-nsea.pdf, Retrieved: July 6th 2015, 18:28.

Jackson, B., Pagella, T., Sinclair, F., Orellana, B., Henshaw, A., Reynolds, B., McIntyre, N., et al. 2013. Polyscape: A GIS mapping framework providing efficient and spatially explicit landscape-scale valuation of multiple ecosystem services. Landscape and Urban Planning, 112: 74-88.

Katsanevakis, S., Stelzenmüller, V., South, A., Sørensen, T. K., Jones, P. J. S., Kerr, S., Badalamenti, F., et al. 2011. Ecosystem-based marine spatial management: Review of concepts, policies, tools, and critical issues. Ocean & Coastal Management, 54: 807-820.

Klain, S. C., and Chan, K. M. A. 2012. Navigating coastal values: Participatory mapping of ecosystem services for spatial planning. Ecological Economics, 82: 104-113.

Klug, H., and Jenewein, P. 2010. Spatial modelling of agrarian subsidy payments as an input for evaluating changes of ecosystem services. Ecological Complexity, 7: 368-377.

Koschke, L., Fürst, C., Frank, S., and Makeschin, F. 2012. A multi-criteria approach for an integrated land-cover-based assessment of ecosystem services provision to support landscape planning. Ecological Indicators, 21: 54-66.

Kovács, E., Kelemen, E., Kalóczkai, Á., Margóczi, K., Pataki, G., Gébert, J., Málovics, G., et al. 2014. Understanding the links between ecosystem service trade-offs and conflicts in protected areas. Ecosystem Services.

Labiosa, W. B., Forney, W. M., Esnard, A. M., Mitsova-Boneva, D., Bernknopf, R., Hearn, P., Hogan, D., et al. 2013. An integrated multi-criteria scenario evaluation web tool for participatory land-use planning in urbanized areas: The Ecosystem Portfolio Model. Environmental Modelling & Software, 41: 210-222.

Lauf, S., Haase, D., and Kleinschmit, B. 2014. Linkages between ecosystem services provisioning, urban growth and shrinkage – A modeling approach assessing ecosystem service trade-offs. Ecological Indicators, 42: 73-94.

MA 2005. Millennium Ecosystem Assessment. Ecosystems and Human Well-being: Synthesis. Island Press, Washington, DC.

Maes, J., Egoh, B., Willemen, L., Liquete, C., Vihervaara, P., Schägner, J. P., Grizzetti, B., et al. 2012. Mapping ecosystem services for policy support and decision making in the European Union. Ecosystem Services, 1: 31-39.

Mayer, I., Zhou, Q., Lo, J., Abspoel, L., Keijser, X., Olsen, E., Nixon, E., et al. 2013. Integrated, ecosystem-based Marine Spatial Planning: Design and results of a game-based, quasi-experiment. Ocean & Coastal Management, 82: 7-26.

McLeod, K. L., Lubchenco, J., Palumbi, S. R., and Rosenberg, A. A. 2005. Scientific Consensus Statement on Marine Ecosystem-Based Management. Signed by 221 academic scientists and policy experts with relevant expertise and published by the Communication Partnership for Science and the Sea at http://compassonline.org/?q=EBM<EBM_Consensus_Statement_v12.pdf>.

Nackoney, J., and Williams, D. 2013. A comparison of scenarios for rural development planning and conservation in the Democratic Republic of the Congo. Biological Conservation, 164: 140-149.

Neuenschwander, N., Wissen Hayek, U., and Grêt-Regamey, A. 2014. Integrating an urban green space typology into procedural 3D visualization for collaborative planning. Computers, Environment and Urban Systems, 48: 99-110.

Palomo, I., Martín-López, B., Potschin, M., Haines-Young, R., and Montes, C. 2013. National Parks, buffer zones and surrounding lands: Mapping ecosystem service flows. Ecosystem Services, 4: 104-116.

Polasky, S., Nelson, E., Camm, J., Csuti, B., Fackler, P., Lonsdorf, E., Montgomery, C., et al. 2008. Where to put things? Spatial land management to sustain biodiversity and economic returns. Biological Conservation, 141: 1505-1524.

Ramos, J., Soma, K., Bergh, O., Schulze, T., Gimpel, A., Stelzenmuller, V., Makinen, T., et al. 2014. Multiple interests across European coastal waters: the importance of a common language. ICES Journal of Marine Science, 72: 720-731.

Roces-Díaz, J. V., Díaz-Varela, E. R., and Álvarez-Álvarez, P. 2014. Analysis of spatial scales for ecosystem services: Application of the lacunarity concept at landscape level in Galicia (NW Spain). Ecological Indicators, 36: 495-507.

Sacchelli, S., De Meo, I., and Paletto, A. 2013. Bioenergy production and forest multifunctionality: A trade-off analysis using multiscale GIS model in a case study in Italy. Applied Energy, 104: 10-20.

Sanon, S., Hein, T., Douven, W., and Winkler, P. 2012. Quantifying ecosystem service trade-offs: the case of an urban floodplain in Vienna, Austria. J Environ Manage, 111: 159-172.

Sherrouse, B. C., Clement, J. M., and Semmens, D. J. 2011. A GIS application for assessing, mapping, and quantifying the social values of ecosystem services. Applied Geography, 31: 748-760.

Sherrouse, B. C., Semmens, D. J., and Clement, J. M. 2014. An application of Social Values for Ecosystem Services (SolVES) to three national forests in Colorado and Wyoming. Ecological Indicators, 36: 68-79.

Stelzenmüller, V., Fock, H. O., Gimpel, A., Seidel, H., Diekmann, R., Probst, W. N., Callis, U., et al. 2014. Quantitative environmental risk assessments in the context of marine spatial management: Current approaches and some perspectives. ICES Journal of Marine Science.

Stelzenmüller, V., Schulze, T., Fock, H. O., and Berkenhagen, J. 2011. Integrated modelling tools to support risk-based decision-making in marine spatial management. Marine Ecology Progress Series, 441: 197-212.

Swetnam, R. D., Fisher, B., Mbilinyi, B. P., Munishi, P. K., Willcock, S., Ricketts, T., Mwakalila, S., et al. 2011. Mapping socio-economic scenarios of land cover change: a GIS method to enable ecosystem service modelling. J Environ Manage, 92: 563-574.

Van der Biest, K., D'Hondt, R., Jacobs, S., Landuyt, D., Staes, J., Goethals, P., and Meire, P. 2014. EBI: An index for delivery of ecosystem service bundles. Ecological Indicators, 37: 252-265.

Van Riper, C. J., and Kyle, G. T. 2014. Capturing multiple values of ecosystem services shaped by environmental worldviews: a spatial analysis. J Environ Manage, 145: 374-384.

Vigerstol, K. L., and Aukema, J. E. 2011. A comparison of tools for modeling freshwater ecosystem services. J Environ Manage, 92: 2403-2409.

Wainger, L. A., King, D. M., Mack, R. N., Price, E. W., and Maslin, T. 2010. Can the concept of ecosystem services be practically applied to improve natural resource management decisions? Ecological Economics, 69: 978-987.

White, C., Halpern, B. S., and Kappel, C. V. 2012. Ecosystem service tradeoff analysis reveals the value of marine spatial planning for multiple ocean uses. PNAS, 109: 4696–4701.

6. Manuscript 5: Multiple interests across European coastal waters: The importance of a common language

Jorge Ramos[a], Katrine Soma[b,h], Øivind Bergh[c], Torsten Schulze[d], Antje Gimpel[d], Vanessa Stelzenmüller[d], Timo Mäkinen[e], Gianna Fabi[f], Fabio Grati[f] and Jeremy Gault[g]

[a] IPMA, Av. 5 Outubro, s/n, 8700-305 Olhão, Portugal

[b] LEI, PO Box 29703, 2502 LS The Hague, The Netherlands

[c] IMR, PO Box 1870 Nordnes, NO-5817 Bergen, Norway

[d] TI-SF, Palmaille 9, 22767 Hamburg, Germany

[e] RKTL, P.O.BOX 2, 00791 Helsinki, Finland

[f] CNR-ISMAR, Largo Fiera della Pesca, 2, 60125, Ancona, Italy

[g] CMRC - UCC, Western Road, Cork Co. Cork, Ireland.

[h] PAP, Wageningen University, Hollandseweg 1, 6706 KN Wageningen, The Netherlands

ICES Journal of Marine Science (2015), *72 (2): 720-731*

6.1 Abstract

Different marine and coastal activities have diverse economic, environmental and socio-cultural objectives, which can lead to conflict when these multi-dimensional activities coincide spatially or temporally. This is sometimes driven by a lack of understanding or other users' needs and consequentially adequate planning and the utilization of a common language is essential. By using a transparent approach based on multi-criteria analysis (MCA), we characterize and establish priorities for future development/conservation for all users in the coastal area using six representative European Case Studies with different levels of complexity. Results varied according to location, but significantly it was found that stakeholders tended to favour ecological and social over economic objectives. This paper outlines the methodology employed, the results derived and the potential for this approach to reduce conflict in coastal and marine waters.

Keywords: Conflict (reduction), Case studies (CS), COEXIST, European coastal zone, marine spatial planning (MSP), multi-criteria analysis (MCA), stakeholders

6.2 Introduction

The use of European marine and coastal areas varies from traditional activities such as fishing and trade shipping, to more recent technical developments of green energy production (Ehlers and Lagoni, 2006). Demand for clean energies has progressed due to the public concerns about the sustainability of energy use (Pinkse and Dommisse, 2008). As a result of the increasing complexity of use, competition for space and for actual or perceived potential resources in the marine and coastal areas, there is an urgent need for coexistence among the different activities (Dempster and Sanchez-Jerez, 2008). This challenge is further complicated by the different degrees of acceptance by different parts of the society about decisions on marine and coastal uses (Brown *et al.*, 2002). It has been shown, however, that there is greater social acceptance when increased transparency is established in the planning and decision-making processes (Curtin and Meijer, 2006). Marine spatial planning (MSP) needs room for a compulsory conciliation, and a compromise of not only sustainable, but also intentional and efficient use of resources (Ostrom *et al.*, 1999). More recently Foley *et al.* [(2010:2) after Douvere (2008)] defined ecosystem based MSP as “an integrated planning framework that informs the spatial distribution of activities in and on the ocean in order to support current and future uses of ocean ecosystems and maintain the delivery of valuable ecosystem services for future generations in a way that meets ecological, economic and social objectives”.

Within the process of planning, conflicts between public and private stakeholders may occur (Pinho, 2007) and different stakeholder types might interact either negatively or positively and a plethora of dissimilar interests may arise (Reed *et al.*, 2009). The outcome of this it that information can appear too complicated to policy makers who therefore make their decisions independently, based on their own experience.

The COEXIST project (Interaction in Coastal Waters: A roadmap to sustainable integration of aquaculture and fisheries) engaged stakeholders from six representative European Case Studies (henceforth CS). The project focused on the interaction of different human activities, conflicting or synergistic, and facilitated interaction between diverse sectors in the coastal zone across several European countries.

The objective of this paper is to apply a 'common language' – in this case a multi-criteria analysis (MCA) approach – designed to ascertain the different stakeholder views and preferences, from different countries, with regard to sustainable use of coastal areas (Soma *et al*., 2013). In the MCA approach used, firstly the legislative framework is identified in general and in specific terms (in each CS). Secondly, under three main overarching objectives, – economic, ecological and socio-cultural –, stakeholder preferences for a range of sub-objectives were determined. Thirdly, the preference patterns were collated by CS and by stakeholder group. Finally, the sub-objective preferences were ranked in each CS.

6.3 Multi-Criteria Analysis: State-of-the-Art

MCA emerged because of the need to develop techniques to be used in processes where difficult decisions about alternative strategies have to be taken (Nijkamp, 1975; Van Delft, 1977; Kickert, 1978). MCA identify each of the choices made under a range of objectives (or sub-objectives) and assign a value to the relative importance of this choice with respect to each objective.

In order to determine the relative importance of the objectives selected, pre-determined multiple choice options are required (Hajkowicz and Collins, 2007). These are subsequently deployed as part of the evaluation process, can be conducted out by diverse individual stakeholders or stakeholder groups and commonly involve a multidisciplinary team (Munda, 2004).

Once the stakeholders’ decisions have been obtained, several methods for judgements can be used to rank preferences (Yan et al., 2007; Shakhnov, 2008) and/or to make pairwise comparisons (Deng, 1999; Macharis et al., 2004; Soma, 2010; Saaty and Vargas, 2013). These methods of judgements are advocated within the MCA scope as suitable for decision problems and for the inclusion of stakeholders' views (Linkov et al., 2006; Hajkowicz and Higgins, 2008).

However, there is a challenge when the frames and understandings of the reality of stakeholders are influenced by their different and sometimes conflicting views, goals and demands (Lahdelma et al., 2000; Mendoza and Prabhu, 2005). In addition, there is criticism of the approach relating to the inconsistencies derived from essentially judgement calls (Mendoza and Martins, 2006).

6.4 Multi-Criteria Analysis for Marine Spatial Decision-Making Processes

Marine spatial planning (MSP) is becoming important not only in Europe but also worldwide due to the needs of different societies have to address marine management concerns (Peel and Lloyd, 2004; Douvere, 2008; Kidd and Shaw, 2013). Some authors advocate that as MSP is a relatively new process that requires adequate and practical tools to be used in the inherent decision processes (e.g. Kidd and Ellis, 2012; Stelzenmüller *et al.*, 2013). Smith *et al.* (2011) suggest that MSP should be part of an integrated terrestrial and marine approach, however, Janßen *et al.* (2013) insist that unlike its terrestrial counterpart, MSP does not present meaningful delimitation of planning areas (apart from somehow vague terms for 'inshore' or 'offshore'), and consequently the adequate management of human activities remains a challenging process.

Some authors (e.g. O'Riordan *et al.*, 2005; Hedelin, 2007) highlight that the potential of applying MCA in decision process dilemmas is justified. The reason being is that MCA allows the inclusion of multiple and complex criteria belonging to different dimensions at a specific location to support analysis and subsequent judgement (Table 1).

While the use of MCA tools in MSP has been recorded for over a decade, more recently, models and other experimental tools have focused not only on the interactions between sectors, such as fisheries and conservation (see for example Klein *et al.*, 2009), but also on diversified human impacts on the marine environment (see for example Ruiz-Frau *et al.*, 2011; Stelzenmüller *et al*., 2011). Douvere and Ehler (2009) advocate the increasing need for new location based strategies in MSP for Europe. To achieve this, new tools will be required and particularly those that bring together stakeholders' views with different activity sectors and spatial contexts (Berkes, 2009; Molle, 2009). Strategies which can enhance accountability, legitimacy and transparency throughout decision making processes are particularly relevant (Soma, 2010; Sparrevik *et al.*, 2011).

In response to these challenges, we believe that the use of a tailored MCA approach developed specifically for the purposes of coexistence in European waters, could be of significant value to the MSP process given its ability to deal with the choices derived from various, and sometimes conflicting criteria. In this MCA approach, in order to deal with incommensurable value dimensions of the criteria, we compare along a scale of 'importance' (Munda, 2004). This is essential as whilst it may be wholly plausible to suggest that social aspects are more important than the economic considerations at a specific site, it is sometimes complicated to attribute monetary values on social dimensions to enable accurate comparison.

The common methodology developed in the COEXIST project has benefited from the trans-national and cross-disciplinary collaboration of the consortium. This stakeholder based MCA approach was adapted to reflect local circumstances in each CS area to facilitate information collection. The main sources of information stemmed from the local stakeholders identified in each case study location and included sector representatives, public managers, researchers and NGOs.

6.5 Methodological approach

6.5.1 Conceptual model

The methodological approach used is part of a multi-criteria analysis (MCA) based on the COEXIST framework and is outlined below. While the complete MCA framework accounts for both the spatial and temporal dimensions, the institutional dimension was central to the success of the analysis. The conceptual model used can be arranged into a hierarchical structure as depicted by Figure 1. The development of the hierarchy starts with the definition that the ultimate goal in each CS, was to 'sustain a viable coastal / marine ecosystem' in their geographic area, aiming for long term coexistence of stakeholders with differing local agendas (economic, social or environmental). In a broader sense a sustainable use of the resource refers not only to activities, but also to achieving or preserving relevant values, such as: competitive economic activities and infrastructures that are utilized, healthy environmental status and good living standards (level 1).

In addition and with direct relevance to sustainability, there are already a substantial number of legislative frameworks and spatial plans in place, which must be taken into account locally, regionally, nationally or even broader scale. These plans were identified and collected in each CS (level 2).

Table 1: Examples of coastal and marine planning dimensions and main objectives.

	Dimensions (activities, actions and people)		
Resources	**Temporal**	**Spatial**	**Institutional**
Economic: biological, energetic and geological exploitation	- Fishermen livelihoods - Trade of goods and services - Energy consumption	- Fishing grounds - Trade routes - Gas and oil fields	- Fishermen, producers organizations - Shipping industry - Energy production companies
Environmental: biodiversity protection, clean seawater, migratory routes	- Biological spawning periods - Search for biomaterials: paint/ fuel - Resources conservation	- Wild areas - MPAs - Nursery areas	- Fishery-dependent communities (FDCs) - Marine biologists - Environmentalists
Socio-cultural: clean sandy beaches, bath-able seawater, pleasant coastal landscapes	- Seasonal holidays - Annual sports competitions - Cultural and gastronomic events	- Beach recreation areas - Sailing routes - Architectural and historic places - Coastal summer houses	- Tourists - Sportspeople - Local city councils

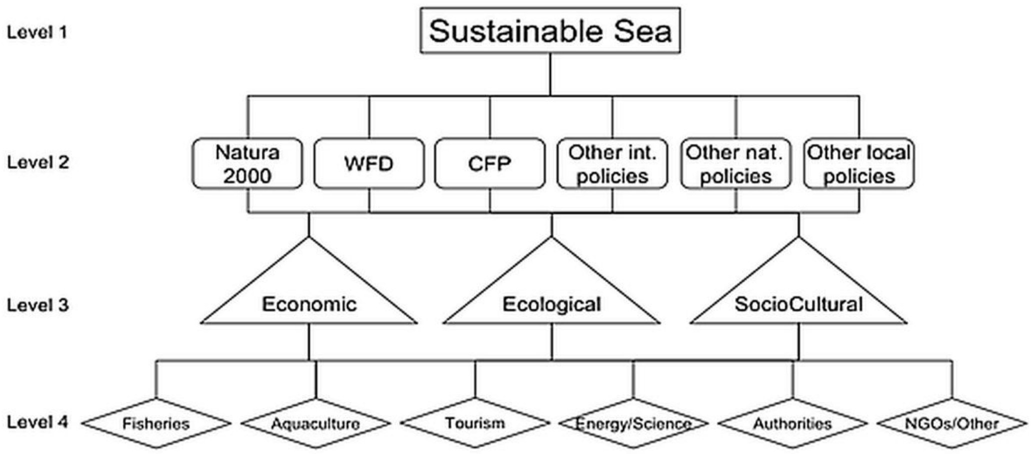


Figure 1: Conceptual model based on the COEXIST framework showing the different levels of analysis for the different case studies: the main goal to achieve, the sets of legislative frameworks/spatial plans consulted, the main objectives addressed and the groups of stakeholders involved in the process. Source: COEXIST (2011).

Each CS developed a specific hierarchical structure, although at a general level the objectives (level 3) are similar in all CS, comprising economic, ecological, and socio-cultural dimensions. Each of these general objectives is subsequently split into more specific objectives and sub-objectives. For instance, in the economic category, stakeholders identified objectives for allowing further developments of the main economic activities in their coastal areas, so for competitiveness, the economic sub-objectives also included issues of infrastructure improvements. In the ecological category the sub-objectives included ensuring good water quality and conditions conducive for living resources (such as fish). Issues related with the preservation of resources as well as pollution control were also relevant and therefore included. When considering the socio-cultural category, issues of employment, constructions or heritage, and lifestyle and healthy living were seen as pertinent.

Finally, a broad range of stakeholders were identified and categorized (level 4). It is important to stress that identification of stakeholders is a pre-requisite of this approach and ideally should be done before identifying the hierarchy. However for completeness when describing this conceptual model we listed the stakeholder groups in the hierarchy below.

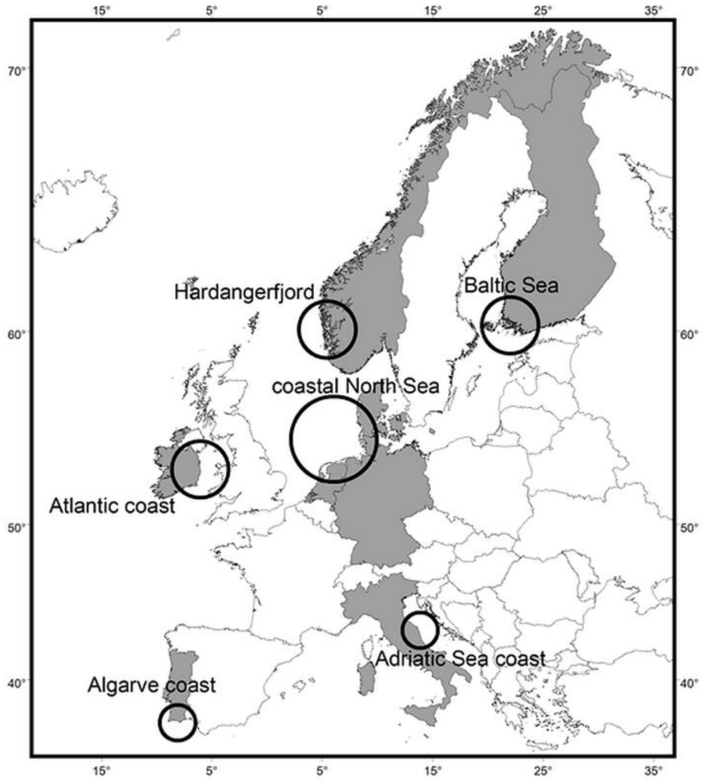


Figure 2: The case study sites of the FP7 COEXIST project that have applied the framework are represented by the circles (in grey are depicted the respective countries involved in the process). Source: Bergh et al. (2012).

6.5.2 Case-studies

The CS in the COEXIST project (Figure 2) that applied, adapted and conducted the framework were at different scales and included: the Hardangerfjord (Norway), the Atlantic coast (Ireland only), the Algarve coast (Portugal), the Adriatic Sea coast (Italy), the North Sea coast (comprising Denmark, Germany and the Netherlands) and the Baltic Sea (Finland) (Bergh *et al.* 2012).

6.5.3 Primary data sources

In order to perceive preferences on coastal planning, a common questionnaire was developed based on a hierarchical disposition of objectives. Accordingly, a set of questions was adapted to each case-study context and specificities. The questions were structured around three objectives (economic, ecological and socio-cultural), and their respective sub-objectives (Table 2). Then local stakeholders were invited to answer the questionnaire, in the context of sustaining a viable coast/sea in their location. These preferences were analysed using a pairwise comparison with a 9-point scale, as suggested by Soma (2003; 2010)

6.5.4 Secondary data sources

Relevant policy documents and legislation were identified; at international, European, regional, national, and local levels (COEXIST, 2011). It should be noted that at a broader level, legal frameworks do not apply evenly to all CS. For instance, the Water Framework Directive (WFD) was common to all CS; whereas the Common Fisheries Policy (CFP) was relevant to all, except CS1 – Hardangerfjord (Norway), which is a non-EU country. Similarly, the Convention for the Protection of the Marine Environment of the North-East Atlantic (OSPAR) was considered in just four CS, exceptions were CS4 – Adriatic Sea (Italy) as Italy is not a signatory state, and CS6 – Baltic Sea (Finland) as although Finland is a signatory of the OSPAR convention, the Baltic Sea is not part of the territory defined under OSPAR.

Table 2: General example of a hierarchy of objectives used in each case study.

Main goal	**Sustainable sea**		
Main objectives	Economic	Ecological	Socio-cultural
	Obtain profitable enterprises	Conserve healthy ecosystems	Preserve high living qualities
Specific objectives	Increase profitability of firms	Ensure good water quality	Ensure coastal employment
Sub-objectives	Increase competitivity of fisheries	Avoid ballast waters	(...)
	Increase competitivity of tourism	Control water pollution	(...)
(...)	(...)	(...)	(...)
(...)	(...)	(...)	(...)

Source: COEXIST (2011).

6.5.5 Stakeholder preferences through a MCA approach

In the present analysis all graphical computations were performed by using R version 2.14.1 (R Development Core Team, 2011), and were carried out as follows:

i. First, the collected answers derived from pairwise comparisons from the stakeholder questionnaires were scaled from 0 to 100.

ii. Second, ternary plots were built aiming to display stakeholders' average position score of the economic, ecological and socio-cultural core objectives across all CS (Graffelman and Camarena, 2007; Koleff *et al.*, 2003; Palmer *et al.*, 2007). The closer a point falls to a vertex, the greater the stakeholder is attached to the objective of that particular vertex. The location of the plot provides the stakeholder's return on the relative importance of the three main objectives (Zafonte and Sabatier, 2004). Continuous data depicted in ternary plots are further analysed in order to find dissimilarities among case-studies. Seefeld (2007) suggests that dissimilarity can be measured by using distance metrics through the method of the Euclidean distance. Coetzer *et al.* (2012) suggest that the Euclidean distance is generally accepted as the most common measure of dissimilarity in the literature. Considering the three-dimensional average points for economic (x), socio-cultural (y) and ecological (z) dimensions, and given that p and q are two case-studies being compared, then:

$$p = (p_x, p_y, p_z) \quad (1)$$

$$q = (q_x, q_y, q_z) \quad (2)$$

The Euclidean distance is computed as:

$$d(p,q) = \sqrt{(p_x - q_x)^2 + (p_y - q_y)^2 + (p_z - q_z)^2}. \quad (3)$$

Next, the analysis weighed up all the main objectives for each CS depicted in the ternary plots and measured the Euclidean distances of their average points. The Euclidean distance between two points in a Cartesian space measures the dissimilarity between pairs of patterns. The value of the distance indicates the extend to which pairs of patterns differ from each other. Smaller dissimilarity between two patterns is indicative of higher visual similarity between the patterns (Honarkhah and Caers, 2010).

iii. Third, different stakeholder preference attributes to each objective across CS were initially depicted in a heat plot and then allocated into clusters (dendrograms). Hentschel and Page (2003) suggest that this technique allows the recognition of patterns (i.e. between CS and stakeholder groups in the present approach). Stakeholder groups from different CS who are similar in terms of their preferences for the main objectives will be located close together in the heat plots. In order to better understand the results, a discrete and a continuous scale for stakeholder group preferences were defined where 0 preference corresponded to 'black square' and 100 preference matched the 'white square' with all the preferences in between varying in different grey hues. Dendrograms show that the most similar elements are merged hierarchically in single clusters. The order of the clusters formed indicates the patterns and relations between the elements. Similarities between elements can also be measured in dendrograms by using Euclidean distances.

iv. Fourth, by sorting sub-objective preferences in descending order, each CS box-plot and whiskers graph shows the range of variation between percentiles. The outliers identify inconsistencies. In this analysis the CS are independent from each other, and the analysis accounts only for the number of stakeholder respondents and the chosen number of sub-objectives. Some of the sub-objectives may be similar across

CS, whereas others may not (i.e., they only make sense in the particular CS context).

6.6 Results

6.6.1 Legislation applied in each case-study

Despite the context differences, there are several legal frameworks that are common to various CS, which are designed to regulate the diverse range of activities and these are often in parallel with more local frameworks that have the intention to address and regulate local problems at a more granular level (Table 3).

Across the different CS in this study, stakeholders involved in the coastal planning and management process have their own sectoral interests and have diverse backgrounds (COEXIST, 2011). These stakeholders typically belong to the private or operational sector, the governmental or public sector, and non-governmental organizations (NGOs). In order to advance the analysis, stakeholder sectors were grouped under the following categories: fisheries, aquaculture, tourism, authorities, energy and science, NGOs and other marine related activities. It is worth noting that stakeholders representing sectors such as shipping, transportation and sand mining were also approached, but due to low returns from these sectors results are grouped under 'others'. Questionnaire responses are presented in Table 4.

Table 3: Identification of relevant legislation found for each case-study.

Case Study	Goal	Large range spatial plans	National and local plans	Main activities and stakeholders
CS1 – Hardangerfjord	Sustainable sea	WFD, Natura 2000, OSPAR	NPBA, CZP, ACTS	Fisheries, Aquaculture, Energy (hydroelectric), Tourism
CS2 – Atlantic Coast of Ireland	Idem	WFD, Natura 2000, CFP, OSPAR	FR, HSBC, OREDP, ABWFL,SACs/SPAs	Fisheries, Aquaculture, Energy (off-shore wind parks), Tourism
CS3 – Algarve Coast	Idem	WFD, Natura 2000, CFP, OSPAR	POOC, POPNRF, PGRH, POEM	Fisheries, Aquaculture, Tourism
CS4 – Adriatic Sea	Idem	WFD, Natura 2000, CFP	RPFA, RPHD, NLTF, NPHD, ZTB, NLCMPA	Fisheries, Aquaculture, Energy (off-shore gas), Tourism
CS5 – Coastal North Sea	Idem	WFD, Natura 2000, CFP, OSPAR	PB, NPs, NSG, MSP, DFL, MPV, IMPNS, MDPDWS, PDNS	Fisheries, Aquaculture, Energy (off-shore oil, gas and wind parks), Shipping
CS6 – Baltic Sea	idem	WFD, Natura 2000, CFP, OSPAR	EPS, ESSWF, FMP, NADP, OPs	Fisheries, Aquaculture, Energy (hydroelectric), Tourism

Source: COEXIST (2011).
NPBA, National Planning and Building Act of 2008; CZP, The coastal-zone plan - at municipality and county levels; ACTS, Several Acts: The Aquaculture Act (Law of 17. June 2005 No. 79); The Food Act (Law of 19. December 2003 No. 124); The Animal Welfare Act (Law of 19. June 2009 No. 197); The Pollution Act (Law of 13. March 2003 No. 6); The Harbour and Waters Act (Law of 17. April 2009 No. 19); FR, fisheries restriction; HSBC, herring spawning box closure; OREDP, Offshore Renewable Energy Development Plan; ABWFL, Arklow Bank windfarm Foreshore Lease; SACs/SPAs, various SACs and SPAs in case study area; POOC, Coastal Edge Management Plan; POPNRF, Ria Formosa Natural Park Management Plan; PGRH, Hydrographic Region Management Plan; POEM, Maritime Space and Activities Plan; RPFA, Regional Plan for Fisheries and Aquaculture; RPHD, Regional Plan for Hydraulic Dredges; NLTF, National Law for Trawl Fisheries; NPHD, National Plan for

Hydraulic Dredges; NTB, National Law for the Creation of Fishing Protected Areas; NLCMPA, National Law for the Creation of MPA Areas; PB, Plaice Box; NPs, National Parks: Schleswig-Holsteinisches Wattenmeer, Niedersa¨chsisches Wattenmeer, Hamburgisches Watternmeer; NSG, NSG Helgoland; MSP, Marine Spatial Plan German EEZ; DFL, Danish Fishery Law; MPV, Management Plan Voordelta; IMPNS, Integrated Management Plan North Sea 2015; MDPDWS, Management and development plan for the Dutch Wadden Sea; PDNS, Policy Document on the North Sea 2009–2015; EPS, environmental permit system; ESSWF, environmental strategy of the South-Western Finland 2021; FMP, fisheries use and management plans; NADP, national aquaculture development programme 2015; Ops, other plans: local coastal master plan; local detailed coastal plan; local detailed plan; local master plan; military areas; MPAs: national commercial fishing development programme 2015; national parks.

Table 4: Number of questionnaire respondents and their distribution by stakeholder group.

Stakeholder group	Fisheries	Aquaculture	Tourism	Authorities	NGOs/Other	Energy/ Science	TOTAL
Case study							
CS1 – Hardangerfjord (Norway)	1	2	1	1	1	1	7
CS2 – Atlantic coast (Ireland)	1	1	2	1	2	1	8
CS3 – Algarve coast (Portugal)	3	8	2	2	6	4	25
CS4 – Adriatic Sea coast (Italy)	2	1	4	4	2	1	14
CS5 – North Sea coast (Denmark, Germany and The Netherlands)	8	2	3	12	6	12	43
CS6 – Baltic Sea (Finland)	2	2	1	2	1	2	10
TOTAL	17	16	13	22	18	21	107

Source: COEXIST (2011).

6.6.2 Preferred objectives by case-study

Weighing stakeholders' views is fundamental in order to determine their position (COEXIST, 2012). We treated stakeholders as having similar importance, and only considered the relative preference they gave to the different objectives (Figure 3).

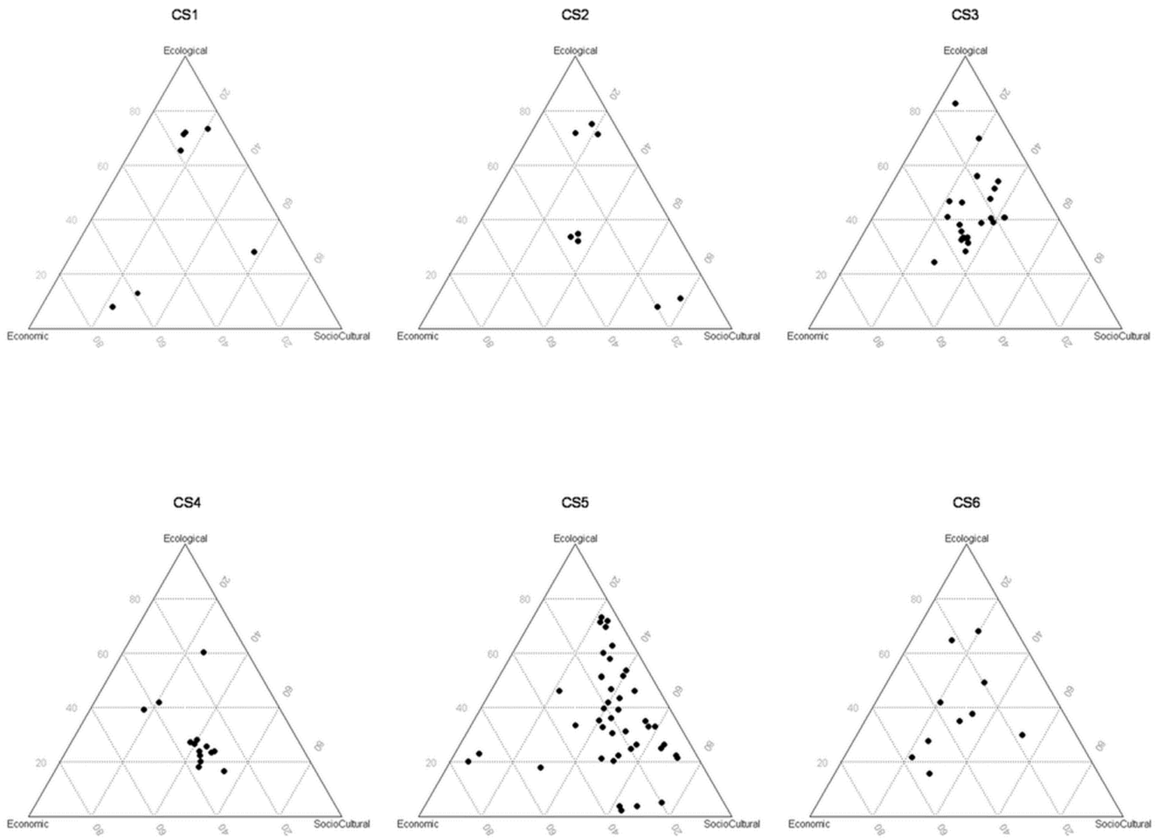


Figure 3: Ternary plots illustrating the relative stakeholders' preferences in relation to economic, ecological and socio-cultural coastal management for the COEXIST framework (n= 107 stakeholders). CS1 – Hardangerfjord (Norway), CS2 – Atlantic coast (Ireland), CS3 – Algarve coast (Portugal), CS4 – Adriatic Sea (Italy), CS5 – North Sea (Denmark, Germany and The Netherlands), and CS6 – Baltic sea (Finland).

The scrutiny of the different stakeholders across CS resulted in different perceptions of what were of most relevance when aiming for sustainability of the coast and sea. The number of respondents by CS differs and by examining the triangular grid analysis it is possible to verify that the dispersion of the results varies among the CS.

The Hardangerfjord case study (CS1) shows that stakeholders' views are dispersed and that stakeholders do not present balanced opinion (in the centre), but instead revealed outcomes tied to specific objectives that closely match their background. For four stakeholders, the summed variable contribution of ecological objectives (62% to 75%) is much more relevant than the two contributions to the economic (58% to 72%) and the one for socio-cultural (58%).

With the Atlantic coast case study (CS2) some stakeholders weighed the objectives evenly, whilst others preferred to focus on ecological or socio-cultural objectives; a similar pattern was found in three stakeholders for ecological prevalence (70% to 75%), two on social (72% to 78%) with three in the central area indicating no prevalent dimension. The Algarve coast case study (CS3) shows that most of the stakeholders allocate their preferences near the central area of the ternary plot, with some predominant preferences towards ecological objectives (up to 82%); no single stakeholder shows higher preference for either economic or socio-cultural objectives. The Adriatic coast case study (CS4) presents higher predominance near the central area, but with clear leaning towards the socio-cultural objective (from 38% to 56%); just two stakeholders show a slight predominance for ecological preference (56% to 63%) and one shows a higher preference (above 80%). The North Sea case study (CS5) shows that most preferences vary between the socio-cultural and ecological objectives; it is however important to consider that only three stakeholders allocated their preferences closest to the economic objective (from 55% to 85%). The Baltic Sea case study (CS6) shows dispersed preferences with tendencies split between the economic and ecological objectives rather than to the socio-cultural one; with just one stakeholder within the socio-cultural area (55%).

Despite several stakeholders having shown no particular preference for any of the main objectives overall, the plotted results showing the different positions reveal that the objective for which there is the highest preference is the ecological objective, followed closely by the socio-cultural and economic objectives. In terms of dissimilarities among the CS, there are three variables (the main objectives), and six CS. The respective Euclidean distances were computed as shown in Table 5.

Table 5: Dissimilarity matrix between case studies.

	Euclidean distance				
Case Study	**CS1**	**CS2**	**CS3**	**CS4**	**CS5**
CS2	16.47				
CS3	27.31	27.54			
CS4	25.49	19.54	28.47		
CS5	23.19	7.88	29.67	16.05	
CS6	13.28	19.94	18.95	18.46	23.41

From the dissimilarity matrix we can see that the Atlantic Coast of Ireland and the Coastal North Sea case studies (CS2 and 5) present the most similar patterns amongst the main objectives. In contrast, the Algarve Coast and the Coastal North Sea case studies (CS3 and 5) present the least similar patterns. These results can be explained by considering theoretical work developed by Honarkhah and Caers (2010), who stated that the smaller the Euclidean distance between two patterns, the higher is similarity between them.

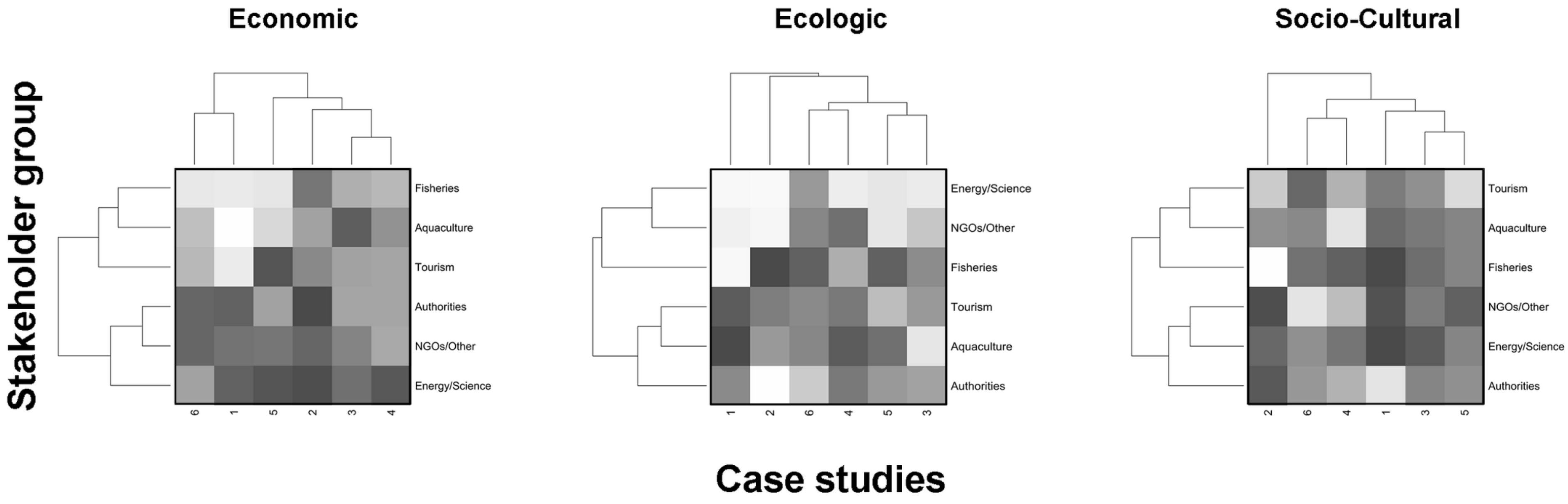


Figure 4: Heat plot screening the discriminate results applied within the scope of the COEXIST framework for each of the main objectives category (Economic, Ecological and Socio-Cultural). Each heat plot shows stakeholder sectors by case study (CS) with their respective dendrograms. CS1 – Hardangerfjord (Norway), CS2 – North Atlantic (Ireland), CS3 – Algarve Sea (Portugal), CS4 – Adriatic Sea (Italy), CS5 – North Sea (Netherlands, Germany and Denmark), and CS6 – Baltic sea (Finland).

6.6.3 Importance of objectives within CS according to stakeholder group

In order to find how important the main objectives are to respective stakeholder groups, a cluster analysis was performed. The cluster analysis results show the permutations within a set of the six CS (columns) and another set of the six stakeholder groups (rows), which are placed so that similar CS-stakeholder categories are near each other. Also, the heat map plot uses a colour scale to show where the data are distributed according to the chosen objectives. The heat map depicts the aggregate results showing the relative position all stakeholder types assume in each CS concerning the main objectives of the questionnaire (Figure 4).

When considering all the stakeholder groups, two main clusters clearly appear across all three main objectives, but when considering the CS the clusters (may) differ. By analysing the structure of the economic dendrograms and their related heatplot it is possible to verify that with respect to the economic objective, for example, the Algarve Coast and the Adriatic Sea case studies (CS3 and 4), corresponding to southern countries, present similar preferences; some similarities are also shared amongst the Hardangerfjord and Baltic case studies (CS1 and 6), i.e., corresponding to Scandinavian countries. Whereas Authorities, NGOs/Other and Energy/Science stakeholder groups do not place a higher importance on the economic objective, the remaining stakeholders have the opposite opinion. Representatives of the fishery sector tend to prioritize the economic objectives more than the operational sectors of Aquaculture and Tourism. The Energy/Science stakeholders tend to give higher importance to ecological objectives in preference to the remaining two objectives. Considering the ecological objectives, the Hardangerfjord and Atlantic Irish coast case studies (CS1 and 2) are the locations that present higher antagonist views among stakeholder groups, i.e., there are stakeholders attributing high priority to these objectives whereas others have an opposite opinion. Within the socio-cultural objectives, the Algarve Coast (CS3) and Coastal North Sea case study (CS5) show a similar pattern, followed by the Hardangerfjord case study (CS1) where only the authorities differ somewhat. The largest distance (discrepancy) was noted from the results of the stakeholders of the Irish Coast case study (CS2).

6.6.4 Preference for sub-objectives in each CS

Each of the CS developed its own sub-objectives and it was noted that common sub-objectives were derived across several CS whilst in some cases unique sub-objectives were developed (Figure 5). These sub-objectives were ranked by importance and despite the inclusion of the latter, it was still possible to ascertain the most influential items as part of the ranking process.

In general, it seems that stakeholders put significant attention on the ecological objective. Namely, they place particular emphasis on 'ensure good water quality', which was the sub-objective most often ranked in the highest positions [1st for Hardangerfjord (CS1) and Atlantic Irish coast (CS2), 2nd for Algarve (CS3) and the Coastal North Sea (CS5), 5th for the Adriatic (CS4)]. 'Preserve target stocks (GES)' was ranked the second most relevant sub-objective [1st for the Coastal North Sea (CS5), 2nd for Adriatic (CS4) and Baltic (CS6), 9th for Algarve (CS3), and 12th for Hardangerfjord (CS1)]. Other highly relevant items are 'provision of employment for coastal communities' [ranked 2nd for Irish coast (CS2), 4th for Adriatic (CS4), 5th for Algarve coast (CS3), 7th for Baltic (CS6), 10th for Hardangerfjord (CS1) and Coastal North Sea (CS5)]; and 'ensure high resource rent' [ranked 1st for Algarve coast (CS3), and 3rd for Coastal North Sea (CS5)].

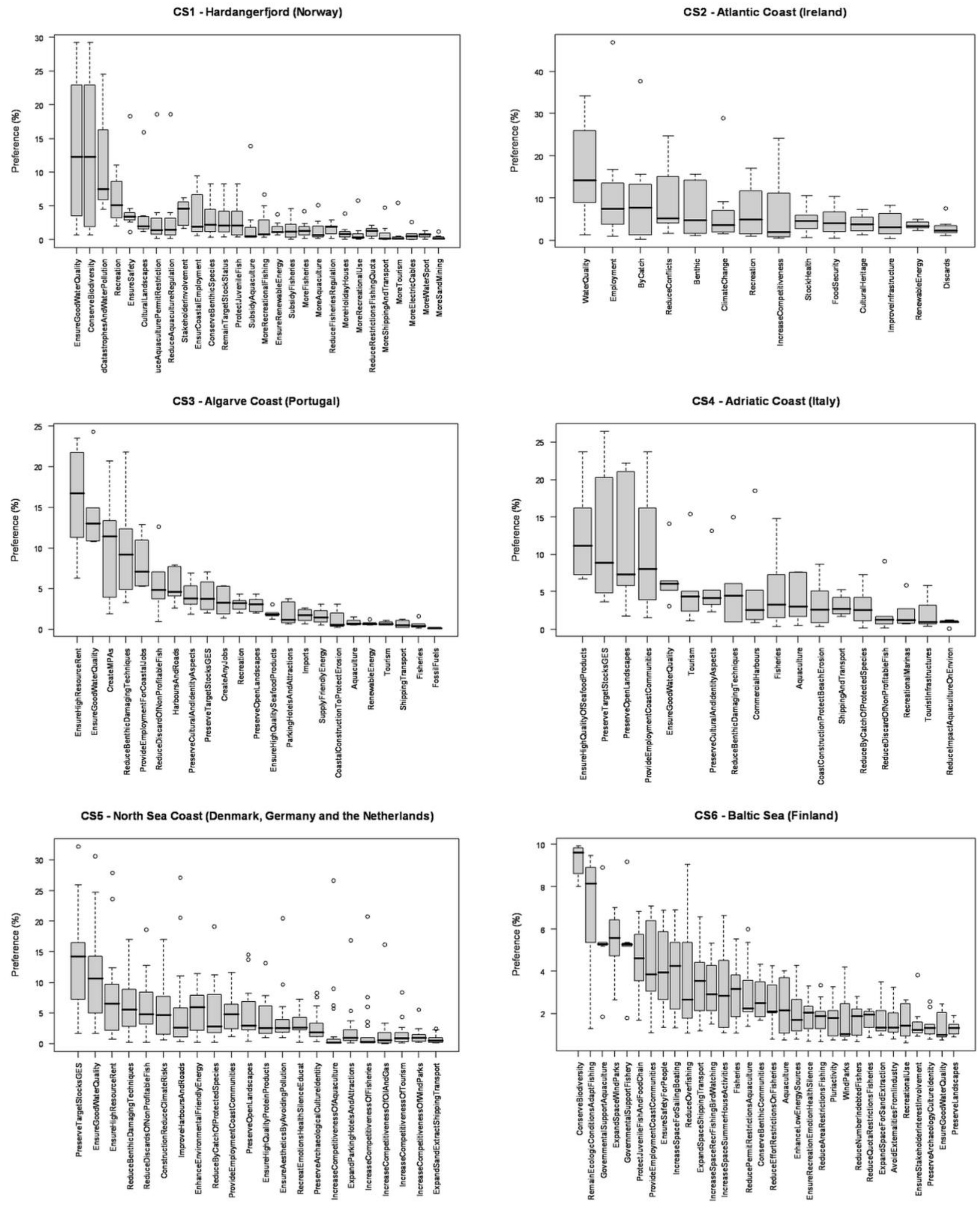


Figure 5 – Box plot showing the ranking of sub-objective preferences for each case study (CS) of the COEXIST framework. A) CS1 – Hardangerfjord (Norway), 28 sub-objectives; B) CS2 – North Atlantic Coast (Ireland), 14 sub-objectives; C) CS3 – Algarve Coast (Portugal), 23 sub-objectives; D) CS4 – Adriatic Sea Coast (Italy), 18 sub-objectives; E) CS5 – North Sea Coast (Netherlands, Germany and Denmark), 22 sub-objectives; and F) CS6 – Baltic Sea (Finland), 32 sub-objectives.

6.7 Discussion

Marine spatial planning is a complex process that involves the interaction between values and interests of many different stakeholders. Proposals from a wide range of economic and technological sectors that are being evaluated by planning authorities (coastal managers) are receiving heightened interest from society as evidenced by the increased level of debate and the close scrutiny that is being paid to every proposal put forward for planning approval. Different projects that are carried out in the coastal area can have various levels of impact on the different stakeholders. As a consequence, therefore, before, during and after the delivery of any such projects, stakeholders may have, or feel, dissimilar degrees of benefit or detriments of the proposed scheme. Similarly, stakeholders may have their own degree of influence on the development of such projects. For instance, Macharis (2007) mentions that the evaluation of the relative importance of stakeholder groups, either in terms of impact or influence, is important in order to understand the value of projects to society as a whole.

Brown *et al.* (2002) point out that there is an increasing need among stakeholders of the coastal areas and the society in general, to get more information about the risks of human activities that coexist but are conflicting. The coastal area is a common ground for an enormous range of activities, and therefore it is crucial to find the best consensual decision. This explains why decisions concerning future developments in coastal areas are so heavily debated.

Planning processes have been developed in the last decades to address the need for increasing resource sustainability and to find trade-offs between human use and natural resources. In Europe MSP is suggested as being beneficial under the *Marine Strategy Framework Directive* (MSFD), but there is no driver (MSP Directive) as yet. Currently, drivers are under European and international legal frameworks (e.g. Water Framework Directive and Natura 2000).

Reed *et al.* (2009) highlight that the interaction between several stakeholders from distinctive institutions is a reality in disputed arenas and that a plethora of dissimilar interests is possible. Monitoring expectations from stakeholders of different groups and origins through an enquiry method, namely by using the COEXIST framework, is a feasible way to collect information on the subject under analysis. This empirically based approach assumes that stakeholders judge the subjects in the analysis against their own interests and evaluate them according to their needs (Ramos *et al.*, 2011).

The MCA approach detailed here has the advantage that the stakeholders involved come from a wide range of activity sectors CS and across a wide geographical spread (i.e., the scope of the COEXIST project). As Schwilch *et al.* (2012) highlight, it is important that the results of an approach like the one presented here are utilised by policy makers before they make final decisions, as this should enhance social acceptance due to the greater transparency and inclusiveness. Given the increasing competition for space in coastal areas it is also important to identify methods to support the implementation of MSP in order to reduce potential conflicts and increase prospective synergies. Despite differences in the geographic locations and contexts of the CS, it is possible to find similarities among their stakeholders by applying the MCA approach described. The authors believe that by using this approach it is possible to develop a common 'language' and make reliable comparisons. The ultimate goal in all the CS was to achieve a 'sustainable coastal / marine ecosystem'. However, the term 'sustainable' is open to debate. For that reason, it is important to perceive qualitatively the range and type of conflicts between activities and stakeholders that exist in each CS. This can be utilised to determine potential future conflicts between economic, biological, and socio-cultural activities and pro-actively debate methods of avoiding this conflict and address the 'sustainability' problem. One of first steps of the MCA approach presented is to collect the view of stakeholders with different perspectives, and subsequently identify the most relevant options to consider when working towards the fulfilment of a defined main goal.

In our approach the aggregated results (ternary plot) show that several of the stakeholders prefer a balance between all three main objectives. However, a large number favoured the ecological objective (and up to a certain extent the socio-cultural), in preference to the economic objective.

The heat plots and dendrogram results show that the three stakeholder groups that are more closely related to the production sector or industry (i.e., fisheries, aquaculture and tourism) tend to give higher importance to economic objectives. In addition, they form a specific larger cluster, whereas the remaining stakeholder groups form a discrete one. A similar pattern is found for the socio-cultural objectives. A cluster swap did emerge between two stakeholder groups (i.e., fisheries and authorities), but only for the ecological objectives.

The heat plot and dendrogram results also illustrate that between CS, despite their different areas, contextualization of activities, and latitudinal distances, some similarities on the ecological and socio-cultural objectives can be found for the pairs: Algarve coast (CS3) – North Sea coast (CS5) and Adriatic coast (CS4) – Baltic Sea (CS6). All the remaining comparisons were dissimilar.

Despite getting an insight into stakeholders and how their background can influence their decisions, this initial treatment only deals in generalities. It does provide some material for decision makers in terms of the categories of stakeholders (i.e., by main objectives, country, and so forth), but it does not pinpoint the most sensitive and controversial issues with respect to planning in the coastal zone. For this reason the ranking of the sub-objectives is a highly important step of the MCA approach as it demonstrates some comparable preferences between stakeholders, as well as highlighting which issues are important or not crucial in each CS.

Although we found several similarities among the coastal areas examined, the complexity and dissimilarity increased when we considered more specific objectives. The high diversity found in each CS, particularly for those sub-objectives attaining higher preference values, shows the variability across the stakeholder groups. In particular, the Coastal North Sea case study (CS5) is the one where there were more outliers, highlighting the difficulty of reaching consensual preferences. Although the number of stakeholders and dispersion of results differ among the CS, several other reasons could explain the occurrence of the diversity in the results.

There is no doubt that the scale of the different activities, as well as the intervention of the different stakeholders, varies across the CS. Traditional economic activities such as fisheries may have to compete with more recent activities such as renewable energies or nature conservation for space and for social acceptance. For instance, a new beneficial development of offshore windfarms may result in the loss of fishing grounds, at least for particular fishing segments (Berkenhagen *et al.*, 2010). Other amenities that society desires and values might not be expressed in economic terms, but more in ecological or cultural aspects. These may have greater acceptance by stakeholders involved in the process of prioritizing aspects for sustainable seas.

Broadly speaking, as found here, it is understandable that different stakeholder groups may have somewhat different positions when considering any given change (Ramos *et al.*, 2007). However, whilst the view of stakeholders amongst peers may be similar across the different CS, their weighting differs when comparing contexts and societies. However, it was not the intention of this paper to rank the stakeholders, not only because this is extremely difficult but also because it is highly controversial.

6.8 Final Remarks

An assessment of the economic, social and environmental dimensions and their more detailed aspects appears to be crucial, for any planning process because this encourages more transparency, accountability and legitimacy in the decision making processes. The stakeholder based MCA approach introduced in this article can be used to analyse the whole range of human activities and interests found in the coastal areas. In each of the six CS, there is a unique consideration of the marine environment, local activities and/or the needs of stakeholders. The identification of the main local activities and their operational demands in spatial, temporal or institutional terms is of fundamental importance in understanding the different sectoral interests and determining an approach to improve mutual understanding. The proven application of the stakeholder based MCA approach to real world situations can help by facilitating debate between sectors so as that they can (mutually) understand their competitors thought processes and why they have certain preferences for any given location.

This particular study observed that despite an overall preference towards ecological preservation – there is strong support for economic growth from the operational sectors, regardless of where they are located. Therefore, the question remains on how to complement the draft MSP Directive, as a tool to promote sustainable growth, given these diametrically opposed views. Thus future ecosystem based management processes such as MSP, must seek the integration of multiple objectives and their associated management measures.

Finally, stakeholders have indicated the significant importance they attribute to being consulted regarding decisions at the European scale (COEXIST 2012), and increased legitimacy could be obtained by using the stakeholder based MCA approach as introduced in this study.

6.9 Acknowledgements

We would like to thank David Sampson, Sue Kidd and two anonymous reviewers for their helpful revision and comments on early versions of the manuscript. The research leading to these results has received funding from the European Community's Seventh Framework Programme (FP7/2007-2013) under grant agreement no 245178. This publication reflects the views only of the authors, and the European Union cannot be held responsible for any use which may be made of the information contained therein. The authors would like to thank all those people who answered the questionnaire across the different countries involved, representing different institutions and sectors of activity named in the present paper as stakeholders. We are also grateful for the feedback given by all those who participated in the joint ICES/COEXIST Workshop: 'Best Practice Guidelines for spatial planning to integrate fisheries, aquaculture and other uses in the coastal zone' held in September 19th, 2012 in Bergen (Norway). All partners in COEXIST are thanked for their input in many discussions and also many other contributions.

6.10 References

Bergh, Ø., Børsheim, K.Y., Vik Ottesen, M., Soma, K., and Gomez, E.B. 2012. CoExist – sameksistens i kystsonen. *In* Havforskningsrapporten 2012. Ressurser, miljø og akvakultur på kysten og i havet. Institute of Marine Research (IMR). Fisken og havet 1–2012. Bergen. pp. 25–27. (In Norwegian).

Berkenhagen, J., Döring, R., Fock, H. O., Kloppmann, M. H., Pedersen, S. A., and Schulze, T. 2010. Decision bias in marine spatial planning of offshore wind farms: Problems of singular versus cumulative assessments of economic impacts on fisheries. Marine Policy, 34(3): 733–736.

Berkes, F. 2009. Evolution of co-management: role of knowledge generation, bridging organization sand social learning. Journal of Environmental Management 90: 1692–1702.

Brown, K., Tompkins E. L., and Adger W. N. 2002. Making waves: integrating coastal conservation and development. Earthscan, London, UK.

Christie, P. 2011. Creating space for interdisciplinary marine and coastal research: Five dilemmas and suggested resolutions. Environmental Conservation, 38: 172–186.

Coetzer, R. L. J., Rossouw, R. F., and Le Roux, N. J. 2012. Efficient maximin distance designs for experiments in mixtures. Journal of Applied Statistics, 39: 1939–1951.

COEXIST 2011. Website of EU FP7 research project CoExist. "http://www.coexistproject.eu/"http://www.coexistproject.eu/

COEXIST 2012. Report: Joint ICES/COEXIST workshop: "Best Practice Guidelines for spatial planning to integrate fisheries, aquaculture and other uses in the coastal zone". 73pp. "http://www.coexistproject.eu/index.php/ices-coexist-workshop/"http://www.coexistproject.eu/index.php/ices-coexist-workshop/

Curtin, D., and Meijer, A. J. 2006. Does transparency strengthen legitimacy? Information Polity, 11: 109–122.

Dempster, T., Sanchez-Jerez, P. 2008. Aquaculture and coastal space management in Europe: an ecological perspective. In: Holmer, M., Black, K., Duarte, C.M., Marbà, N., Karakassis, I. (Eds.), Aquaculture in the Ecosystem. Springer Verlag. ISBN: 1402068093, pp. 87–116.

Deng, H. 1999. Multicriteria analysis with fuzzy pairwise comparison. In Fuzzy Systems Conference Proceedings, 1999. FUZZ-IEEE'99. 1999 IEEE International 2: 726–731).

Douvere, F. 2008. The importance of marine spatial planning in advancing ecosystem-based sea use management. Marine Policy, 32: 762–771.

Douvere, F., and Ehler, C. N. 2009. New perspectives on sea use management: Initial findings from European experience with marine spatial planning. Journal of Environmental Management, 90: 77–88.

Ehlers, P. and Lagoni, R. 2006. International Maritime Organisations and their Contribution towards a Sustainable Marine Development. LIT Verlag Hamburg.

Foley, M. M., Halpern, B. S., Micheli, F., Armsby, M. H., Caldwell, M. R., Crain, C. M., Prahler, E., *et al*. 2010. Guiding ecological principles for marine spatial planning. Marine Policy 34: 955–966.

Graffelman, J., and Camarena, J. M. 2007. Graphical tests for Hardy-Weinberg equilibrium based on the ternary plot. Human Heredity, 65: 77–84.

Hajkowicz, S., and Collins, K. 2007. A review of multiple criteria analysis for water resource planning and management. Water Resources Management, 21: 1553–1566.

Hajkowicz, S., and Higgins, A. 2008. A comparison of multiple criteria analysis techniques for water resource management. European Journal of Operational Research, 184: 255–265.

Hedelin, B. 2007. Criteria for the assessment of sustainable water management. Environmental Management, 39: 151–163.

Hentschel, M. L., and Page, N. W. 2003. Selection of descriptors for particle shape characterization. Particle and Particle Systems Characterization, 20: 25–38.

Honarkhah, M., and Caers, J. 2010. Stochastic simulation of patterns using distance-based pattern modeling. Mathematical Geosciences, 42: 487–517.

Janßen, H., Kidd, S., and Kvinge, T. 2013. A spatial typology for the sea: A contribution from the Baltic. Marine Policy, 42: 190–197.

Kickert, W. J. 1978. Fuzzy theories on decision making: A critical review. Kluwer Academic Publishers, Boston.

Kidd, S. and Ellis, G. 2012. From the land to sea and back again? Using terrestrial planning to understand the process of marine spatial planning. Journal of Environmental Policy and Planning, 14: 49–66.

Kidd, S. and Shaw, D. 2013. Reconceptualising Territoriality and Spatial Planning: Insights from the Sea. Planning Theory and Practice, 14: 180–197.

Klein, C. J., Steinback, C., Watts, M., Scholz, A. J., and Possingham, H. P. 2009. Spatial marine zoning for fisheries and conservation. Frontiers in Ecology and the Environment, 8: 349–353.

Koleff, P., Gaston, K. J., and Lennon, J. J. 2003. Measuring beta diversity for presence–absence data. Journal of Animal Ecology, 72: 367–382.

Lahdelma, R., Salminen, P., and Hokkanen, J. 2000. Using multicriteria methods in environmental planning and management. Environmental Management, 26: 595–605.

Linkov, I., Satterstrom, F. K., Kiker, G., Batchelor, C., Bridges, T., and Ferguson, E. 2006. From comparative risk assessment to multi-criteria decision analysis and adaptive management: Recent developments and applications. Environment International, 32: 1072–1093.

Macharis, C. 2007. Multi-criteria analysis as a tool to include stakeholders in project evaluation: the MAMCA method. *In* Transport Project Evaluation. Extending the Social Cost–Benefit Approach, pp.115–131. Ed. by E. Haezendonck. Edward Elgar, Cheltenham.

Macharis, C., Springael, J., De Brucker, K., and Verbeke, A. 2004. PROMETHEE and AHP: The design of operational synergies in multicriteria analysis.: Strengthening PROMETHEE with ideas of AHP. European Journal of Operational Research, 153: 307–317.

Mendoza, G. A., and Martins, H. 2006. Multi-criteria decision analysis in natural resource management: a critical review of methods and new modelling paradigms. Forest Ecology and Management, 230: 1–22.

Mendoza, G. A., and Prabhu, R. 2005. Combining participatory modeling and multi-criteria analysis for community-based forest management. Forest Ecology and Management, 207: 145–156.

Molle, F. 2009. River basin planning and management: The social life of a concept. Geoforum 40: 484–494.

Munda, G. 2004. Social multi-criteria evaluation: Methodological foundations and operational consequences. European Journal of Operational Research, 158: 662–677.

Nijkamp, P. 1975. A multicriteria analysis for project evaluation: Economic Ecological Evaluation of a Land Reclamation Project. Papers in Regional Science, 35: 87–111.

O'Riordan, T., Watkinson, A. and Milligan, J. 2005. Living with a changing coastline: exploring new forms of governance for sustainable coastal futures, Tyndall Centre Technical Report, School of Environmental Sciences, University of East Anglia, Norwich.

Ostrom, E., Burger, J., Field, C. B., Norgaard, R. B., and Policansky, D. 1999. Sustainability-Revisiting the commons: Local lessons, global challenges. Science 284: 278–282.

Palmer, M. C., Wigley, S. E., Hoey, J. J., and Palmer, J. E. 2007. An evaluation of the Northeast Region’s study fleet pilot program and electronic logbook system: Phases I and II. NOAA Technical Memorandum NMFS-NE, 204, 1–22.

Peel, D., and Lloyd, M. G. 2004. The social reconstruction of the marine environment: Towards marine spatial planning?. Town Planning Review, 75: 359–378.

Pinho, L. 2007. The role of maritime public domain in the Portuguese Coastal management. Journal of Coastal Conservation, 11: 3–12.

Pinkse, J., Dommisse, M. 2008. Overcoming barriers to sustainability: an explanation of residential builders' reluctance to adopt clean technologies. Business Strategy and the Environment,18: 515–527.

R Development Core Team 2011. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. ISBN 3-900051-07-0, [online] URL: http://www.R-project.org.

Ramos, J., Santos, M. N., Whitmarsh, D., and Monteiro, C. C. 2007. Stakeholder perceptions regarding the environmental and socio-economic impacts of the Algarve artificial reefs. Hydrobiologia, 580: 181–191.

Ramos, J., Santos, M. N., Whitmarsh, D., and Monteiro, C. C. 2011. Stakeholder analysis in the Portuguese artificial reef context: winners and losers. Brazilian Journal of Oceanography, 59(SPE1): 133–143.

Reed, M. S., Graves, A., Dandy, N., Posthumus, H., Hubacek, K., Morris, J., Prell, C., *et al.* 2009. Who's in and why? A typology of stakeholder analysis methods for natural resource management. Journal of Environmental Management, 90: 1933–1949.

Ruiz-Frau, A., Edwards-Jones, G., and Kaiser, M. J. 2011. Mapping stakeholder values for coastal zone management. Marine Ecology Progress Series, 434: 239–249.

Saaty, T. L., and Vargas, L. G. 2013. Criteria for Evaluating Group Decision-Making Methods. *In* Decision Making with the Analytic Network Process, pp. 295–318. Springer, US.

Seefeld, K., and Linder, E. 2007. Statistics Using R with Biological Examples. University of New Hampshire.

Schwilch, G., Bachmann, F. and de Graaff, J. 2012. Decision support for selecting SLM technologies with stakeholders. Applied Geography, 34: 86–98.

Shakhnov, I. F. 2008. A problem of ranking interval objects in a multicriteria analysis of complex systems. Journal of Computer and Systems Sciences International, 47: 33–39.

Smith, H. D., Maes, F., Stojanovic, T. A., and Ballinger, R. C. 2011. The integration of land and marine spatial planning. Journal of Coastal Conservation, 15: 291–303.

Soma, K. 2003. How to involve stakeholders in fisheries management – A country case study in Trinidad and Tobago. Marine Policy, 27: 47–58.

Soma, K. 2010. Framing participation with multicriterion evaluations to support the management of complex environmental issues. Environmental Policy and Governance, 20: 89–106.

Soma, K., Ramos, J., Bergh, Ø., Schulze, T., van Oostenbrugge, H., van Duijn, A. P., Kopke, K., *et al.* 2013. The "mapping out" approach: effectiveness of marine spatial management options in European coastal waters. ICES Journal of Marine Science, doi:10.1093/icesjms/fst193.

Sparrevik, M., Barton, D. N., Oen, A. M., Sehkar, N. U., and Linkov, I. 2011. Use of multicriteria involvement processes to enhance transparency and stakeholder participation at Bergen Harbor, Norway. Integrated Environmental Assessment and Management, 7: 414–425.

Stelzenmüller, V., Lee, J., South, A., Foden, J., and Rogers, S. I. 2013. Practical tools to support marine spatial planning: A review and some prototype tools. Marine Policy, 38: 214–227.

Stelzenmüller, V., Schulze, T., Fock, H. O., and Berkenhagen, J. 2011. Integrated modelling tools to support risk-based decision-making in marine spatial management. Marine Ecology Progress Series, 441: 197–212.

Van Delft, A. 1977. Multi-criteria analysis and regional decision-making (Vol. 8). Springer.

Yan, J., Dagang, T., and Yue, P. 2007. Ranking environmental projects model based on multicriteria decision-making and the weight sensitivity analysis. Journal of Systems Engineering and Electronics, 18: 534–539.

Zafonte, M., and Sabatier, P. 2004. Short Term Versus Long Term Coalitions in the Policy Process: Automotive Pollution Control, 1963–1989. Policy Studies Journal, 32: 75–107.

7. General discussion

In the course of Integrated Maritime Policy (IMP) and the Europe 2020 strategy tall orders are placed with the European countries (EC, 2012). The member states are faced with multiple objectives such as Good Environmental Status (GES) or Blue Growth (EC, 2014b). Further, an optimization of fisheries and aquaculture contributions to food security was raised (EC, 2015). Consequently, Marine Spatial Planning (MSP) was identified as the cross-cutting policy tool applying an ecosystem-based approach to the management of human activities (EC, 2014b). Accordingly, MSP shall not only implement multi-sector planning but also multi-objective planning. In contrast to the Marine Strategy Framework Directive (MSFD), which sets out indicators describing what the environment will look like when GES has been achieved (EC, 2008b), no indicators or targets are given for MSP (EC, 2014a). In order to meet multi-objective requirements, optimal solutions need to be based on trade-offs. Consequently, guidance towards spatial management should reflect the efficiency of management strategies, to get the best out of it. Accordingly, scientific underpinning is needed to identify the most efficient management strategies towards Ecosystem-Based Marine Spatial Planning (EB-MSP).

The overall aim of this thesis was to develop and test concrete, place-based tools which allow a transparent evaluation of spatial management options in the ‘German Bight’. Eventually tools useful to identify the most efficient management strategies towards EB-MSP as well as the risks and returns coming from alternative management objectives were identified. Hereby the attention is directed to the fishery sector, demersal fish populations and the benthic ecosystem in particular. Subsequently, recommendations towards EB-MSP balancing sustainable use and ecosystem health are given.

7.1 General case study results: towards an EB-MSP approach to the German EEZ of the North Sea

As mentioned above, MSP is identified as the cross-cutting policy tool that promotes multiple objectives. Usually, aiming to test a set of operational objectives and indicators for their performance towards EU legislations, MSP procedures follow a cyclic evaluation process (section 1.1). Such a process is e.g. described by Stelzenmüller et al. (2013a) or Katsanevakis et al. (2011) and shown in Figure 1. It explains in a logical way how MSP measures can be established, so that they can be evaluated.

In line with the Europe 2020 strategy, joint efforts are recently undertaken towards a general principle for spatial planning in the federal territory of Germany. The overarching goal is to concretize and prioritize collective objectives and management strategies (section 1.2). Hereby a multidisciplinary approach is needed by nature. Tools have to provide holistic views on the system to understand trade-offs in response to management measures (Katsanevakis et al., 2011). This thesis considers spatially explicit tools as a support of the practical implementation of EB-MSP in the 'German Bight'. The studies performed provide concrete tools supporting the evaluation of management effectiveness, linked to the tasks EB-MSP has to accomplish.

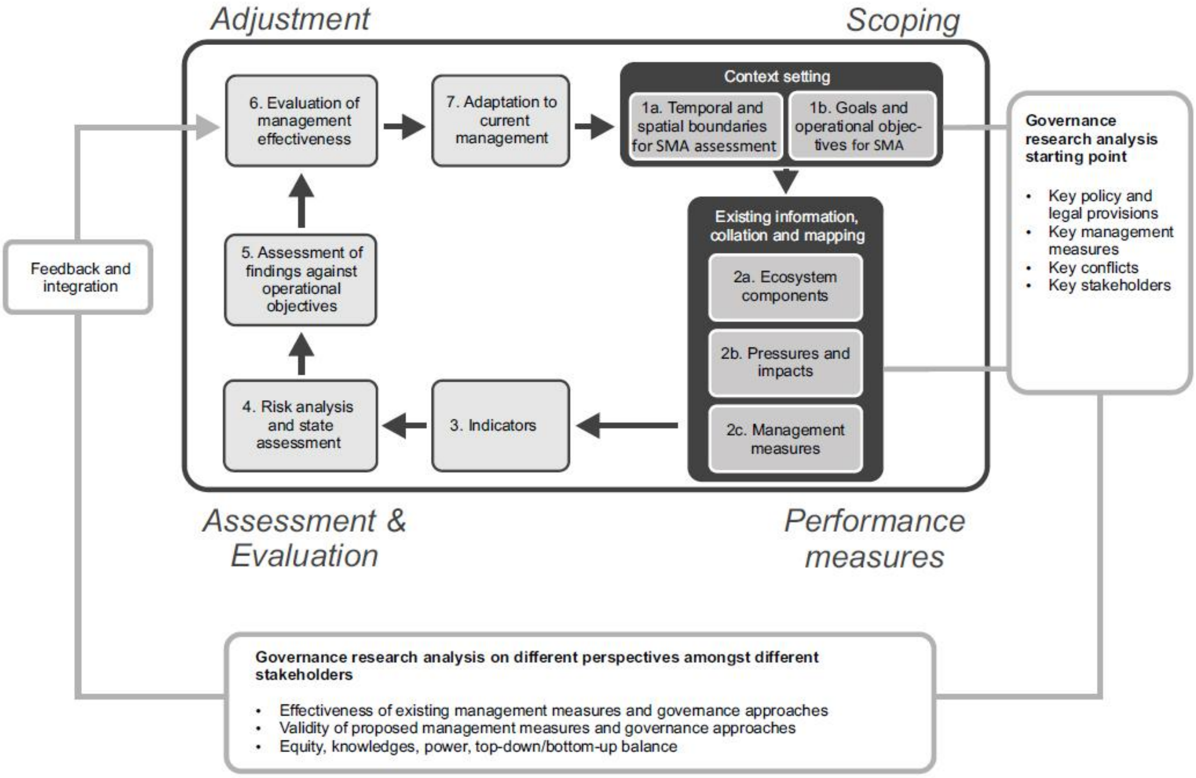


Figure 1: The flowchart shows a proposed framework to monitor and evaluate spatially managed areas (SMAs) through seven key steps; step1: definition of the spatial and temporal boundaries (step 1a). In relation to those boundaries the high level goals and operational objectives are delineated for the respective run through the process (step 1b); step2: identification, collation and mapping of existing information; step 2a: mapping of ecosystem components relevant to the set of objectives; step 2b: mapping of pressures and impacts; step 2c: summary of existing or proposed management measures; step 3: definition of performance measures or indicators together with their reference points will be defined; step 4: risk analysis or state assessment; step 5: summary of assessment results against operational objectives; step 6: evaluation of management effectiveness; step 7: summary of assessment results and formulation of recommendations (e.g., alternative management scenarios). Further, proposed framework steps and the links to the governance research elements are shown. Taken from Stelzenmüller et al. (2013a).

Aiming to give a first overview about the tool performances, a Strengths Weaknesses Opportunity Threat (SWOT) -Analysis is shown in Table 1. Further explanations are given in the subchapters below with reference to the cyclic evaluation process shown in Figure 1 and the manuscripts (MS) from the previous chapters. The findings demonstrate the advantages and limitations of the tools selected. In a nutshell, they provide aid for thinking about complex systems and synthesizing knowledge, as consequences of management actions can be too diverse for human brains to get the whole picture. Further, they facilitate the communication of scientific results to decision makers. The information about the (spatial) extent of management effects is a fundamental requirement for taking decisions. The risks and returns coming from alternative management objectives assessed for the German Bight are summarised in Table 2 with further explanations given in the subsequent chapters. Hereby the attention is directed to the fishery sector, demersal fish populations and the benthic ecosystem in particular. In summary, results suggest that current and future spatial management strategies have a considerable effect on the fishery sector and the resources it is depending on (Tab. 2).

Table 1: SWOT-Analysis of overall place-based methods as applied during dissertation process. The strength and opportunities are defined as being helpful, the weaknesses and threats as being harmful to achieving the objectives of a study. Offshore Wind Farms are abbreviated (OWF).

	Strength	**Weakness**	**Opportunity**	**Threats**
Conflict analysis (MS 1)	First overview about conflicting uses, conflicts can be spatially assigned to overlapping activities, highlighting of potential conflicting uses, cross-sector or multi-sector approach	No differentiation of closed areas (i.e. nature conservation sites) in regard to activity combination of low conflict potential (e.g. OWFs), no temporal dimension (duration and intervals of human activities), no consideration of coastal areas (e.g. tourism)	Integrate temporal dimension (duration of activities), differentiations of areas closed for fisheries (permission of pelagic fisheries in OWFs), include coastal sectors	Management decisions biased by imprecise descriptions of conflict
Synergy analysis (MS 2)	Synthesize multiple factors determining the suitability of a site, uncertainty assessment related to model prediction, simple overlay mapping that facilitates the designation of co-use sites, suitability-scale that facilitates the final choice of co-location	No uncertainty assessment in model parameterisation and factor weighting (expert opinion), no environmental impact assessment	Analysis of IMTA profitability, environmental carrying capacity assessment, environmental impact assessment, analysis of economic viability of co-locations, integrated assessment process of the German MSP concerning measures to grant facilities for volunteering co-location developers	Requires a coherent knowledge of environmental system function and processes (e.g. waves, currents)
Qualitative risk assessment (MS 1)	Enables the mapping of species specific habitats, illustrates cause-effect pathways of human impact, facilitates the integration of multiple data sets (pressures), eases the mapping of risks (pressures and impacts)	No consideration of land-based pressures (e.g. fertilizers), no temporal dimension and pressure intensity addressed (drivers and pressures considered to be temporarily and spatially constant for a time period of one year), lacking environmental footprint of OWF development (e.g. noise), no representation of food web dynamics or natural disturbance, no consideration of cumulative or positive effects	Inclusion of cumulative or positive effects of the drivers, consider coastal areas and further sectors (tourism, land based influences), analyse pressure intensity, include food chain effects and natural disturbance	Requires coherent knowledge about the factors the risk is composed of (e.g. vulnerability, sensitivity), focus is on one species at a time
Quantitative risk assessment (MS 3)	Percentage change of occurrence probabilities of different states of a pressure (e.g. benthic disturbance) and change of the state (scenario simulation),	No detailed measures of recovery, mortality or benthic disturbance for the benthic communities (i.e. infaunal and epifaunal recovery), no integration of natural	Redefine fleet specific impact resp. recovery rates (combination of infaunal and epifaunal recovery), integrate more realistic fishing effort	Model construction challenging and nontrivial, require a detailed, likewise coherent knowledge of system

	Strength	Weakness	Opportunity	Threats
	transparent and flexible parameterisation, represents multi-dimensional distributions of drivers (pressure), integrates uncertainty assessment in impact prediction and cause-effect pathways	disturbance (e.g. waves)	displacement and fleet behaviour scenarios to predict likely local values of disturbance, include interactions between fishing and natural disturbance	function and processes (e.g. recovery rates), focus is on one pressure at a time, observed recovery rates might incorporate indirectly local effects of natural disturbance
Trade-off assessment (MS 4)	Facilitates the integration of various data (e.g. activities, habitats), performance measures of ecosystem services or indicators, trade-offs facilitate the formulation of recommendations (e.g., alternative management scenarios) by identifying the best possible level of return for its given level of risk	No illustration of uncertainties (input data, model prediction), no holistic assessment (only a choice of human activities and habitats integrated), the response of the ecosystem is assumed to be static, an adaption of both, the ecosystem and the stakeholders is poorly addressed, no consideration of positive effects, no measures of recovery	Inclusion of cumulative or positive effects of the drivers affecting ecosystem services, include an adaption of the fishery sector to spatial closures, integrate uncertainty assessments or expert knowledge, integrate temporal dimensions (spawning and nursery seasons)	Ignoring uncertainties can modify decision making
Stakeholder preference analysis (MS 5)	Condensation of stakeholder interests, supports the identification of the most relevant management option, justification of decision making, participatory, empirical, integrative and future-oriented elements, development of a common language throughout the process (i.e. wording), visualise interactions (e.g. future conflicts)	Low comparability of stakeholders surveyed (wide range of activity sectors and geographical spread), low degree of respondents	Ensure a higher number of respondents when surveying the stakeholders preferences to facilitate the task of finding similarities and consensus	Stakeholders preferences could be biased based on conflicts

7.1.1 Place-based tools to support a practical implementation of EB-MSP

If the spatial boundaries of an area such as the German Bight are defined, then as a first step supportive tools need to identify, collate and map existing information. The application of a Geographic Information System (GIS) facilitated the visualization of current and future conditions. Limited available marine space may lead to spatial and temporal conflicts. Knowledge about the extent of e.g. risks and conflicts on a spatial and temporal scale will increase the understanding of activities' characters and their interaction (Parravicini et al., 2012; Kovács et al., 2014). The conflict analysis applied in MS 1 gave a first overview on conflicting uses (Stelzenmüller et al., 2013b; Cormier et al., 2010). Combined with a GIS, conflict maps could be generated by spatially assigning conflicts to overlapping activities. Applied pro-actively, this approach promotes the adaptive element of MSP in highlighting potential conflicting uses. Further, if all economic sectors in the region are included, the integrative part EB-MSP should contribute to is also covered (Katsanevakis et al., 2011).

In the light of lacking space for sustainable development, integrative and adaptive planning processes will furthermore focus on synergistic effects such as co-locations. Given that the spatial extent of both synergistic activities is known, the mapping of simple overlays allows the designation of suitable sites (Stelzenmüller et al., 2013b; Stelzenmüller et al., in preparation). If the spatial context of a new activity such as offshore aquaculture (MS 2) is not known, a Multi-Criteria Evaluation (MCE) technique is advantageous in synthesizing multiple factors determining the suitability of a site. Integrating an Ordered Weighted Average (OWA) process, uncertainties can be addressed (Gorsevski et al., 2012; Boroushaki and Malczewski, 2008; Gemitzi et al., 2007). This was demonstrated in MS 2, where different weighting combinations were compared to weightings made by experts. Further, the integration of a suitability-scale facilitated the choice of co-location sites. Nevertheless, these kinds of analyses require a coherent knowledge of system function and processes such as wave height.

As shown in the case of aquaculture, different factors described the suitability of a site for each species tested.

To gain a tendentious overview of the recent state of a marine ecosystem (or rather its components), Environmental Risk Assessments (ERA) can be performed (Cormier et al., 2010). Conducting a qualitative risk assessment of management strategies, an EB-MSP can be promoted by integrating both ecosystem components and human activities. The mapping of relevant ecosystem components requires coherent knowledge about the factors it is composed of (Zucchetta et al., 2010; Freitas et al., 2010). As shown in MS 1, plaice juveniles prefer shallow waters. In contrast, plaice adults move to deeper waters. Here, Species Distribution Models (SDM) were useful in describing the relationship of the distribution of an organism to its environment, enabling the mapping of species and stage specific habitats. The conceptual Driver Pressure State Impact (DPSI) model illustrated cause-effect pathways (Cormier et al., 2010). It showed the links between drivers of human activities to predict where their pressures have the potential to cause effects that could change the integrity of ecosystem components such as habitats (Elliott, 2002; Cormier et al., 2010). It further facilitated the integration of multiple data sets and therefore of multiple drivers by using a single metric. This strength arose from the definition of general pressures such as e.g. abrasion, which is exerted by sediment extraction and mobile bottom contact gear fisheries (MS 1). The added value of the habitat management qualitative risk assessment applied in MS 1 becomes apparent when the exposure of anthropogenic effects is characterised and estimated. It also provides a basis for assessing sensitivity, rarity and resiliency of essential habitats. When considering regional effects of localised human activities, it reflects the risk potential of present regional management actions altering the state of the ecosystem (van Deurs et al., 2012). This was exemplified in MS 1, where the risk of OWF development on plaice nursery grounds was estimated to be low.

Coupling the Bayesian Belief Network (BBN) with GIS, the occurrence probabilities of different states of a pressure (e.g. benthic disturbance) could be predicted in a spatially resolved way (MS 3). The change of the state due to simulated spatial management objectives could be expressed in percentage of the area affected. Regarding its application, the added value was evident in the transparency of tool parameters. Therefore, updates could be handled flexible, providing an adaptive element successful MSP is depending on. BBNs indeed are advantageous, especially when considering the input from various data types, but model construction often is challenging and nontrivial due to uncertainties which need to be quantified. Admittedly, while tendentious effects can already be identified in qualitative analyses, the likelihood of a certain pressure needs to be assessed using quantitative methods. Albeit probabilistic methods are restricted to focus on one pressure at a time, this one is explicitly assessed (Kjræulff and Madsen, 2012; Stelzenmüller et al., 2010a), which enables to summarise assessment results against multiple operational objectives. In addition, uncertainty in impact prediction and cause-effect pathways can be visualised on a common spatial scale (MS 3). These kinds of analyses require a detailed, likewise coherent knowledge of system function and processes as well as all the drivers of the pressure examined (Stelzenmüller et al., 2011; Johnson et al., 2012; Grêt-Regamey et al., 2013a; Grêt-Regamey et al., 2013b).

In order to identify the most efficient management strategy, multiple management strategies need to be assessed. The concept of ES facilitated the identification of efficient management measures towards EB-MSP as shown in MS 4. Moreover, the approach supported the integration of various data sets and therefore of multiple activities and ecosystem components by using a single metric (e.g. provisioning services). Based on a definition of performance measures or indicators, an evaluation of management effectiveness is enabled (Polasky et al., 2008; White et al., 2012; Guerry et al., 2012). The depiction of trade-offs summarizes assessment results and facilitates the formulation of recommendations by identifying the best possible level of return for its given level of risk (Sanon et al., 2012; Grêt-Regamey et al., 2013c). Accordingly, direct and indirect consequences of management strategies can be depicted in a transparent form and linked to decision making processes in the course of MSP. Applied pro-actively, this approach promotes the adaptive and future-oriented elements of MSP in highlighting the most efficient management strategies (MS 4).

The MCA, applied to condense stakeholder interests in MS 5 supported the identification of the most relevant options to consider for future management. Consequently, identified high level goals and operational objectives can be communicated based on the justification in the interest of majority. These participatory, integrative and future-oriented elements strengthen planning processes (Wever et al., 2015; Eastern Research Group, 2010). Further, interactions between the stakeholders were facilitated when their interests and priorities given to management objectives were empirically analysed. It is important to be aware of conflicts between stakeholders presenting different sectors when interpreting the results. Those are likewise important when determining potential future conflicts between economic, biological, and socio-cultural activities to pro-actively debate methods of avoiding these conflicts and addressing the 'sustainability' problem.

Here, the process of evaluating management strategies is starting new, analysing and bridging interactions between human activities (MS 1 and MS 2). Such cyclic evaluation processes are ensued when applying e.g. the framework described by Stelzenmüller et al. (2013a) in Figure 1.

In consideration of these results there are many different tools assessing spatial management strategies, demonstrating even more strength in being spatially explicit, adaptive, participative, ecosystem-based, integrative and/or future-oriented. Nevertheless, in the course of planning processes, limitations need to be highlighted.

7.1.2 Limitations of the concepts and tools applied to support EB-MSP

Aiming to showcase the application of spatial explicit tools, weaknesses in both, the tools and their application need to be presented. In particular, the limitations of the studies tied together within this dissertation and their transferability to EB-MSP should be emphasized. For reasons of clarity and comprehensibility, those are adjusted by means of the 10 EU principles to MSP (section 1.1).

(1) Using MSP according to area and type of activity, (2) Defining objectives to guide MSP, (3) Developing MSP in a transparent manner, (4) Stakeholder participation, (5) Coordination with Member States - Simplifying decision processes, (6) Ensuring the legal effect of national MSP, (7) Cross border cooperation and consultation, (8) Incorporating monitoring and evaluation in the planning process, (9) Achieving coherence between terrestrial and maritime spatial planning relation with Integrated Coastal Zone Management (ICZM), (10) A strong data and knowledge base (EC, 2008a).

Starting with the conflict matrix applied in MS 1, limitations were given due to the list of the human uses included. Nature conservation sites were subordinated to closed areas. But such sites may differ in regard to the habitats, species or cultural assets they contain, being e.g. not sensitive to other human uses such as wind farming (Hoffmann et al., 2000; van Deurs et al., 2012). Such essential information might affect management decisions and should be regarded accounting to principle 1 and 10. Further limitations were based on the application of the matrix. The conflict categories mapped within the conflict analysis were assessed according to their spatial, but not to their temporal dimension (MS 1). Indeed, sediment extraction might be an activity specific areas are allocated for, but which is taking place irregularly according to the author's knowledge. In accordance with principle 1, 8 and 10, the duration and intervals of human activities should be considered (Desprez et al., 2009). Moreover, recent management perspectives could change anytime soon: Fishing activities overlapping with wind farms resulted during the future scenario in a conflict potential “mutually exclusive”. According to the authors’ knowledge, planners are nowadays reflecting about giving permissions to pelagic fisheries within wind farm areas.

During synergy analysis, the level of uncertainty already introduced (e.g. coming from the data and kriging outputs) increased by gathering expert opinion in identifying the factors which determine the suitability of an area (MS 2). This is reasoned by the fact, that expert judgement can be wrong or formulated misleading (Guerry et al., 2012; Jacobs et al., 2015). Besides, the factors were weighted by the experts according to optimal growth under farmed conditions. The conditions offshore are hardly comparable to the land based, experimental ones. In order to guard against risks coming from such harsh conditions offshore, ERA is needed (principle 10). Indicating to principle 3, a transparent approach has to underlie such an assessment, in particular to call attention on potential risks aquaculture ventures involve (Stelzenmüller et al., in preparation; Neori et al., 2007).

In MS 1, as a proxy of the ecosystem, solely human activities occurring within the German EEZ were considered during the qualitative risk analysis. Taking the whole Wadden Sea into account would have been justified by principle 9, but likewise associated with numerous conservation laws (national level down to state level) as well as additional different tourism and aquaculture activities which have had to be considered. Consequently, land based influences such as nutrient and organic matter enrichment due to input of fertilizers or organic matter and marine litter (Samhouri and Levin, 2012) are missing. Moreover, all drivers and pressures were considered to be temporarily and spatially constant for a time period of one year, which does not reflect reality (MS 1) (Foden et al., 2011; Stelzenmüller et al., 2010b). Nevertheless, pressure occurrences were reflected during sensitivity analysis. The assessment of the driver footprints revealed that the information regarding the wind farms was lacking, defying principle 10. Although the scale of impact varied with different pressures and with the sensitivity of the nursery grounds to these pressures, the intensity of the general pressures was not reflected. When talking about the nursery grounds, an ecosystem-based approach to ERA would also include a representation of different trophic level species (Cormier et al., 2010). In addition, cumulative or positive effects of the drivers were not considered (cf. human "pressures") in MS 1, even though the inclusion would alter the impacts and satisfy principle 10.

When conducting the quantitative risk assessment, sediment specific parameters were weighted with likely species habitat preferences given in Rachor and Nehmer (2003) to get an overall measures of recovery and mortality for the benthic communities (MS 3). As a consequence, those benthic community specific estimates on mortality and recovery rates reflect rather rough estimates of those parameters. Further, benthic disturbance was only calculated for infaunal benthic communities, while epifaunal species may be more vulnerable to fishing disturbance. Empirical data for instance revealed longer recovery times of benthic epifaunal communities (7 - 8 years) compared to infaunal communities (2 - 5 years) in the German Bight, at least after the impact of cold winters (MS 3). Further, there are effects of natural disturbance, for example by waves. Such details, which should be accounted for when assessing the risks of spatial management scenarios (Hiddink et al., 2006) and which should be reflected according to principle 10, were not included.

As a simplification of real conditions, only a choice of human activities and habitats was integrated when conducting the trade-off assessment. While offending principle 10, the real value of EB-MSP is in the first place shown when considering multiple uses (White et al., 2012). Nevertheless, in MS 4 conflicts got obvious in all scenarios as they already got the norm for the German Bight. The management scenarios were driven by long term objectives (principle 2) carried over from the EU 2020 strategy (principle 5). In contrast to principle 1, the response of the ecosystem was assumed to be static, an adaption of both, the ecosystem and the fishery sector was poorly addressed (MS 4).

Finally, limitations performing the stakeholder preference analysis comprised the comparability of stakeholders surveyed in MS 5. Those included a wide range of activity sectors and geographical spread, getting apparent when comparing the sub-objectives included in the CS surveys: While the North Sea case study only ranked pedestrian objectives such as “promote shipping/transport”, the Baltic Sea case study ranked objectives such as “Space for sailing/boating”, “More angling/bird watching” or “More summer house activity” (MS 4). Although all of the principles were pursued, the interpretation constituted a challenge: Finding similarities among those stakeholders and make reliable comparisons (Van Riper and Kyle, 2014; Gilliland and Laffoley, 2008). Reaching a higher number of respondents when surveying the stakeholders might have leavened this task.

Table 2: Methods and management strategies tested during dissertation process, objectives identified associated with preferences given by stakeholders in Ramos et al. (2014), scenarios, scenario measures, indicators, tools applied, and risk and returns assessed.

Methods	**Management strategies**	**Objectives / *Stakeholder preferences**	**Scenarios**	**Measures**	**Indicators**	**Tools**	**Risk**	**Returns**
Conflict analysis (MS 1)	OWF development for 2025 (Blue Growth)	Enhance friendly energy**	Current and future scenario	OWF development	Human activities	Conflict matrix	Increased conflict potential "mutually exclusive" at the expense of the fishery sector	NA
Synergy analysis (MS 2)	Europe 2020 strategy (Blue Growth, GES, Horizon 2020)	Competitively of aquaculture*	Potential future scenario	Co-location of OWFs and aquaculture	OWF development, aquaculture	AHP, MCE, OWA	NA	Potential of synergies / IMTA at all OWFs tested, depending on season / species selected
Qualitative risk assessment (MS 1)	OWF development for 2025 (GES)	Enhance friendly energy**, reduce benthic disturbance***	Current and future scenario	OWF development	Human activities, nursery grounds	DPSI, Sensitivity / pressure assessment	Drivers for the pressures 'smothering' / 'obstruction' increased	Decreased pressures 'abrasion' and 'extraction' in OWF areas
Quantitative risk assessment (MS 3)	Europe 2020 strategy (GES)	Preserve GES*****, enhance friendly energy**, reduce benthic disturbance***	Current and future scenario	Shift of benthic trawling due to OWF development	Benthic trawling, benthic communities	Sensitivity / pressure assessment, BBN	Increased disturbance in 8% of the remaining area / benthic communities	NA
Trade-off assessment (MS 4)	Europe 2020 strategy (Blue Growth, GES)	Ensure high resource rent****	Multiple scenarios	OWF development, Natura 2000, Co-location	Ecosystem services (ES)	ES valuation, Trade-off assessment,	Decrease in supporting services (habitats)	Increase in provisional services (food from aquaculture, wind energy)

Methods	Management strategies	Objectives / *Stakeholder preferences	Scenarios	Measures	Indicators	Tools	Risk	Returns
Stakeholder preference analysis (MS 5)	Natura 2000; WFD; CFP; International, national and local policies	Multiple economic, ecological and socio-cultural objectives	NA	Multiple measures for the North Sea	Stakeholder interests	Stakeholder questionnaire, MCA	No consensual preferences for “competitively of aquaculture”, “competitively of fisheries”, “preserve target stocks/GES” or “ensure high resource rent"	Consensus preferences were given to “reduce benthic disturbance” and "enhance friendly energy"

* Stakeholder preferences given (***** = highly prioritized); AHP, Analytical Hierarchy Process; BBN, Bayesian Belief Network; DPSI, Driver Pressure State Impact model; ES valuation, Ecosystem Service evaluation; GES, Good Environmental Status; IMTA, Integrated Multi-Trophic Assessment; MCA, Multi Criteria Analysis; MCE, Multi Criteria Evaluation; OWA, Ordered Weighted Average process; OWF, Offshore wind farm

7.1.3 Risks and returns of marine spatial management strategies for the German Bight

First results for the German Bight suggest that current and future spatial management strategies (Table 2) have a considerable effect on the fishery sector and the resources it is depending on:

The conflict analysis (MS 1) revealed risks in terms of overlapping drivers acting in the German EEZ. Assessing the current management, approx. 20 % of the whole area was affected by the conflict level "mutually exclusive". The results were reasoned by fishing activities overlapping Natura 2000 sites, which were allocated in order to achieve GES. The future scenario performed in MS 1 highlighted an increasing risk of potential conflicts. The development of offshore wind farms, designated in the course of the IMP and the Blue Growth objective, doubled its area. This was leading in combination with other human uses to the conflict potential "mutually exclusive". As vessel traffic is prohibited in wind farm areas in addition to 500 m-wide marginal buffer zones, an area of 4525.64 km² was allocated to wind farms, leading to a loss of 9.27 % of potential fishing grounds in the entire German EEZ. In addition, 31.5 % of the EEZ is lost for demersal fishing due to designated Natura 2000 sites (MS 1). Since profitable fishing grounds in the North Sea are relatively stationary for numerous of fish species, fisheries may need to change their target species. This might be possible for larger fishing fleets but not for individual fishermen or small fishing associations given the costs involved (Berkenhagen et al., 2010). As a potential loss of fishing grounds could lead to an increased competition and conflicts, catch rates will likely decrease and individual fishermen as well as small fishing associations will suffer economically (Berkenhagen et al., 2010).

Sustainable development, as demanded by IMP, should meet the needs of the present without compromising the ability of future generations to meet their own needs. As the European 2020 strategy pursues economic growth while referencing to aquaculture as a future perspective desired, some fishermen might change sides - from fisheries towards aquaculture.

Results acquired during the synergy analysis performed in MS 2 showed, that the conditions for fish proved to be highly appropriate, especially during summer. The evaluation revealed further suitable sites in coastal areas (e.g. Atlantic cod *Gadus morhua*). Such a management strategy would already be advantageous due to logistics alone (Stelzenmüller et al., in preparation; Buck and Krause, 2012). Though fish can be cultured offshore the whole year around, they require a high degree of care (feeding, clearing of cages etc.) (MS 2). Moreover, results showed several wind farms were de facto suitable sites for Integrated Multi-Trophic Aquaculture (IMTA) systems combining fish species, bivalves and seaweeds. As the IMTA technique would accommodate the GES requirements due to its neutral environmental load, such an approach would be preferred in management. Further, some of the species assessed (oarweed *Laminaria digitata,* cuvie *Laminaria hyperborea*) bring along better growth rates when they are cultured near fish farms (Grote and Buck, 2014). Consequently, management strategies considering a combination of candidates (e.g. *G. morhua,* blue mussel *Mytilus edulis* and sea beech *Delessaria sanguine*) in coastal areas prove to be highly lucrative – complying with the Europe 2020 Strategy (MS 2).

In accordance with the environmental pillar of the strategy, namely the MSFD, potential impacts of major human pressures were assessed during the qualitative risk assessment. The risks of future spatial management scenarios were examined according to their effects on plaice nursery grounds (MS 1). Given that the nursery grounds are associated to soft bottom areas, their vulnerability under the current management strategy was affected mainly by fisheries or dredging, respectively (Stelzenmüller et al., 2010b). When taking into account the future offshore wind farm development objectives for 2025, the drivers for smothering and obstruction increased as a direct effect for the plaice nursery grounds. In contrast, abrasion and extraction exerted by demersal fisheries decreased from a magnitude of 100 % to 85.57 %. It can therefore be expected, that for plaice the direct risks due to wind farm development are on a limited scale (MS 1). However, as an indirect effect, fisheries would be displaced to other grounds since the wind farm areas are inaccessible. In need for new areas, this driver could increase its pressure for abrasion and extraction towards the most sensitive juvenile plaice nursery grounds. Next to siltation, those pressures gained the highest risk scores when evaluating the sensitivity of plaice juveniles. Another potential effect discussed was an increased turbidity, caused by obstruction, which could affect larval stages through reduced feeding efficiency. In turn, slow growth rates of juvenile plaice would affect the recruitment. Another indirect effect is caused by the demersal fisheries, which lead to larger by catch of non-target fish species as for juvenile plaice. In the light of the GES requirements such effects should be prevented. Nevertheless, it might be assumed that offshore wind farms are highly attractive to juvenile plaice given the typical increase in productivity and biodiversity for several meters around the OWFs (MS 1) (Burkhard et al., 2011).

In order to conduct a probabilistic measure of the risk of changing the current state of benthic disturbance by bottom trawling due to future MSP measures in the German EEZ of the North Sea, the quantitative risk assessment was applied in MS 3. The benthic disturbance was assessed as the ratio between relative local mortality by benthic trawling and the recovery potential after a trawl event. The analysis was based on the assumption that vessels conducting demersal mixed or crustacean fishery reallocate their effort in areas of potential large catch or previous knowledge and experience. Results showed that the assumed shift in fishing frequencies did not result in significant changes of the average likely value of the disturbance indicator (MS 3). The spatial management scenario simulated a general spatial shift of fishing effort from medium fished areas to low and highly fished areas due to the development of offshore renewables in areas where 15% and 3 % of the total average beam trawl effort took place. However, the second disturbance indicator (assuming unequal environmental impact by different fleets) still worsens in approximately 8 % of the study area. Even though the affected area is smaller than expected, this is a conservative estimate as uncertainties were eliminated during the assessment process. This observed difference in spatial pattern of the two disturbance indicators, resulting from the weighting of the impact of the different fishing fleets, is highly important: Even when distinguishing between the impact induced by the fleets, the consequences of increased fisheries lead to increased disturbance (Fock et al., 2011). Furthermore, the both disturbance indicators implemented described a range of likely outcomes of disturbance modelling (MS 3). In this sense it reflects a transparent assessment of uncertainty in impact prediction. When integrating the factor of natural disturbance, e.g. by tidal and wave stress as well as daily and seasonal temperature variability, the resilience to fishing disturbance will be even lower in zones with larger water depths (e.g. Hiddink et al. (2006)). Communities in stressed environments such as shallow coastal areas are well adapted to natural stress and will probably never show a recovery to "undisturbed" communities (Becker et al., 1992; Neumann et al., 2013; Elliott and Quintino, 2007).

In order to ensure a 'Green Growth' which combines the elements of GES and Blue Growth, both, risk and returns were incorporated when assessing future management strategies based on a spatially explicit ES trade-off assessment. The highest risk appeared to be a decrease in supporting services such as habitats. In coastal zones, this was caused by demersal trawling, within the German EEZ by OWF development (MS 4). As a consequence thereof an increase in provisional services, caused by wind energy and food from aquaculture, occured. On a long term 'food from fisheries' could decline, again related to OWF development and demersal trawling activities in the remaining areas (MS 4). Those activities exert pressures on the seafloor and alter the benthic habitat structure and the state of benthic communities (Fock et al., 2011). In turn, slow growth rates of juvenile fish species could affect the recruitment and the Spawning Stock Biomass (SSB) later on. Nevertheless, first evidence is given to OWFs attracting fish species such as *G. morhua*, reasoned by shelter and feeding opportunities inside the OWF areas (Stenberg et al., 2015). Moreover, an increased productivity can be expected due to wake effects in between the OWF turbines, leading to an increased suitability for demersal and pelagic fish species (Christiansen and Hasager, 2005). Such positive effects could not only compensate the pressures, they could even exceed them. Here, scientific uncertainty should be incorporated in a transparent way. Further, more studies need to be undertaken to assess and integrate positive effects of wind farm development. Moreover, only the area effectively covered by artificial hard substrates should be considered in future studies.

The results of the stakeholder preference analysis conducted in MS 5 showed, that several of the European stakeholders prefer a balance between socio-cultural, economic and ecological objectives. However, a large number favoured the ecological objective (and up to a certain extent the socio-cultural) in comparison to the economic objective (MS 5). This is confirming the Green Growth course the Juncker commission is setting its sights on (EC, 2014b). When analyzing the results individually for each stakeholder group, the fisheries, aquaculture and tourism tend to give higher importance to economic and socio-cultural objectives (Stelzenmüller et al., 2013b). In addition, they showed similar results, whereas the remaining stakeholder groups did not (MS 5). The ranking of the sub-objectives demonstrated an increasing complexity and dissimilarity. In particular, the Coastal North Sea case study is the one where there were more outliers highlighting the difficulty of reaching consensual preferences for e.g. “preserve target stocks/GES”, “ensure good water quality” or “ensure high resource rent” (MS 5 and Tab. 2). The highest outliers were given for the sub-objectives “competitively of aquaculture” and “competitively of fisheries”. More or less consensus preferences were given to the fourth highest sub-objective “reduce benthic damaging” and “enhance friendly energy”. Here, traditional economic activities such as fisheries may have to compete with more recent activities such as renewable energies or nature conservation for both, space and social acceptance (MS 5). Nevertheless, the two sub-objectives ranked last were the ones the German MSP is giving priority to: the competitiveness of wind farms and shipping/transport. This might be reasoned to the fact, that such objectives are already pursued by the German MSP.

In summary, the German Bight is undergoing multiple modifications coming from current and future spatial management strategies, in particular for the fishery sector (Tab. 2). Considering the information given above, the practical implementation of an informed approach to EB-MSP and its underlying decision making processes would be a giant stride towards a balance in between sustainable use and ecosystem health.

7.2 How to put EB-MSP in practice

EB-MSP is a marine management tool geared to organise human activities in space and time, accounting for a balance between sustainable use and ecosystem health. In support of EB-MSP, the application of place-based tools is highly recommended to resolve human activities and the use of marine resources in space and time. As shown in the previous chapters, there is great deal of work to build on. Further, the potential risks and returns already identified as being related to current and future spatial management options should be used to inform the decision making processes. Nevertheless, there remain some issues open to debate, which could smooth the way towards EB-MSP when effectively addressed in future.

7.2.1 Recommendations towards EB-MSP balancing sustainable use and ecosystem health

IMP, the Europe 2020 Strategy and the MSFD call for a concrete policy tool that accounts for resource stability and interests of many different stakeholders (section 7). In Germany, proposals from a range of economic and technological sectors need to be evaluated by planning authorities (e.g. offshore wind energy, aquaculture) (Ehler and Douvere, 2009). Further, heightened interest is posed from society (headed by NGOs) as evidenced by the case of the Natura 2000 sites (section 1.3). Such occurrences reflect an increasing level of debate and underline the importance of an efficient EB-MSP approach. However, EB-MSP should be future oriented and not only describe the status quo. It should be adaptable to new visions and desires, integrate multiple objectives and feature economic valuations to allow pro-actively decision making (Katsanevakis et al., 2011; Foley et al., 2010).

As the German MSP is recently under revision, decisions concerning future developments in coastal areas are so heavily debated. Incentives must be provided, generating planning certainty towards developments such as sustainable offshore wind energy or sustainable offshore IMTA. Stakeholders of the coastal areas and the society in general are in the need to get more information about the risks of human activities that coexist but are conflicting. Decision-Support Tools (DST) should be applied to facilitate the presentation of spatial allocation decisions, even if it is about priority uses (Halpern et al., 2012). If the current German MSP would include all sectors and spatial uses, all consequences of management strategies (e.g. loss of fishing grounds) could be clarified. Planning authorities would be enabled to communicate potential effects before implementing management options, encouraging the affected stakeholders to participate in plan development to find the best consensual decision (MS 1). Therefore, results such as the ones coming from the conflict analysis (revealing an area of nearly 41 % lost due to wind farms and areas designated to Natura 2000 sites) need to be integrated in trade-off assessments of management strategies (MS 4). Given the increasing competition for space in coastal areas it is also important to increase prospective synergies. Negative effects such as increasing competition and conflicts or decreasing catch rates could be counteracted by developing adaptive strategies such as the here presented co-use locations (MS 3). Indeed, the competition for maritime space and the need for sustainable food production could offer new facilities to the fishery sector. Ventures towards aquaculture development are encouraged and desired by the EU (section 7). Accordingly, the concept of co-locating offshore wind farms with IMTA systems receives increased significance (Buck et al., 2004). Therefore, different spatial co-location scenarios regarding aquaculture and IMTA as the ones assessed during synergy analysis present a milestone. While the fishery sector is adapted to fish products per se, IMTA techniques prove to be more lucrative while having reduced impact on the environment (MS 2). Though it does not account for economic viability analyses for the respective candidates, the approach presented in MS 2 illustrated how competing needs might be balanced by strategic planning for the needs of sectors. Nevertheless, being aware of the positive effects OWF development could have (sec. 7.1.3), more studies need to be undertaken before giving permissions to the aquaculture sector.

Even more important are the environmental risks coming from indirect effects such as displaced bottom trawling. If demersal fisheries get displaced from the Natura 2000 sites and OWF areas and do not change their target species or get involved in aquaculture development, they will probably increase their effort in the remaining areas not closed for fisheries (MS 1 and 3). Having in mind the GES requirements, potential impacts of such major human pressures and risks of future spatial management scenarios on the environment need to be considered (Halpern et al., 2008; Stelzenmüller et al., 2010b). Applying risk assessments by taking into account the number of fishing vessels occurring in the German EEZ and the dimensions of the gears deployed needs to be part of EB-MSP. Being aware of recovery times of benthic communities (2 - 8 years, MS 3) and e.g. nursery grounds of plaice (28 month to several years, MS 1) and the fact, that a proportion of 60 % of the surface area of the North Sea is trawled at least once a year (section 1.5.2) should be reason enough to eliminate destructive fisheries from sensitive areas (Rijnsdorp et al., 2015).

When selecting specific outcomes, the success of a management strategy can be examined in application of indicators or reference targets which consequently enables adaptive management (Collie et al., 2013; Van der Biest et al., 2014). Integrating those criteria in holistic assessments, the effectiveness of management strategies is assessed (Burkhard et al., 2012). Beyond, the efficiency of management measures can be evaluated by assessing the trade-off between the costs (risks) and benefits (economic return) of a management option (Polasky et al., 2008; White et al., 2012; Guerry et al., 2012). Doing this in consideration of a common ground such as 'per unit area' enables planners to conduct fully integrated assessments (MS 4). Recommendations can be formulated across environmental, economic and social dimensions of marine systems. Consequently, EB-MSP and its underlying decision making processes can be informed.

Aiming to empirically prioritize alternative management strategies, they can be analysed being ranked by stakeholders. The interests and needs highlighted by several stakeholders from distinctive institutions reflect reality. Bringing those in line with final management decisions facilitates the communication and consequently the implementation of strategic plans (MS 5).

It is important that the results of an approach like the one presented here is utilised by policy makers before they make final decisions, as this should enhance social acceptance due to the greater transparency and inclusiveness. Right now, as no risk or conflict has been dealt with, accordingly there is no mechanism for conflict resolution.

7.2.2 Future perspectives: the importance of scientific advice to underpin EB-MSP

In future, German EB-MSP requires an integrated assessment process considering all ecosystem functions and the potential impacts of the direct, indirect and combined effects of human drivers. In regard to the fishery sector a comprehensive analysis defining principal areas for all vessels operating in a given planning area has to be included. Such an integrated assessment is also promoted by the IMP, the European 2020 Strategy, MSFD obliging EU member states to achieve GES by 2020 and the MSP directive to achieve Blue Growth. In turn, this requires member states to conduct an initial assessment of the current state of the marine environment as shown in Figure 1. Subsequently, a strategy for Green Growth needs to be developed as the combined alignment of MSP and GES management strategies should be considered in future planning processes. In addition, a coherent planning and assessment system that integrate coastal (under the jurisdiction of the Federal States) and offshore areas should be considered.

EB-MSP implements multi-objective planning when contributing to the MSFD objective GES. Nevertheless, in contrast to the MSFD, no indicators describing the effectiveness of management measures are given. Accordingly, guidance towards spatially explicit optimization measures should reflect management efficiency.
The tools presented can be further developed to aid real life decision making, being problem-focused and needs-driven. Their use increase transparency of decisions made by end users, and increase the commitment of the stakeholders – provided that decisions are made based on environmental needs. When having in mind the Green Growth objective, the best compromise tends to reduced conflicts and increased synergies between human activities, avoidance of risks and impacts on ecosystem components and a high degree of stakeholder satisfaction.

Analytical approaches to environmental assessment require a coherent knowledge of marine systems and underlying ecosystem processes. Further investigations are needed to integrate such comprehensive knowledge. Consequently, uncertainty margins in model prediction can be diminished. In turn, this requires to integrate elements from a variety of interdisciplinary sources (models, data and assessment methods). Nevertheless, even when applying off-the-shelf DSTs such as InVest (MS 4) to support decision making, system knowledge, skills to edit the input data and operate the model are still required. The quality of the input determines the quality of the output and most of the tools require a high level of expertise. The work conducted during dissertation process was highly multidisciplinary and got complex when the data behind the tools needed to be processed. Geostatistics (ordinary kriging), modelling (condensation of the VMS data) or sociology (the compilation of stakeholder questionnaires) take time to get familiar with the wordings and methods applied. From the authors' perspective and given the interdisciplinary background of EB-MSP, it has been a tough time to bring all tools and data sets together. Consequently, scientific underpinning is still inevitable to identify the most suitable management strategies for the ecosystem German Bight.

7.3 Conclusion

The work presented here has described a range of approaches to assess the effects of management strategies and therefore, the (environmental) criteria a holistic trade-off assessment requires. The implementation of such methods ease trade-offs, facilitates the communication of these and offer guidance to decision makers concerning efficient future management strategies. The present results illustrate how competing needs might be balanced in planning for both, sustainable use and ecosystem health in the German EEZ of the North Sea. An extra value for EB-MSP is already in the process, as the tools do not provide any more quality of outputs than what the quality is of the inputs. In turn, this requires planners to deal with environmental issues. Done well, the assessments can help to examine future returns and clarify potential risks by involving future management scenarios while demonstrating the need for an ecosystem approach to risk management techniques using geo-spatial tools. In Germany, EB-MSP requires an integrated assessment process considering all ecosystem functions and the potential impacts of the direct, indirect and combined effects of human drivers as well as the related uncertainties at a common spatial scale. The focus should be on participatory, adaptive, ecosystem-based, integrative and future-oriented GIS-based tools in particular, which allow the spatially explicit description of conflicts, synergies, risks or benefits in their full dimension to support the concept of EB-MSP.

7.4 References

Becker, G. A., Dick, S., and Dippner, J. W. 1992. Hydrography of the German Bight. Marine Ecology Progress Series, 91: 9-18.

Berkenhagen, J., Döring, R., Fock, H. O., Kloppmann, M. H. F., Pedersen, S. A., and Schulze, T. 2010. Decision bias in marine spatial planning of offshore wind farms: Problems of singular versus cumulative assessments of economic impacts on fisheries. Marine Policy, 34: 733-736.

Boroushaki, S., and Malczewski, J. 2008. Implementing an extension of the analytical hierarchy process using ordered weighted averaging operators with fuzzy quantifiers in ArcGIS. Computers and Geosciences, 34: 399-410.

Buck, B. H., and Krause, G. 2012. Integration of Aquaculture and Renewable Energy Systems. Encyclopaedia of Sustainability Science and Technology, 1: 511-533.

Buck, B. H., Krause, G., and Rosenthal, H. 2004. Extensive open ocean aquaculture development within wind farms in Germany: the prospect of offshore co-management and legal constraints. Ocean & Coastal Management, 47: 95-122.

Burkhard, B., Kroll, F., Nedkov, S., and Müller, F. 2012. Mapping ecosystem service supply, demand and budgets. Ecological Indicators, 21: 17-29.

Burkhard, B., Opitz, S., Lenhart, H., Ahrendt, K., Garthe, S., Mendel, B., and Windhorst, W. 2011. Ecosystem based modeling and indication of ecological integrity in the German North Sea—Case study offshore wind parks. Ecological Indicators, 11: 168-174.

Christiansen, M. B., and Hasager, C. B. 2005. Wake effects of large offshore wind farms identified from satellite SAR. Remote Sensing of Environment, 98: 251-268.

Collie, J. S., Adamowicz, W. L., Beck, M. W., Craig, B., Essington, T. E., Fluharty, D., Rice, J., et al. 2013. Marine spatial planning in practice. Estuarine, Coastal and Shelf Science, 117: 1-11.

Cormier, R., Kannen, A., Morales Nin, B., Davies, I., Greathead, C., Sarda, R., Diedrich, A., et al. 2010. Risk-based frameworks in ICZM and MSP decision-making processes. ICES ASC, 2010.

Desprez, M., Pearce, B., and Le Bot, S. 2009. The biological impact of overflowing sands around a marine aggregate extraction site: Dieppe (eastern English Channel). ICES Journal of Marine Science, 67: 1 - 8.

Eastern Research Group, I. E. 2010. MARINE SPATIAL PLANNING STAKEHOLDER ANALYSIS. 76.

EC 2008a. COMMISSION OF THE EUROPEAN COMMUNITIES. COMMUNICATION FROM THE COMMISSION Roadmap for Maritime Spatial Planning: Achieving Common Principles in the EU COM(2008) 791 final.

EC 2008b. European Commission. DIRECTIVE 2008/56/EC OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL of 17 June 2008 establishing a framework for community action in the field of marine environmental policy (Marine Strategy Framework Directive). Official Journal of the European Union, L 164/19.

EC 2012. European Commission. Progress of the EU's Integrated Maritime Policy – Report from the commission to the European Parliament, the council, the European Economic and social committee and the committee of the Regions. Luxembourg: Publications Office of the European Union: 11 pp.

EC 2014a. European Commission. DIRECTIVE 2014/89/EU OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL of 23 July 2014 establishing a framework for maritime spatial planning. Official Journal of the European Union, L 257/135.

EC 2014b. European Commission. Press release: The Juncker Commission: A strong and experienced team standing for change. IP/14/984. http://europa.eu/rapid/press-release_IP-14-984_en.htm, Retrieved: June 13th 2015, 18:04.

EC 2015. European Commission. Aquatic Ressources. http://ec.europa.eu/programmes/horizon2020/en/area/aquatic-resources, Retrieved: April the 6th 2015, 14:21.

Ehler, C., and Douvere, F. 2009. Marine Spatial Planning. *In* A Step-by-Step Approach toward Ecosystem-based Management, p. 99. Ed. by I. O. Commission. Intergovernmental Oceanographic Commission.

Elliott, M. 2002. The role of the DPSIR approach and conceptual models in marine environmental management: An example for offshore wind power. Marine Pollution Bulletin, 44: iii-vii.

Elliott, M., and Quintino, V. 2007. The Estuarine Quality Paradox, Environmental Homeostasis and the difficulty of detecting anthropogenic stress in naturally stressed areas. Marine Pollution Bulletin, 54: 640-645.

Fock, H. O., Kloppmann, M., and Stelzenmüller, V. 2011. Linking marine fisheries to environmental objectives: a case study on seafloor integrity under European maritime policies. Environmental Science & Policy, 14: 289-300.

Foden, J., Rogers, S. I., and Jones, A. P. 2011. Human pressures on UK seabed habitats: a cumulative impact assessment. Marine Ecology Progress Series, 428: 33-47.

Foley, M. M., Halpern, B. S., Micheli, F., Armsby, M. H., Caldwell, M. R., Crain, C. M., Prahler, E., et al. 2010. Guiding ecological principles for marine spatial planning. Marine Policy, 34: 955-966.

Freitas, V., Campos, J., Skreslet, S., and van der Veer, H. W. 2010. Habitat quality of a subarctic nursery ground for 0-group plaice (Pleuronectes platessa L.). Journal of Sea Research, 64: 26-33.

Gemitzi, A., Tsihrintzis, V. A., Voudrias, E., Petalas, C., and Stravodimos, G. 2007. Combining geographic information system, multicriteria evaluation techniques and fuzzy logic in siting MSW landfills. Environmental Geology, 51: 797-811.

Gilliland, P. M., and Laffoley, D. 2008. Key elements and steps in the process of developing ecosystem-based marine spatial planning. Marine Policy, 32: 787– 796.

Gorsevski, P. V., Donevska, K. R., Mitrovski, C. D., and Frizado, J. P. 2012. Integrating multi-criteria evaluation techniques with geographic information systems for landfill site selection: a case study using ordered weighted average. Waste Management, 32: 287-296.

Grêt-Regamey, A., Brunner, S. H., Altwegg, J., and Bebi, P. 2013a. Facing uncertainty in ecosystem services-based resource management. Journal of Environmental Management, 127: S145-S154.

Grêt-Regamey, A., Brunner, S. H., Altwegg, J., Christen, M., and Bebi, P. 2013b. Integrating expert knowledge into mapping ecosystem services tradeoffs for sustainable forest management. Ecology and Society, 18.

Grêt-Regamey, A., Celio, E., Klein, T. M., and Wissen Hayek, U. 2013c. Understanding ecosystem services trade-offs with interactive procedural modeling for sustainable urban planning. Landscape and Urban Planning, 109: 107-116.

Grote, B., and Buck, B. H. 2014. The IMTA-approach for nutrient balanced aquaculture: Evaluating the potential of North Sea species from onshore RAS to offshore environments. Aquaculture (in review).

Guerry, A. D., Ruckelshaus, M. H., Arkema, K. K., Bernhardt, J. R., Guannel, G., Kim, C.-K., Marsik, M., et al. 2012. Modeling benefits from nature: using ecosystem services to inform coastal and marine spatial planning. International Journal of Biodiversity Science, Ecosystem Services & Management, 8: 107-121.

Halpern, B. S., Diamond, J., Gaines, S., Gelcich, S., Gleason, M., Jennings, S., Lester, S., et al. 2012. Near-term priorities for the science, policy and practice of Coastal and Marine Spatial Planning (CMSP). Marine Policy, 36: 198-205.

Halpern, B. S., Walbridge, S., Selkoe, K. A., Kappel, C. V., Micheli, F., D'Agrosa, C., Bruno, J. F., et al. 2008. A global map of human impact on marine ecosystems. Science, 319: 948-952.

Hiddink, J. G., Jennings, S., Kaiser, M. J., Queirós, A. M., Duplisea, D. E., and Piet, G. J. 2006. Cumulative impacts of seabed trawl disturbance on benthic biomass, production, and species richness in different habitats. Canadian Journal of Fisheries and Aquatic Sciences, 63: 721-736.

Hoffmann, E., Astrup, J., Larsen, F., Munch-Petersen, S., and Støttrup, J. 2000. Effects of marine windfarms on the distribution of fish, shellfish and marine mammals in the Horns Rev area. ELSAMPROJEKT A/S. Baggrundsrapport nr. 24

Jacobs, S., Burkhard, B., Van Daele, T., Staes, J., and Schneiders, A. 2015. 'The Matrix Reloaded': A review of expert knowledge use for mapping ecosystem services. Ecological Modelling, 295: 21-30.

Johnson, S., Low-Choy, S., and Mengersen, K. 2012. Integrating bayesian networks and geographic information systems: Good practice examples. Integrated Environmental Assessment and Management, 8: 473-479.

Katsanevakis, S., Stelzenmüller, V., South, A., Sørensen, T. K., Jones, P. J. S., Kerr, S., Badalamenti, F., et al. 2011. Ecosystem-based marine spatial management: Review of concepts, policies, tools, and critical issues. Ocean & Coastal Management, 54: 807-820.

Kjræulff, U. B., and Madsen, A. L. 2012. Bayesian Networks and Influence Diagrams: A Guide to Construction and Analysis: A Guide to Construction and Analysis, Springer.

Kovács, E., Kelemen, E., Kalóczkai, Á., Margóczi, K., Pataki, G., Gébert, J., Málovics, G., et al. 2014. Understanding the links between ecosystem service trade-offs and conflicts in protected areas. Ecosystem Services.

Neori, A., Troell, M., Chopin, T., Yarish, C., Critchley, A., and Buschmann, A. H. 2007. The need for a balanced ecosystem approach to blue revolution aquaculture. Environment, 49: Research Library Core, pg. 37.

Neumann, H., Reiss, H., Ehrich, S., Sell, A., Panten, K., Kloppmann, M., Wilhelms, I., et al. 2013. Benthos and demersal fish habitats in the German Exclusive Economic Zone (EEZ) of the North Sea. Helgoland Marine Research, 67: 445-459.

Parravicini, V., Rovere, A., Vassallo, P., Micheli, F., Montefalcone, M., Morri, C., Paoli, C., et al. 2012. Understanding relationships between conflicting human uses and coastal ecosystems status: A geospatial modeling approach. Ecological Indicators, 19: 253-263.

Polasky, S., Nelson, E., Camm, J., Csuti, B., Fackler, P., Lonsdorf, E., Montgomery, C., et al. 2008. Where to put things? Spatial land management to sustain biodiversity and economic returns. Biological Conservation, 141: 1505-1524.

Rachor, E., and Nehmer, P. 2003. Erfassung und Bewertung ökologisch wertvoller Lebensräume in der Nordsee. Alfred Wegener Institute for Polar and Marine Research, Bremerhaven.

Ramos, J., Soma, K., Bergh, O., Schulze, T., Gimpel, A., Stelzenmuller, V., Makinen, T., et al. 2014. Multiple interests across European coastal waters: the importance of a common language. ICES Journal of Marine Science, 72: 720-731.

Rijnsdorp, A. D., Eigaard, O. R., Hintzen, N. T., and Engelhard, G. H. 2015. Oceans Past V - The evolution of the impact of bottom-trawling on demersal fish populations and the benthic ecosystem. http://www.ices.dk/news-and-events/news-archive/news/Pages/%E2%80%98The-evolution-of-bottom-trawling-impact-on-demersal-fish-populations-and-the-benthic-ecosystem.aspx, Retrieved: June 18th 2015, 18:45.

Samhouri, J. F., and Levin, P. S. 2012. Linking land- and sea-based activities to risk in coastal ecosystems. Biological Conservation, 145: 118-129.

Sanon, S., Hein, T., Douven, W., and Winkler, P. 2012. Quantifying ecosystem service trade-offs: the case of an urban floodplain in Vienna, Austria. J Environ Manage, 111: 159-172.

Stelzenmüller, V., Breen, P., Stamford, T., Thomsen, F., Badalamenti, F., Borja, Á., Buhl-Mortensen, L., et al. 2013a. Monitoring and evaluation of spatially managed areas: A generic framework for implementation of ecosystem based marine management and its application. Marine Policy, 37: 149-164.

Stelzenmüller, V., Gimpel, A., Gopnik, M., and Gee, K. in preparation. Aquaculture site-selection and marine spatial planning: The roles of GIS-based tools and models. Richard Langan and Bela H. Buck: Aquaculture perspective of multi-use sites in the open ocean. The untapped potential for marine resources in the anthropocene. Springer book.

Stelzenmüller, V., Lee, J., Garnacho, E., and Rogers, S. I. 2010a. Assessment of a Bayesian Belief Network-GIS framework as a practical tool to support marine planning. Mar Pollut Bull, 60: 1743-1754.

Stelzenmüller, V., Lee, J., South, A., and Rogers, S. I. 2010b. Quantifying cumulative impacts of human pressures on the marine environment: a geospatial modelling framework. Marine Ecology Progress Series, 398: 19-32.

Stelzenmüller, V., Schulze, T., Fock, H. O., and Berkenhagen, J. 2011. Integrated modelling tools to support risk-based decision-making in marine spatial management. Marine Ecology Progress Series, 441: 197-212.

Stelzenmüller, V., Schulze, T., Gimpel, A., Bartelings, H., Bello, E., Bergh, O., Bolman, B., et al. 2013b. Guidance on a Better Integration of Aquaculture, Fisheries, and other Activities in the Coastal Zone: From tools to practical examples. COEXIST project: 79pp.

Stenberg, C., Støttrup, J. G., van Deurs, M., Berg, C. W., Dinesen, G. E., Mosegaard, H., Grome, T. M., et al. 2015. Long-term effects of an offshore wind farm in the North Sea on fish communities. Marine Ecology Progress Series, 528: 257-265.

Van der Biest, K., D'Hondt, R., Jacobs, S., Landuyt, D., Staes, J., Goethals, P., and Meire, P. 2014. EBI: An index for delivery of ecosystem service bundles. Ecological Indicators, 37: 252-265.

van Deurs, M., Grome, T. M., Kaspersen, M., Jensen, H., Stenberg, C., Sørensen, T. K., Støttrup, J., et al. 2012. Short- and long-term effects of an offshore wind farm on three species of sandeel and their sand habitat. Marine Ecology Progress Series, 458: 169-180.

Van Riper, C. J., and Kyle, G. T. 2014. Capturing multiple values of ecosystem services shaped by environmental worldviews: a spatial analysis. J Environ Manage, 145: 374-384.

Wever, L., Krause, G., and Buck, B. H. 2015. Lessons from stakeholder dialogues on marine aquaculture in offshore wind farms: Perceived potentials, constraints and research gaps. Marine Policy, 51: 251-259.

White, C., Halpern, B. S., and Kappel, C. V. 2012. Ecosystem service tradeoff analysis reveals the value of marine spatial planning for multiple ocean uses. PNAS, 109: 4696–4701.

Zucchetta, M., Franco, A., Torricelli, P., and Franzoi, P. 2010. Habitat distribution model for European flounder juveniles in the Venice lagoon. Journal of Sea Research, 64: 133-144.

8. Significant acronyms and abbreviations

AHP	Analytical Hierarchy Process
BfN	The German Federal Agency for Nature Conservation
BLE	The German Federal Office for Agriculture and Food
BMVI	The German Federal Ministry of Transport and Digital Infrastructure
BBN	Bayesian Belief Network
BSH	The German Federal Agency for Shipping and Hydrography
CBD	Convention on Biological Diversity
CFP	Common Fisheries Policy
COEXIST	Interaction in Coastal Waters: A roadmap to sustainable integration of aquaculture and fisheries (EU FP7 project)
CPUE	Catch per Unit Effort
DG MARE	Directorate-General for Maritime Affairs and Fisheries
DG ENV	Directorate-General Environment
DPSI/R	Driver Pressure State Impact / Response model
DST	Decision Support Tools
EBM	Ecosystem-Based Management
EB-MSP	Ecosystem-Based Marine Spatial Planning
EC	European Commission
EEZ	Exclusive Economic Zone
EMFF	European Maritime and Fisheries Fund
ERA	Environmental Risk Assessment
ES	Ecosystem Services
EU	European Union
FAO	Food and Agricultural Organization
FFH	Flora and Fauna Directive
GAM	Generalised Additive Model
GES	Good Environmental Status
GIS	Geographic Information System
HBD	Habitat and Birds Directive
HELCOM	Baltic Marine Environment Protection Commission
ICES	International Council for the Exploration of the Seas
ICZM	Integrated Coastal Zone Management
IKZM	Integrated Coastal Zone Management for the Coast of Schleswig Holstein
IMP	Integrated Maritime Policy
IMTA	Integrated Multi-Trophic Aquaculture
JRC IES	Joint Research Centre - Institute for Environment
MCA/MCE	Multi-Criteria Analysis or Multi Criteria Evaluation
MKRO	Conference of Ministers for Spatial Planning
MPA	Marine Protected Area
MS	Manuscript
MSFD	Marine Strategy Framework Directive
MSP	Marine Spatial Planning or Maritime Spatial Planning
NGO	Non-Governmental Organisation
NOAA	National Oceanic and Atmospheric Administration
OSPAR	Commission for the Protection of the Marine Environment of the Northeast Atlantic
OSS	Offshore Site Selection project

OWA	Ordered Weighted Average
OWF	Offshore Wind energy Farm
ROKK	Spatial Planning Concept for the Coast of Lower Saxony
SET-plan	Strategic Energy Technology Plan
SDM	Species Distribution Model
SMA	Spatial Managed Areas
UNCLOS	United Nations Convention on the Law of the Sea
SSB	Spawning Stock Biomass
UNESCO	United National Educational, Scientific, and Cultural Organization
VMS	Vessel Monitoring System
WFD	Water Framework Directive
WGMPCZM	ICES Working Group on Marine Spatial Planning and Coastal Zone Management

9. Glossary: Significant terms as denoted during this thesis

Blue Growth	EU long term strategy to support sustainable growth of maritime economies, the sustainable development of marine areas and the sustainable use of marine resources
Common language	Usage of similar wordings to align MSP procedures and ensure flawless communication across all (environmental, economic, and social) disciplines
Cumulative effects	Combined impact of multiple pressures over space and time which can be additive, antagonistic, synergistic
Decision Support Tools	Knowledge-based system that facilitates organizational processes to support decision making
DPSIR model	Conceptual model to illustrate the pathways of effects showing the links between drivers of human activities (Driver) to predict where their pressures (Pressure) have the potential to cause effects that could change the integrity (State) of a subject leading to impact (Impact) on society and their response (Response) to that
Ecosystem-Based Management	Considers the whole ecosystem, including humans featuring the cumulative pressures they are exerting
Ecosystem component	Elements of the natural environment (communities, habitats, resources)
Ecosystem Services	The goods and services marine ecosystems provide: provisioning (e.g. food), regulative (e.g. clean water), supporting (e.g. habitats) or cultural (e.g. aesthetics), depending on the environmental conditions
Effects	Any response by an environmental or social component to an action's impact
Effectiveness	The degree to which objectives are achieved (targeted)
Efficiency	Determined with reference to cost-efficiency (economic)
Efficiency frontier	Best possible level of return for a given level of risk, evaluated by assessing the trade-off between costs (risks) and benefits (economic return)
Environmental Risk Assessment	Frameworks with quantitative or probabilistic measures of risk to evaluate spatial management scenarios comprising Risk identification, Risk analysis and Risk evaluation
Europe 2020 Strategy	Resolution implementing an integrated approach to maritime affairs, features the European demand employment, competitiveness and social cohesion by 2020
Good Environmental	Main goal of the MSFD to promote an environmental status of marine waters where these provide ecologically diverse and dynamic oceans and

Status	seas which are clean, healthy and productive
Green Growth	Combines the elements of GES and Blue Growth
Integrated Maritime Policy	Approach to ocean management and maritime governance to reaffirm the maritime dimension of the EU, to support the sustainable development of seas and oceans, to provide better protection of the state of the ecosystem and to develop coordinated, coherent and transparent decision-making in relation to the Union's sectoral policies
Integrated Multi-Trophic Aquaculture	Combination of aquaculture species to recycle effluent dissolved and particulate nutrients from a higher trophic-level species (fish) to nourish extractive, lower trophic-level species, such as filter feeders
Integrated assessment process	Analytical approach to environmental assessment, bringing together knowledge and elements from a variety of disciplinary sources (models, data and assessment methods)
Management goal	Overarching management goal (e.g. EBM).
Management objective	General objective that describes the direction towards the achievement of the overarching management goal (e.g. GES, Blue Growth)
Management scenario	Simulation of spatial management options
Management strategy	Management option (e.g. sustainable use) that need to be conducted to attain management objectives. Its achievement can be assessed based on operational objectives (e.g. maintain provisioning services) and indicators (e.g. food from fisheries)
Marine Strategy Framework Directive	Strategy to achieve or maintain GES of marine ecosystems which shall apply an EBM, ensuring that the collective pressure of human activities is kept within levels compatible with the achievement of GES by 2020
Marine Spatial Planning	Cross-cutting policy tool that contributes to Blue Growth while applying an EBM to GES
Natura 2000	Network of nature protection areas established under the 1992 Habitats Directive
Spatially explicit approach	The application of place-based tools which allow a transparent evaluation of spatial management effects
Spatial management	The management of all activities (natural and non-natural) within a defined (marine) area
Sustainable development	Development that meets the needs of the present without compromising the ability of future generations to meet their own needs

Trade-off analysis	Comparison of linked costs (risks) and services (economic returns)
Uncertainty	Scientific uncertainty which includes Uncertainty about impact prediction, Uncertainty about the effectiveness of measures and Uncertainty about future states of nature

10. Acknowledgements

At first I would like to express my gratitude to my supervising team, Prof. Dr. Axel Temming and Dr. Vanessa Stelzenmüller. I would like to thank Professor Dr. Axel Temming for his comments and advice given throughout the whole dissertation process. Discussions often provided me with fruitful thoughts. You gave me free rein to determine the focus and methodology of my research while challenging me to answer even the difficult questions.

Special thanks go to Dr. Vanessa Stelzenmüller for her early support, her help during the drafting and realisation of this dissertation, her motivation and, in addition, for her endless patience. Thanks for always being available when I needed you and for your excellent advice and comments during all stages of preparation. Without you I wouldn’t be where I am today. I look forward to work with you in the future!

Further, special thanks go to Dr. Jens Floeter, who never lost sight of the essentials throughout this multidisciplinary work. Thanks for guiding me, for your patience, your ultimately proven spontaneity and your big efforts in reviewing and commenting on my thesis. You challenged me to see the bigger picture. This thesis would not have been the same without you.

I am grateful to all my co-authors who have contributed to the research articles and shared their expertise and ideas - it has been great working with you. Thanks to the Thünen-Institute of Sea Fisheries, the University of Hamburg and ICES for the provision of funding for my studies and administrative support that allowed me to travel and share my research on international conferences.

Finally, I would like to thank my colleagues at the Thünen-Institute, especially my research unit and the fourth floor, for the exciting conference-times and contributions to my work-life balance. Sarah Laura Simons, Henrike Rambo – it was a pleasure. Special thanks to Bettina Walter for coaching me. Marie-Christin Rufert – thank you for looking at the bright side of life and inspiring me to do the same. Thanks to friends and family for their love and support, especially to my stunning sisters – just for being who you are.

11. Declaration on oath

I hereby declare, on oath, that I have written the present dissertation by my own and have not used other than the acknowledged resources and aids.

Hiermit erkläre ich an Eides statt, dass ich die vorliegende Dissertationsschrift selbst verfasst und keine anderen als die angegebenen Quellen und Hilfsmittel benutzt habe.

Hamburg, 23rd July 2015 Signature