Fisheries and Environmental Management Group
Department of Environmental Sciences
Faculty of Biological and Environmental Sciences
University of Helsinki

KNOWLEDGE-BASED MARINE CONSERVATION IN OIL SPILL RISK MANAGEMENT

Riikka Venesjärvi

ACADEMIC DISSERTATION

To be presented, with the permission of the Faculty of Biological and Environmental Sciences of the University of Helsinki, for public examination in the Hall 5 of Main Building (Fabianinkatu 33), on May 13th at 12 noon.

Helsinki 2016

Supervisors:	Prof. Sakari Kuikka Department of Environmental Sciences University of Helsinki, Finland
	Dr. Mika Rahikainen Department of Environmental Sciences University of Helsinki, Finland
Thesis advisory committee:	Adj. Prof. Laura Uusitalo Marine Research Centre Finnish Environment Institute, Finland
	Dr. Miina Karjalainen Maritime Research Centre, Finland
	Dr. Elina Leskinen Department of Environmental Sciences University of Helsinki, Finland
Reviewers:	Prof. Mikael Hildén Climate Change Programme Finnish Environment Institute, Helsinki
	Prof. Charles H. Peterson Institute of Marine Sciences University of North Carolina, United States
Opponent:	Assoc. Prof. Mark Dickey-Collas Science Programme in General Secretary International Council for the Exploration of the Seas, Denmark
Custos:	Prof. Jukka Horppila Department of Environmental Sciences University of Helsinki, Finland

Cover design by Henri Haka

ISBN 978-951-51-2147-9 (paperback)
ISBN 978-951-51-2148-6 (PDF)

Painotalo Westman
Kotka 2016

Contents

List of original publications

This thesis is based on the following publications:

I Venesjärvi R. & Karjalainen J. 2013. Impacts of crude oil on the early development of Baltic herring (*Clupea harengus membras*). The Open Environmental Pollution & Toxicology Journal 4: 1–15.

II Venesjärvi R., Helle I., Mäntyniemi S. & Kuikka S. Estimating the loss and recovery of breeding and migrating seabirds after oil spills using in the northern Baltic Sea. Manuscript.

III Helle I., Jolma A. & Venesjärvi R. In Press. Species and habitats in danger: Estimating the relative risk posed by oil spills in the northern Baltic Sea. Ecosphere.

IV Altartouri A., Ehrnsten E., Helle I., Venesjärvi R. & Jolma A. 2013. Geospatial Web services for responding to ecological risks posed by oil spills in marine environment. Photogrammetric Engineering and Remote Sensing 79: 905–914.

V Venesjärvi R., Helle I. & Jolma A. Conservation prioritization scheme for the threatened species and habitat types in oil spill response. Submitted manuscript.

The publications are referred to in the text by their Roman numerals.

Contributions of authors

I R. Venesjärvi was responsible for the original idea. R. Venesjärvi designed the laboratory experiment and performed the experiment with the assistance of J. Karjalainen. R. Venesjärvi interpreted the results and wrote the manuscript, on which J. Karjalainen provided comments.

II R. Venesjärvi was responsible for the original idea together with I. Helle and S. Kuikka. R. Venesjärvi constructed the BNs with the assistance of I. Helle. R. Venesjärvi gathered the data and carried out the expert interviews to collect estimates related to the ecological impacts of oil spills on seabirds. All authors designed, I. Helle and S. Mäntyniemi coded and R. Venesjärvi provided the model inputs. R. Venesjärvi and I. Helle interpreted the results, and R. Venesjärvi wrote the manuscript and the others provided comments on it.

III I. Helle was responsible for the original idea, constructed the BN, interpreted the results and wrote the manuscript. A. Jolma processed the data and designed, coded, and implemented the risk assessment tool. R. Venesjärvi carried out expert interviews to collect estimates related to ecological valuation. Species and habitat type data were acquired from the OILRISK project, and were further processed by all authors. A. Jolma and R. Venesjärvi provided comments on the manuscript.

IV A. Altartouri was responsible for the description of requirements for informed operational oil spill management and for analyzing and discussing the results. A. Altartouri and E. Ehrnsten mainly designed the map application and A. Altartouri was responsible for its implementation. I. Helle, E. Ehrnsten and R. Venesjärvi participated with A. Altartouri in the description of requirements for informed operational oil spill management, in the description of data and the case study, and in writing the paper. R. Venesjärvi was responsible for the description of the prioritization index. A. Jolma computed the prioritization index and developed server-side geospatial services.

V R. Venesjärvi was responsible for the original idea together with I. Helle. R. Venesjärvi gathered the data and carried out the expert elicitation process to collect estimates related to the conservation value, oil-induced loss, and recovery potential of threatened species and habitat types. R. Venesjärvi and I. Helle designed the conservation prioritization scheme and A. Jolma coded and implemented the prioritization algorithm. Venesjärvi interpreted the results alongside I. Helle and A. Jolma. R. Venesjärvi wrote the article and both co-authors commented on the manuscript.

Abstract
Maritime transport is an efficient way to ferry goods, oil, and chemicals but shipping poses a threat to marine ecosystems. Oil spills have a potential to extinguish or debilitate fish and wildlife populations and habitat types important to the marine ecosystem.

In this thesis, I study the resources and methods for collecting data and knowledge about the adverse impacts of oil on sensitive species and habitat types. Furthermore, I study how ecological knowledge could be passed to decision-makers and how the risks should be communicated. Finally, I discuss future policy improvements and scientific needs for ecological knowledge in oil spill risk management. This forms a synthesis of what kind of ecological information is required for the environmental risk management and conservation of marine ecosystems under oil spill threat.

The thesis includes five papers, where we develop methods to assess the environmental impacts of oil spills and the effectiveness of management practices to mitigate their adverse ecological effects. Improved strategies combine theoretical disciplines, such as population biology with practical oil spill response.

The results demonstrate that environmental risk assessment models can be used to structure problems, integrate knowledge and uncertainty, and persuade decision-makers by visualizing the results. Since the objective of risk assessment is to synthesize information for environmental management and policy design, which should rely on the extensive use of scientific evidence, communication between academia and decision-makers is of great importance. The use of Bayesian networks would improve the current oil spill risk management in the Baltic Sea, since all the variables affecting oil spill risk can be presented in one framework in a transparent manner. Many geospatial services work as tools of informative policy instruments, as they deliver ecological data and knowledge for oil spill risk management. Researchers could also participate more often in the contingency planning or practical management of oil spills as experts. Thus, all the relevant knowledge could be integrated into the decision-making process.

This thesis offers new insights into oil spill risk management in the Baltic Sea and provides examples showing how evidence-based management actions should be chosen and carried out in order to minimize the risks. Policy recommendations are also provided. First, in oil spill risk management, the marine ecosystem should be prioritized based on its conservation value, recovery potential and protection effectiveness. Second, because preventive measures against oil accidents are considered cost-effective, maritime safety should be increased, with stricter and regional ship inspection practices. The effects of policy innovations should be assessed using probabilistic policy-support tools.

1. Introduction

Over the last forty years, humans have changed marine ecosystems more rapidly and extensively than in any comparable period of time in our history, mostly to meet the demands for natural resources, such as food (The Zoological Society of London 2015, Millennium Ecosystem Assessment 2005). Fish populations have declined greatly due to overexploitation, and because marine ecosystems are interconnected, these declines can affect dynamics of marine food webs and ecosystem functioning. Human exploitation also causes habitat loss and degradation which are significant causes of species decline (Halpern et al. 2014). In addition, marine species face sustained harm because of an increase in the amount of pollutants from human activities (e.g. maritime traffic) (The Zoological Society of London 2015).

Our current society demands oil for energy production and to manufacture various consumables. Maritime transport is the most efficient way to move large volumes of cargo, but oil shipping may create a threat to marine ecosystems (Helle et al. 2011, Kokkonen et al. 2010). Oil spills have a potential to extinguish or debilitate populations and habitats important to marine ecosystems. Large oil accidents may cause major costs to society, as they involve expensive oil spill responses, and have adverse impacts on ecological functions that might cause loss of ecosystem services (Helle et al. 2015). At the same time, expanding ports to take more ships can further degrade coastal ecosystems leading to habitat loss (Wan et al. 2016). Ships cause emissions to air in ports and port operations pollute seas and shoreline (Wan et al. 2016). This creates an increasing threat to coastal species and it may decrease their ability to survive from oil spills.

Decisions concerning species and habitat types that should be safeguarded must be made quickly after an oil accident. Since the operational decision-makers are not ecologists, the evaluation process (weighting of species and trade-offs between ecological and human constructs) must be assessed and communicated in advance (Ihaksi et al. 2011). Marine conservation practice generally emphasizes species at risk of imminent extinction, and therefore tends to prioritize species with small populations and/or geographical ranges to avoid a rapid loss of biodiversity (Gaston and Fuller 2008). As well as keystone species, rare and threatened species maintain functions in ecosystems (Reece et al. 2014, Mouillot et al. 2013). Moreover, oil spills are likely to have a relatively small long-term impact on the populations of common species (Ihaksi et al. 2011, Lecklin et al. 2011). Thus, threatened species need to be safeguarded by oil spill responses.

A dispute over whether the conservation objectives should prioritize biodiversity protection over preservation of summer cottages is familiar to decision-makers and policy scholars (Minteer and Miller 2010). An implicit ethical commitment in conservation planning is to embrace the intrinsic value of wild species and ecosystems (Sarkar 2005). However, environmental management is a human endeavor and is legitimately motivated by several anthropogenic concerns (e.g. exploitation of natural resources, transportation and energy production) (Minteer and Miller 2010).

Marine ecosystems and the species and communities within them are intrinsically variable, and the understanding of how they work might be rudimentary (Halpern et al. 2006, Ralls and Taylor 2000). Data are sampled in monitoring programs and researchers analyze them to provide informed advice for management. In addition to epistemic uncertainty, the decision-making process adds social dimensions of uncertainty, such as political, economic, and legal considerations (Ralls and Taylor 2000). That is, the reaction of society to management actions is not certain, as the behavior of people and their commitment to decisions are difficult to predict (Fulton et al. 2011, Haapasaari and Karjalainen 2010). Making decisions in the face of uncertainty is often unavoidable, and thus, it may be helpful to develop quantitative approaches that incorporate uncertainty at each step, from collecting the information to managing the oil spill risk. Failure to acknowledge and treat uncertainty can lead to outcomes with unexpected or undesired results, especially in the management of threatened species and habitat types (Halpern et al. 2006, Regan et al. 2005).

1.1. Risks and probabilities

One of the problems in risk communication is that in ordinary language we use the word risk quite vaguely (e.g. Aven 2010, Calow 2009). This has led to potential confusion in defining the term, as common usages lack precision. In my thesis, I use oil spill risk as a combination of the probability of an event and its adverse effects within a time frame (Burgman 2005, Christensen et al. 2003). This is a common definition in quantitative analysis (Assmuth and Hildén 2008). Concepts related to oil spill risk include a ship accident and its environmental consequences, which are used differently in different disciplines: maritime safety assessments tend to stress accident probabilities, whereas environmental risk assessments focus on ecological vulnerability (Goerlandt et al. 2015, Lecklin et al. 2011). Oil spill risk is spatially distributed: in other words, the spatial distribution of the accident and oil exposure probabilities, and the occurrences of species and habitat types along the shoreline vary (III, Jolma et al. 2014).

Concepts of probability influence interpretation of models and communication. It is essential that the meaning is well-defined already in risk assessments (Aven and Zio 2014). Probability has two dimensions: it can be viewed as statistical uncertainty (aleatory) or systematic uncertainty (epistemic) (Skinner et al. 2014, Burgman 2005, O'Hagan and Oakley 2004). The difference is that aleatory uncertainty represents the observed inherent variation associated with a studied environment, and epistemic uncertainty is the lack of knowledge about evidential elements (Burgman 2005). The latter reflects incomplete knowledge and can be reduced with new evidence.

1.2. Use of information

Information is facts or details about a subject. Data represent values attributed to parameters, and knowledge signifies understanding of systems or concepts. Information has a significant role in oil spill risk management. However, it can be a challenging task to provide information on risks (Assmuth and Hildén 2008). In assimilating, processing and giving meaning to information, different frameworks can be used as the conceptual and procedural constructs (Assmuth and Hildén 2008).

In uncertain and unpredictable business environments, knowledge management systems have been developed to capture collective intelligence and improve performance (Yim et al. 2004). Goerlandt and Montewka (2015) present a framework that functions as an argumentation process based on available evidence and a tool for communication between stakeholders for the risk analysis of maritime transport. In this framework, first, a risk analysis is performed and second, the results from the risk model are communicated to the decision-makers.

1.3. Risk assessment and management

Environmental risk assessment (ERA) aims to predict the nature of the adverse impacts of human pressures on the environment. The process evaluates how likely it is that the environment may be affected as the result of one or more stressors and the magnitude of any harmful effects (spatial and temporal) (Burgman 2005). Assessments are linked to research and policy, and are influenced by the general societal context as well as more specific drivers affecting management (Assmuth and Hildén 2008). In the quantitative assessment of risks, an attempt is made to determine numerically the probabilities of different events and the likely extent of the losses if a particular event takes place.

Risk assessments should involve findings about values and preferences; this is based on the view that risks are subjective and context-dependent (Burgman 2005). Nevertheless, Calow (2009) suggests that the assessment process should be separated from management decisions. The rationale is that the assessment should be as independent as possible, depending upon scientific criteria rather than political or social views or judgments. However, if ERAs are carried out properly, they are evidence-based and include definitions of potential management actions and estimations of their effects. When pursuing comprehensive ERAs that comply with the reasoning described in the literature, risk assessments could follow, for example, the five steps of Formal Safety Assessments (FSAs) introduced by the International Maritime Organization (IMO): 1) identification and 2) evaluation of risk factors, 3) devising regulatory measures to control and reduce the identified risks, 4) cost-benefit assessment, and 5) recommendations for decision-making. Thus, the ERA is based on scientific criteria (steps 2 and 4) and is context-dependent (steps 3 and 5).

In Finland and the Baltic Sea area, if oil spill risk is proposed to be decreased, the probability of accidents and/or their possible harms should be reduced. Thus, the risk should be managed, and an expected utility defined. The goal is necessary in order to set targets for management. Environmental risk management (ERM) is linked to ERA, since ERM utilizes the results and insights produced by an ERA. ERM is the management of anthropogenic stressors and ecological consequences, which in this thesis is equivalent to the management of oil spill risk. It includes technical and non-technical aspects in various stages from risk prevention, avoidance and reduction to compensation (Assmuth and Hildén 2008). The main idea is to safeguard and enhance the good environmental state, as well as sustain the economic and social benefits of ecosystems (Elliot 2013).

One of the prerequisites for successful ERM is the incorporation of stakeholders during all phases (Pita et al. 2010, Burgman 2005, Lundqvist and Granek 2005). Evident objectives are necessary for improved communication and standardizing the views of stakeholder groups

(Lundqvist and Granek 2005). Stakeholder involvement during the ERM process, for example, in model development and evaluation of the results, increases the transparency and acceptance of the results (Pita et al. 2010, Jakeman et al. 2006). If stakeholders accept the chosen policies and implementation measures, management actions are likely to be more successful (Haapasaari et al. 2012, Chen and Pollino 2012).

1.4. Risk communication

Taking the complexities of ERA and ERM into account, decision-makers are typically in situations where they have to face uncertain information. Even so, it is necessary for decisions and policies to be informed by the best available knowledge provided by science. Highly uncertain knowledge, if unbiased, is better than no knowledge, so long as the scientists provide, and decision-makers use, the knowledge with due regard to its uncertainty (Mulligan 2014). Thus, it is of high importance that probabilities, if they are known or defined, are communicated well and clearly enough (Burgman 2005).

As ERA is at least multidisciplinary and at best interdisciplinary, functioning internal communication between researchers is necessary. Successful communication requires a continuous flow of information and interaction. Scientific communities have their own publication forums and cooperation networks for this. However, communication between scientists and decision-makers can be more difficult than communication within the scientific community (Mulligan 2014). Thus, new tools and courses of action should be developed and tested.

Most questions in ERM are so multilayered and complicated that traditional mechanisms for the communication of knowledge (e.g. reports, scientific papers, policy briefs) are not sufficient to provide information about the uncertainty of spatial and temporal pressures of human activities and their adverse effects on marine ecosystems (Mulligan 2014, Rahikainen et al. 2014). Models and advisory tools that support decisions have the advantage of being able to combine the rules of general processes with the highly specific spatial data describing the condition or state of different species or habitat types (Mulligan 2014).

2. Outline of the thesis

2.1. Aims of the study

In my thesis, I assess what kind of ecological information is required for the environmental risk management and conservation of the marine ecosystem in oil spills. The thesis is split into three thematic sections (Table 1). I seek to achieve the following objectives:

1. To study how Bayesian methods and index frameworks can be used to estimate the harmful impacts of oil on sensitive species.
2. To study how ecological knowledge could be transferred to decision-makers and how risks and knowledge should be communicated.
3. To assess the future approaches and scientific needs for ecological knowledge in oil risk spill management.

The aim of this summary is to form a comprehensive synthesis between the findings of five papers and to propose novel practices for oil spill risk management. Furthermore, this thesis demonstrates my own trajectory as a researcher.

Table 1. Description of the papers.

Thematic section	Paper number	Synopsis of the paper
Methods for gathering knowledge on the impacts of oil on sensitive species	I	We study how extensive loss of Baltic herring eggs and juveniles may occur after an oil spill based on laboratory experiments. Larval stages are noted to be the most sensitive to the effects of oil.
	II	We study how the population of two migratory seabird species may recover after an oil spill using a Bayesian modeling approach. Our models project oil spills to be harmful for the Baltic Sea population of common guillemot and long-tailed duck.
How ecological knowledge could be given to the decision-makers and the suitable policy instruments in coastal conservation	III	We analyze the oil spill risk in the northern Baltic Sea based on the conservation value of threatened species. Our paper demonstrates the species and habitat types most sensitive to oil spills, and also underlines the importance of including several risk factors in spatial risk analyses.
	IV	We develop a map application to support ecosystem-based oil spill response and on-shore cleaning in coastal areas of southern Finland.
Improved approaches to the ecological knowledge required in oil spill risk management	V	We study the impacts of oil spills on threatened coastal species and habitat types, and assess the prioritization of these in oil spill response situations. We use the literature and expert knowledge regarding threatened species and habitat types in the northern Baltic Sea to build an index to estimate the differences in their potential to be safeguarded after an oil spill. We take the conservation value, legislative status, oil-induced loss and recovery potential of species and habitat types, and the effectiveness of response methods into account in the process. Based on our results, this is a suitable method for reducing oil spill risks in the Baltic Sea.
	Thesis discussion	The impacts of the Allee effect and life-history attributes on recovery and environmental management are assessed. In addition, I discuss at which stage policy instruments should be applied, the alternatives being, for example, before possible population collapse or during restoration. This approach has novelty value in research on oil spill risk management and should act as an introduction for new research.

2.2. Taxonomy of the papers

This thesis includes five papers, where we develop methods to assess the environmental impacts of oil spills and the efficiency of management practices to mitigate adverse ecological effects (Table 1). Papers I–III contribute to the assessment of oil spill risk, Papers IV and V provide advice on oil spill risk management, and Papers III and IV demonstrate how the risk can be communicated to decision-makers.

In the first paper, we estimate the sensitivity of the early developmental stages of Baltic herring (*Clupea harengus membras*) to crude oil through laboratory experiments. In this study, newly fertilized embryos, embryos in the late stages of development and hatched larvae of Baltic herring are exposed to a water-accommodated fraction (WAF) made from crude oil and artificial sea water. As the spawning and nursery grounds of Baltic herring are located widely on the shoreline, they occur in the vicinity of oil terminals and shipping lanes, and any oil spillage there could affect recruitment of this commercially important species. Thus, exposures are carried out in levels that correspond to the estimated field conditions after an oil spill during the early development of Baltic herring in spring.

In Paper II, we create two Bayesian networks (BNs) to estimate the impacts of an oil spills on the common guillemot breeding population and the long-tailed duck migrating population in the Gulf of Finland. For the networks, prior information is mainly elicited from experts, since published literature of this nature is not available. In addition, we construct a stochastic simulation model to forecast the population dynamics of common guillemot in two scenarios: with and without an oil spill. BNs are probabilistic directed acyclic graphical models that are considered valuable to researchers and managers due to the probability theory involved. They provide a quantitative method for studying the probabilistic relationships (Aguilera et al. 2011). Explicit accounting for uncertainty adds substantial insight to problems, and the graphical presentation of model structures and probability distributions are useful, especially in risk communication (Uusitalo 2007).

In Paper III, we conduct a spatial risk assessment to study the risk that potential tanker accidents pose to threatened species and habitat types in the northern Baltic Sea. We apply a probabilistic method, which combines three components: a BN describing tanker accidents and uncertainties related to them, probabilistic maps showing the movement of oil, and a database of threatened species and habitat types in the area. Here we utilize the mortality estimates and conservation values of species and habitat types gathered for Paper V.

Paper IV demonstrates the benefits of spatial and on-site accessible tools in oil spill response. First, we analyze and discuss geospatial Web services, and then develop an application to respond to the ecological risk to coastal ecosystems induced by oil spills.

In Paper V, we use literature and expert knowledge regarding the life-history attributes of coastal animals, plants, fungi and lichens in the northern Baltic Sea to build an index to assess the differences in their potential to be safeguarded. Species are selected for

assessment using quantitative criteria and expert evaluation. Data for threatened species are collected in a database, and further narrowed down by choosing only species that face a risk of being exposed to oil. Similarly, coastal habitat types categorized as threatened are included. Based on the gathered information, we develop an index-based method to prioritize the natural values. The index is incorporated into a map application created for the spatial prioritization of oil spill response (IV).

3. Oil spill risk in the northern Baltic Sea

3.1. Case study area

The Baltic Sea and its northern regions are a unique ecosystem. Because of its physical properties (e.g. small water volume, slow water exchange, absence of tides, lack of oxygen in the bottom layers, ice cover during winter, and fragmented archipelago), it is a sensitive sea area (Leppäranta and Myrberg 2009). Low temperatures and the absence of tides slow down the natural weathering processes. Sea ice has implications for the detection and distribution of oil spills, as well as for clean-up. Fragmented archipelago significantly increases the length of shoreline impeding the oil spill response.

The Baltic Sea has been moderately to severely altered by human activities. Anthropogenic pressures that affect the ecosystem include physical damage to the seabed, human-induced disturbance, interference with hydrological processes, contamination by hazardous substances, nutrient and organic matter enrichment, and the introduction of non-indigenous invasive species (Korpinen et al. 2012). Existing stress to the marine ecosystem increases the sensitivity to oil spills. Several species in the Baltic Sea are living at their extreme limits of distribution (Rydén et al. 2003). Species diversity is very low compared to oceans or lakes and rivers because the brackish water creates a harsh living environment. Some subspecies, such as Baltic herring (*Clupea harengus membras*) can only be found in the Baltic Sea. The eastern Baltic Sea is an important migration route for arctic seabird species, and many birds breed on the shoreline during summer including the endangered black-backed gull (*Larus fuscus*). Some of the unique life-forms adapted to the conditions of this area could suffer damage as the result of an oil spill.

This thesis concentrates on oil spill risks in the Gulf of Finland (GOF) and the Finnish Archipelago Sea (AS) in the northern Baltic Sea (Fig. 1). The GOF is the easternmost part of the Baltic Sea and its coastal states are Finland, Russia and Estonia. The western boundary is the line between the Hanko peninsula and the island of Osmussaar. This is more a convention than a physical boundary with the Baltic Sea proper (Alenius et al. 1998). The length is 400 km and the width varies between 48–135 km (Alenius et al. 1998). The coast of southern Finland is fragmented due to the last ice age, but the coast of Estonia is straighter and steeper (Alenius et al. 1998). The sea is shallow, its mean depth being only 37 m. Its volume (1,103 km^2) is only 5% of the Baltic Sea volume (Alenius et al. 1998). Winds and density-driven currents cause water circulation in the GOF: the strongest inflow is at the Estonian coast and the outflow is at the Finnish coast. There are differences in circulation patterns at different depths (Andrejev et al. 2004). No tides can be found, but changes in sea level can be distinct (Alenius et al. 1998). The size of waves varies with the season (Pettersson 2007). The average number of ice days is 40 in the west and 130 in the east (Alenius et al. 1998).

The AS is characterized by a high topographical complexity including 25,000 islands (Hänninen et al. 2000). The fragmented shoreline, in addition to a low average depth of 23 m, creates the prerequisites for a unique environment in Finnish terms. In terms of the number of islands, it is one of the largest in the world. The sea covers a triangular area with the cities of Mariehamn, Uusikaupunki and Hanko at its corners. Its total area is 9,436 km^2 (Hänninen et al. 2000). It is shallow with a mean depth of 23 m. Most of the channels are not navigable for large ships. The number of permanent residents on the islands is approximately 60,000. During summer, the population doubles, as there are thousands of summer cottages in the area.

Figure 1. The Finnish Archipelago Sea and the Gulf of Finland in the northern Baltic Sea[1].

3.2. Maritime oil accidents

Spatial oil accident probability varies with shipping lane location, challenging navigation, and amount of intersecting vessel traffic. In addition, accident probability is affected by time of year, weather conditions, and human factors, such as inadequate communication on the bridge and fatigue of the ship's crew (Lehikoinen et al. 2015, Mazaheri et al. 2015). Oil spills can occur even without ship accidents: spills may be operational, caused by loading/discharging, tank cleaning or bunkering, or even intentional. In the case of an accident, large amounts of oil may be spilled into the sea, which will cause serious harm to the sea environment. The impacts depend on the amount of oil spilled, the chemical characteristics of the oil, the prevailing weather conditions and whether the oil remains in the sea or is washed ashore (Albers 2003). In the sea or on the shoreline, oil is always a harmful substance.

Previous large-scale ship accidents in Europe have demonstrated that oil spills have major and long-lasting adverse impacts on marine ecosystems. M/T *Prestige* sunk off the coast of

[1] Modified from d-maps.com

Spain in 2002, and spilled 60,000 tons of emulsified oil affecting more than 800 km of the northwest Spanish coast (González et al. 2006). Significant reductions in the abundance of some keystone species were noticed after the spill (Sánchez et al. 2006). In addition, sand beaches suffered from the spill and a decrease in their species richness was evident (de la Huz et al. 2005). Three years earlier, in 1999, M/T *Erika* broke apart on the French coast, resulting in a spill of 10,000 tons of oil. Other tankers also took advantage of the *Erika* spill and cleaned out their tanks along the coast. The estimates of killed birds in France vary between 80,000 and 150,000, and 80% of these birds were common guillemots (*Uria aalge*) (Cadiou et al. 2004).

The titles of European Union (EU) maritime legislation packages carry usually the names of accidents (e.g. Erika I-III). The Erika legislative packages are maritime laws intended to improve safety in the shipping industry, and thereby reduce environmental damage to the oceans. The laws enforce, for example, certification requirements for shipping (Directive 2001/105/EC) and establish inspection and verification controls (Directive 2009/16/EC). They also imply greater responsibilities for the shipowners (Directive 2009/20/EC). After the Erika packages, each EU country was required to install appropriate authorities and new or reinforced methods of control (Directive 2002/59/EC).

The above-mentioned naming convention for laws is an indication of inertia in science-policy interaction influencing maritime risk governance in the EU: that is, major accidents and their consequences seem to improve environmental legislation, and not the other way round. It is claimed that assuming the first two Erika packages had been fully implemented and enforced, the *Prestige* oil spill would have not been possible at the time. If the same approach was followed in the fields of aviation or nuclear safety management, we might ask whether society would tolerate it. In nuclear risk management, many actions improving safety are based on model outcomes and risk assessments instead of proven mistakes, demonstrating the strong role of science (Haapasaari et al. 2015).

One of the most well-known oil accidents occurred in Alaska, when M/T *Exxon Valdez* ran aground in 1989 and approximately 42,000 tons of crude oil were spilled into the sea (Brown et al. 1996). The spill contaminated 1,990 km of shoreline in Prince William Sound (Wiedmer et al. 1996). Both the short-term and long-term effects of the spill have been studied comprehensively (Peterson et al. 2003). In all, 250,000 seabirds, 1,800 sea otters, and 302 harbor seals died immediately in the days following the spill. In addition to individuals, the spill was harmful to entire populations: it has been estimated that more than 50% of the local pigeon guillemot (*Cepphus columba*) population was killed, including one colony of over 100,000 birds (National Research Council 2003, Piatt and Ford 1996, Piatt et al. 1990). Larger marine mammals and ducks, meanwhile, suffered long-term negative effects because their prey was contaminated. Commercial fisheries declined when Pacific herring (*Clupea pallasi*) eggs and pink salmon (*Oncorhynchus gorbuscha*) juveniles were exposed to oil, resulting in the collapse of local populations (Brown et al. 1996, Wiedmer et al. 1996).

The *Exxon Valdez* oil spill showed that existing research practices for assessing the ecological risks of oil in marine ecosystems should be changed (Peterson et al. 2003).

Previously, it was assumed that population effects derived mainly from acute mortality. However, in the Alaskan ecosystem, the unexpected persistence of oil and sublethal chronic exposure continued to affect wildlife (Peterson et al. 2003). The ecosystem suffered from cascades of delayed indirect impacts, and this expanded the scope of injury well beyond the initial direct losses, and thereby delayed recovery (Peterson et al. 2003).

In the Baltic Sea, the number of sailing tankers has doubled since 2000 (Brunila and Storgård 2012). The probability of oil accidents has increased during this time¸ even though maritime safety has improved. Oil accident probability is based on expert judgment since no data exist. Collisions and groundings are the most frequent reasons for ship accidents in the Baltic Sea, and these often as the result of technical malfunctions or human errors (Mazaheri et al. 2015, Hänninen and Kujala 2012). The IMO is aiming to reduce the causes of accidents, and its regulations are mostly related to, for example, ship building, crew competence and navigational equipment (Haapasaari et al. 2015). The most important and extensive treaty concerning the safety of merchant ships is the International Convention for the Safety of Life at Sea (SOLAS) (1974). Its first version was adopted in 1914 in response to the RMS *Titanic* disaster. The SOLAS Convention of 1974 specifies, for example, minimum standards for the construction, equipment, and operation of ships, compatible with their safety. To minimize the adverse environmental consequences of shipping, IMO has adopted the International Convention for the Prevention of Pollution from Ships (MARPOL). The aim of MARPOL is to prevent accidental pollution and operational discharges, such as oil spills from ships.

IMO classified the Baltic Sea, excluding the Russian waters, as a Particularly Sensitive Sea Area (PSSA) in 2005. Areas with PSSA status need special protection through action by the IMO because of their significance for recognized ecological and cultural reasons, which may be vulnerable to damage by shipping (MARPOL 73/78). The Baltic Marine Environment Protection Commission (HELCOM) is the governing body of the Baltic Sea Action Plan (BSAP), which aims to restore the good ecological status of the Baltic marine environment by 2021. One of its goals is for maritime activities to be carried out in a safe and environmentally friendly way in the Baltic Sea area. At EU level, the Marine Strategy Framework Directive (MSFD) aims to protect the European marine environment (2008/56/EC). Member states with marine territories are required to introduce a series of measures to achieve and maintain a Good Environmental State (GES) according to key descriptors of environmental status by 2020. Descriptor 8 concentrates on preventing and reducing the amounts of anthropogenic contaminants in seas. This includes the prevention of pollution by ships.

Control systems monitoring shipping movements are working in the northern Baltic Sea. The Gulf of Finland Reporting System (GOFREP) has been shared by Finland, Estonia and Russia since 2003. The GOFREP covers the international waters of the GOF and territorial waters of Finland and Estonia. The system is able to provide information about navigational hazards and weather conditions. An Automatic Identification System (AIS) identifies and locates vessels, and it is mandatory for all vessels over 300 gross tonnage, which, in practice, denotes all cargo vessels (SOLAS 1974). AIS can provide information that can prevent collisions. The double hull requirement is in force for all tankers operating in the GOF

(MARPOL 73/78). According to this requirement, the use of single-hull vessels is not permitted. The newest system is the ENSI system (Enhanced Navigation Support information) where Vessel Traffic Services (VTS) are given access to a ship's route plans.

Over the next few years, only moderate growth is estimated in the transportation of oil and its products (Brunila and Storgård 2012). However, local port capacity is increasing, so rapid growth in tanker traffic is still possible. It is noteworthy that other types of ships can also cause oil accidents, not only tankers. The number of container ships and vessel sizes has been forecasted to increase (Brunila and Storgård 2012, Petersen et al. 2011).

3.3. Environmental consequences of oil spills

Oil can alter coastal and marine ecosystems in various ways. Negative effects on organisms arise from physical smothering, toxicity of aromatic hydrocarbons, habitat modifications or a combination of these (Pezeshki et al. 2000, Morales-Caselles et al. 2006, Alonso-Alvarez et al. 2007, Lecklin et al. 2011, Ihaksi et al. 2011). The consequences of oil exposure include acute death and various kinds of sublethal effects, which decrease the fitness of an individual. In addition to individual- and population-level impacts, the structure and function of entire communities may change (National Research Council 2003, Peterson et al. 2003).

3.3.1. Effects of oil on fish in brackish water

Exploitation is defined as the major threat of marine fish populations; other threats include habitat degradation/loss and climate change impacts (The Zoological Society of London 2015). Oil spills could substantially reduce commercially important fish populations that have already been overexploited (The Zoological Society of London 2015).

Exposure to oil can induce changes in fish metabolism (Cohen et al. 2001). Fish can sense the smell of oil and will escape it (Davis et al. 2002). Furthermore, they are able to avoid drifting oil better in the open sea than in shallow shoreline water, where oil often moves toward them and the concentration of oil in the water becomes higher. The early developmental stages of fish are considered the most vulnerable to the impacts of oil (e.g. Short 2003). Fish spawning and nursery grounds have a high probability of becoming exposed to oil in coastal waters (I, Lecklin et al. 2011). Oil exposure of eggs and larvae may cause mortality or serious developmental disorders, such as malformations in internal organs, decreased reproductive capability, and curving of the notochord (I, Incardona et al. 2005, Brown et al. 1996). Sublethal effects, such as edemas, changes in progeny, neuronal cell death, failed inflation of the swim bladder, and anemia caused by embryonic polycyclic aromatic hydrocarbons (PAH) exposure are possible (Incardona et al. 2009, Carls et al. 2008, Barron et al. 2004, Incardona et al. 2003). Since the oil affects the early development of fish, adverse effects on population dynamics might be detectable only after several years (Rahikainen et al. Submitted).

High molecular weight PAHs are carcinogenic, teratogenic, and toxic to aquatic organisms (Juhasz and Naidu 2000). Temperature, sunlight and salinity alter their physiochemical properties (Goldstein et al. 2011). However, these parameters are often low in the Baltic Sea. Atmospheric photochemical activity which is common in spring and summer, converts volatile hydrocarbons into reactive aldehydes and leads to ozone formation – both of which can cause harm in marine ecosystems (Lee 2003). In addition, oil spills represent a national seafood-safety issue (Goldstein et al. 2011).

3.3.2. Effects of oil on seabirds

Oil spills are affecting birds and their breeding. Seabirds are often the most visible victims of oil accidents – at least in the media. They are highly vulnerable to oil spills as they spend time on the shoreline and forage for food offshore where they easily become exposed to drifting oil (Henkel et al. 2012).

Smothering of plumage by oil is harmful to seabirds, as it may cause sudden death via hypothermia or drowning (Gaston 2004, Kennish 1997). Additional adverse effects arise, for example, from the toxicity of PAHs: birds get oil into their system while preening, drinking oiled sea water, consuming contaminated food or inhaling toxic vapors (Briggs et al. 1996). The effects of ingested oil include anemia, pneumonia, intestinal irritation, kidney damage, altered blood chemistry, decreased growth, impaired osmoregulation, decreased reproduction capability and viability of eggs, abnormal parenting behavior, and decreased growth of the offspring (Albers 2002, Kennish 1997, Briggs et al. 1996, Wood and Heaphy 1991). Some toxic effects may not be evident immediately or may not cause death. These sublethal effects have an impact on birds' physiology and behavior, and can have population-level impacts through diminished health and reproductive fitness (Alonso-Alvarez et al. 2007, Scholz et al. 1992).

Arctic seabirds migrating to their northern breeding areas fly over the GOF. Resting seabirds might alight on an oil slick, which may seem calm to them (II). Oil exposure during migration can cause acute mortality, delayed departure and diminished plumage, and thus, decreased survival (Henkel et al. 2012). During breeding season, offspring may be exposed to oil and this might result in the loss of one year's reproduction. Adults can carry oil to nests and nesting sites can be exposed directly (II, Scholz et al. 1992). Direct exposure of eggs has the greatest potential for damage, and the early stages of incubation are considered to be the most sensitive (Scholz et al. 1992). Oil smothers the pores of an egg shell and this, for example, reduces the hatching success (Finch et al. 2011). Adults that have ingested harmful doses of oil can produce decreased numbers of eggs or cease laying altogether (Scholz et al. 1992). Some adults have been observed to leave their nesting site even after being exposed to small amounts of oil (Scholz et al. 1992).

Predators live in large territories, and thus may avoid oil exposure. However, oiled prey are easy to catch and can attract predators to oiled shorelines. In this way, oil might move to sea eagles (*Haliaeetus albicilla*) via the food web (Peterson 2001).

3.3.3. Effects of oil on shoreline habitat types and coastal communities

After oil accidents, it is likely that the oil cannot be recovered at sea, and it drifts ashore. Oil affects the shore vegetation as well as animals. The adverse effects of oil depend on the geology of the shoreline: open calcareous rock outcrops are not that sensitive to oil because their vegetation is often meagre, oil cannot absorb into the rock and the waves clean the shore habitat naturally. On the other hand, lichen living on rock may recover slowly from oil exposure and manual cleaning of rock outcrops is time-consuming and labor-intensive.

On coastal sand beaches and seashore meadows, oil can penetrate into the beach sediments. Thus, oil persists for decades within anoxic soft bottoms where oil degradation is inhibited, and the adverse effects of oil residuals on ecosystem will become long-term (Boufadel et al. 2010, Culbertson et al. 2008, de la Huz et al. 2005). Sand beaches are the most sensitive to the adverse effects of oil in the northern Baltic Sea (III) and they often act as habitats for threatened plants and animals specialized for beaches. Contamination of beach ecosystems has the potential to cause severe environmental as well as economic consequences since the ecosystem services degrade (Peterson et al. 2012, Kostka et al. 2011). Coastal habitat types in the outer archipelago experience substantial oxygenation by wave action.

Oil affects shore vegetation by covering the plants. This inhibits absorption of sunlight that is necessary for photosynthesis and growing (Pezeshki et al. 2000). Oil may physically prevent or disturb plant gas-exchange and cause stress (Pezeshki et al. 2000). Toxic compounds of oil may, for example, inhibit root growth, lower nutrient and water absorption ability, and reduce leaf numbers and growth (Lopes et al. 2009, Zhang et al. 2007). However, coastal and marine macrophytes are able to emerge from the propagule bank (e.g. seeds or roots) after disturbance (Nishihiro et al. 2003, Ozimek 2005, Pahl et al. 2003).

The most vulnerable of the shoreline animals are those whose territory is small and located near the water's edge. The threatened beetle *Aegialia arenaria* lives on coastal sand beaches and dunes. When the shore becomes polluted, the poorly moving beetle will not have time to escape. Even if its habitat is cleaned properly and its food plants recover, the recovery of the beetle is uncertain, since the local occurrence will be lost (V). Adults of mobile species, such as butterflies are able to escape oiled shorelines, and thus avoid oil-induced harm. If species breed on the shore, their offspring are more sensitive to the harmful effects of oil. If the accident occurs during breeding season and oil drifts ashore, one year's cohort might be lost. Occurrences of some threatened insects, such as the beetle *Psylliodes marcida*, are linked to the occurrences of certain shore plants. These interactions need to be taken into account when oil spill risk is assessed and managed.

The vulnerable environment can continue underwater: eelgrass (*Zostera marina*) meadows occur on sandy seabed. Eelgrass is important in coastal ecosystems because it helps to form a habitat for other species (Ort et al. 2014). Underwater nature is almost impossible to clean using current methods. Thus, drifting of oil into coastal and marine species and habitat types should be prevented. As on shore, oil may smother aquatic plant communities. The effects of oil on eelgrass have been noticed to vary from minor to severe, and depend on, among other things, depth, oil type and local conditions (Dean et al. 1998). Bladder wrack (*Fucus vesiculosus*) tolerates short oil exposure rather well (Ryzhik 2012). The reason for this might be that oil does not adhere easily to the cell wall. However, an oil layer on an alga can cause waves to detach the alga from the seabed due to its increased weight. Entire coastal and marine habitat types can be exposed to oil and their vulnerability should be considered in relation to their recovery potential. In the AS and the GOF, eelgrass meadows and

Charophyte communities are vulnerable to oil as they grow on soft bottoms and are sensitive to sedimented oil (V). Adverse sublethal effects of aquatic plants may be delayed, and for that reason, monitoring after accidents is necessary.

Species living on plants have been noted to disappear rapidly from oiled seabeds (Peterson et al. 2003). Acute short-term mortality from oil deposited on shore and shallow seabeds or through smothering accounts some of the important losses of shoreline plants and invertebrates (Peterson et al. 2003). Of the aquatic invertebrates, crustaceans and mobile herbivores, *Amphipoda* in particular are sensitive to oil spills (Lecklin et al. 2011). *Mollusca* are able to avoid oil for a short period of time by closing their shell. If lightly polluted areas are cleaned, *Mollusca* communities may recover, as they are able to filter harmful substances (Moles 1998). Invertebrates are prey for several fish species and their oiling or disappearing may diminish the nutrient available for fish.

3.4. Preparedness planning and oil spill response

In oil spills, risk management can be divided into a) preparedness planning, that is, the identification of risks and vulnerabilities and preparatory risk management before any spill has happened, and b) the operational phase, including oil spill response and clean-up during and after spill. When an oil spill is detected, response officers face difficult decisions on what actions to choose or reject (Peterson et al. 2012). Assessing trade-offs between measures is made troublesome by imperfect knowledge (Anastas et al 2010).

HELCOM advises that the adverse effects of oil on wildlife should be prevented and minimized by protecting sensitive areas (Helsinki Commission 2010). As a best practice, they suggest using vulnerability maps that include the distribution of wildlife along the shoreline. The aim of ecological sensitivity maps is to highlight the locations of the highest concentrations of valued elements, the areas of most sensitive life stages, and the occurrences of the most vulnerable species in order to emphasize their need for protection (e.g. Tortell 1992, Jensen et al. 1998). Furthermore, habitat types can be ranked based on their vulnerability to oil contamination and ease of clean-up. Same approach is applied in the Baltic Sea area (Leiger et al. 2012). Often, however, these sensitive areas are too large and difficult to safeguard. In addition, sensitivity maps based only on ecological values may give high prioritization values for areas that are inhabited by species that cannot be safeguarded with available oil spill response equipment. Nevertheless, vulnerability maps, such as Environmental Sensitivity Index (ESI) maps (Leiger et al. 2012, Jensen et al. 1998) are good for planners to use before a spill happens in order to scan vulnerable locations, establish protection areas and prepare regional clean-up strategies.

Game et al. (2013) describe six common mistakes in conservation priority setting, one of them being not acknowledging the risk of failure in conservation actions. Oil retention booms are highly susceptible to waves leading to their damage and transport to shore and deposition on a sensitive habitat type. This renders them non-functional and even harmful to wildlife movement (Peterson et al. 2012). The probability that a conservation action will fail affects the expected costs and benefits of the selected action, and this should influence the decision on whether an action is implemented (Bottrill et al. 2008). However, the risk of failure may often be absent from conservation prioritization schemes (Game et al. 2013). To overcome this issue, Ihaksi et al. (2011) combine the expected benefits of oil spill response measures with the ecological sensitivity of species.

In addition to ecological attributes, the spatial aspects related to accidents and the spreading of oil should be taken into account when assessing the environmental risks related to oil spills (Jolma et al. 2014). Jolma et al. (2014) have developed a method for assessing the spatially distributed ecological risk posed by oil accidents in the GOF. By integrating a BN with geospatial information, the method enables probabilistic oil spill scenarios to be computed and the consequent ecological risks to be evaluated. The visual risk map shows the risk level for each grid cell across the study area.

A worst case scenario which oil spill response in the GOF is based on is a spill of 30,000 tons of crude oil caused by a rupture of two large cargo tanks. In the AS, the corresponding spill is estimated at 15,000 tons (Hietala and Lampela 2007). In a scale of the Baltic Sea, these estimates are considered high. When the scenarios were assessed and the spill sizes determined, they were considered to act as a policy instrument. After that, the level of Finnish marine pollution response has improved. Furthermore, the estimates have a role in risk communication. In Canada similar magnitudes have been used with the highest oil spill categories estimated as spills of 16,000–32,000 tons and spills over 32, 000 tons (Marty 2014). In a ship accident leading to an oil spill, the principal aim of the oil spill response authorities, after lifesaving, is to stop the spill and recover as much as possible offshore. As the GOF is a narrow sea area, spilled oil will most likely drift to shore if not recovered in the open sea. HELCOM recommends that if the oil slick spreads toward the shoreline, valuable locations should be protected using, for example, oil retention booms and coastal protection sheets (Helsinki Commission 2001). The use of dispersants is not a preferred option due to uncertainties regarding their effectiveness and impacts on the ecosystem (Helle et al. 2011). Booms can also be used to direct the slick to less valuable areas or areas that are easier to clean.

This means that sites of less value that are easier to clean have to be specified. The information should be given to the oil spill response authorities and it should be based on scientific knowledge. Particularly in large-scale accidents, existing oil spill response resources (equipment, time and personnel) may not be enough to ensure that everything will be protected (Wirtz et al. 2007). In such events, environmental values have to be prioritized in a cost-effective manner (Ihaksi et al. 2011). Ihaksi et al. (2011) suggest that there is a need in decision-making on the hierarchy of vulnerable areas to be safeguarded after an oil spill to compare the biological impacts and recovery potential, the relative value of a population among other impacted populations, and the technical possibilities of protecting the population in question.

It is noteworthy that clean-up efforts can be futile (Kostka et al. 2011) or damaging (Peterson et al. 2003). A blowout of the *Deepwater Horizon* well resulted in the world’s largest accidental oil spill in April 2010 in the Gulf of Mexico (Kostka et al. 2011). Although the cleaning up after accident was powerful, a substantial portion of oil persisted in coastal ecosystems, especially in benthic areas (Kostka et al. 2011). In addition, the removal of penetrated oil by beach excavation may kill intertidal invertebrates, disturb sensitive shorebirds by vehicular traffic, and cause running over seabird chicks (Peterson et al. 2012). However, clean-up activities are often not as urgent as rapid oil spill response, and therefore time should be allocated to careful planning. An exception occurs if an accident happens during spring before the bird breeding or fish spawning season. Then the cleaning of known colony islets or nursing grounds should be hastened.

4. Sources of ecological information

4.1. Databases

Several accident-triggered monitoring programs demonstrate the need for *a priori* information on ecological values (Wirtz et al. 2007). The debate about the effects of the *Exxon Valdez* oil spill appears to be driven by a lack of baseline pre-spill data (Henkel et al. 2012). Furthermore, the implementation of a legislatively mandated natural resource damage assessment for the *Deepwater Horizon* drilling rig revealed serious gaps in the baseline information on deep-sea communities, their functioning and ecotoxicological vulnerability (Peterson et al. 2012). ERAs that are used to quantify the intensity and duration of impacts of oil spills on marine ecosystems require the establishment of recovery end points that represent the return of a population or a habitat type to a defined state. Without sufficient baseline data, endpoints are difficult to estimate (Henkel et al. 2012). The goal to define recovery or restoration targets can be generalized to most coastal regions around the world, where the use of natural resource conflicts or imminent damage risks call for management advice, marine spatial planning or pre-emptive stakeholder involvement (Wirtz et al. 2007).

The recovery of the ecosystem and its biological resources to the pre-spill baseline levels is not the target after an oil spill. Restoration should create resource levels that would have existed in the spill-affected area in the absence of spill. These levels may be quite different from those that prevailed just before the oil spill – effects of other human actions and climate change would have continued to operate and modify the ecosystem. This concept of targeting the system status that would have prevailed had there been no oil spill can be noticed in ERAs and any continuous human pressure that increase the uncertainty of recovery taken into account. Other unexpected causes (e.g. toxic algal blooms) of precipitous declines in biological resources are, however, difficult to predict in practice.

Relevant information and knowledge about marine ecosystems need to be collected and analyzed in advance, and then taken into account for the adequate management of oil spill risk (Aizpurua et al. 2015, Lundqvist and Granek 2005). Existing ecological data must be integrated into problem-adapted frameworks which account for ecological sensitivity and other oil spill-related characteristics of coastal species and habitat types (Wirtz et al. 2007). ERM needs both spatial and non-spatial data (Table. 2). In Paper IV, for example, spatial data include occurrences of species and habitat types, estimations of ecological values in different areas, and, of course, background maps (e.g. satellite imagery or topographical maps). Non-spatial data here consist of information about the sensitivity of threatened species and habitat types to oil spills.

Table 2. Required ecological information for marine conservation in oil spill risk management.

Process	Paper number	Spatial data	Non-spatial data
ERA	I–III	Occurrences of species and habitat types, spatial estimations of ecological values, oil drifting trajectories	Estimated conservation value, loss, and recovery potential
ERM	IV–V	Occurrences of species and habitat types, spatial estimations of ecological values	Estimated conservation value, loss, recovery potential, and protection effectiveness
Risk commu-nication	III–IV	Spatial estimations of ecological values	Conservation value, estimated loss of ecosystem

In Finland, the source for collecting geographical data on the occurrence of coastal species is the national Hertta environmental information system[2], and for marine biotopes, species and communities, the VELMU map service[3]. A new national and open data center for biodiversity information, the Finnish Biodiversity Information Facility (FinBIF) will be set up by the Finnish Museum of Natural History and the Finnish Environment Institute by 2017. FinBIF aims to support species-related research, management and education. Information about Important Birds Areas (IBAs) is compiled into the Tiira information system maintained by BirdLife Finland which is an environmental Non-Governmental Organization (NGO). Data about economically important fish stocks and fisheries is collected and provided by the Natural Resources Institute Finland. However, it is noteworthy that occurrence data on threatened species is concealed from the public by the Finnish Environment Institute and spatial data on water depth by the Finnish Defense Forces. Causing damage to a threatened species or destroying a habitat important to the survival of them is prohibited. Thus, the occurrence data are confidential to avoid the disturbance or the removal by collectors. Bathymetry data are classified information due to national security. In the risk assessments and protection of threatened species, high spatial accuracy is needed. The information on threatened species and habitat types is given to the researchers and response authorities in preparedness planning and oil spill response. For risk communication, information with lower accuracy is often sufficient. Other data should be open for research and management purposes, even though this may not always be the case.

4.2. Laboratory tests

If conservation goals are at the ecosystem level, the single-species limitation can be ameliorated by selecting a limited number of species that are representative of the ecological processes or sensitive to the disturbance of interest (Acevedo et al. 2015). Ecotoxicology

[2] syke.fi/avoindata
[3] paikkatieto.ymparisto.fi/velmu

provides a basis for assessing whether chemicals are likely to have adverse effects on species (Calow and Forbes 2003). The goal is to be able to predict the effects of pollution in order to identify the most efficient action to prevent any detrimental effects (National Research Council 2003). Measures of impacts include quantitative biological responses, such as toxic effects on survival, and estimates can be created from laboratory tests. These represent the sensitivities of species to toxic substances and act as surrogates for other ecosystem elements (Burgman 2005).

Single-species studies often have the advantage of being focused, but they produce an incomplete picture of the total effects on the ecosystem. Extrapolation is a process of estimating the value of a variable based on its relationship with another variable. The results of laboratory tests on a small group of organisms are used to infer the effects that might be expected on a much larger group, assuming that a few species provide a useful guide to the sensitivities of a much larger range of taxa (Burgman 2005). However, estimations that invoke generalizations about ecology are likely to be uncertain and exposed to additional assumptions (Boersma et al. 2001). Moreover, when concerning recovery potential, criteria must be linked to species biology to ensure that recovery estimates are appropriately suited to threatened species (V, Boersma et al. 2001).

Natural selection favors life-history traits that improve a species' chances of survival and reproductive success. There are trade-offs between survival and reproductive traits, such as the frequency of reproduction, the number of offspring, and the amount of parental care (Reece et al. 2014). When oil affects only one life-history stage, it may not have an effect on population recovery, even given substantial mortality. At the same time, small alterations in life-history parameters may have significant effects on the population's resistance against oil spills (Burgman 2005). That is, equal levels of mortality and reductions in fecundity may have different impacts on threatened species with different life-history traits (Stark et al. 2003). Thus, studies should be conducted to determine the life-history strategies of species affected by oil (Cadiou et al. 2004). The adverse effects of oil exposure could be estimated by propagating the impact through a population model (Burgman 2005).

The downsides of laboratory tests are the high cost and poor availability of facilities. There is also a matter of ethics when using laboratory animals in toxicity tests. Thus, the learning from laboratory tests should be maximal in order to reduce the number of animals used in tests.

4.3. Previous studies as a source of knowledge

As part of the knowledge elicitation process, a substantial amount of scientific and gray literature can be reviewed (Lecklin et al. 2011). Assembling prior probabilities for modeling from the existing literature is noted to be a useful approach in risk management (Kuikka et al. 2014).

Meta-analysis is based on the idea that there is a common truth behind all studies, and individual studies aim to measure this with a defined uncertainty. The possible biological effects of oil accidents can be estimated using a statistical approach to combine the results from databases and other published studies. Uncertainty can be reduced by interpreting existing data more efficiently through utilizing biological knowledge to a higher extent instead of collecting new data (Pulkkinen et al. 2011). In addition, combining information from different sources within a hierarchical meta-analysis can be more efficient when known parameter correlations are taken into account instead of treating them as conditionally independent (Pulkkinen et al. 2011). However, as each oil spill represents a context-specific situation, it might be difficult to extrapolate findings from studies initiated after different spills, especially regarding long-term effects on resident species and habitat types (Lecklin et al. 2011). Moreover, different study designs, approaches and survey perspectives, as well as varying definitions of loss and recovery, can produce diverse conclusions concerning the consequences of an oil spill (Lecklin et al. 2011), and thus increase uncertainty.

4.4. Expert knowledge

Experts can offer relevant information for decision-making. Structured expert judgment can be integrated into modeling approaches to improve predictions (Krueger et al. 2012). This is an important resource that may even provide completely new information for a particular case (Yamada et al. 2003). In some studies, as also in Papers II, III and V, it can be particularly valuable, since no relevant published data can be found. Expert elicitation is an opportunity to obtain high-quality and structured information on species distribution with a low cost (Cerquera et al. 2013). Experts have achieved high knowledge on a particular subject through their work and life experiences, and are defined by their qualifications, track record and professional status (Kuhnert et al. 2010, Burgman et al. 2011). Nevertheless, there remains uncertainty regarding the reliability of expert elicitation (Yamada et al. 2003).

Human estimators may be prone to overconfidence, which means that they might give estimates that are too near zero or one (Burgman 2005, Morgan and Henrion 1990). However, the opposite phenomenon was noticed in Paper II, where most of the experts were rather underconfident in their own beliefs. When experts consider an event improbable, they may provide overly low estimates (Van der Laag et al. 2002). This can be the situation with unusual cases, such as oil spills or the extinction of threatened species. Furthermore, experts have a tendency to anchor their judgments to initial estimates, especially if they can defer their opinions to people they believe have higher authority (Burgman 2005). In order to diminish these biases, researchers should follow structured guidelines (Kuhnert et al. 2010). In addition, a careful pre-elicitation analysis of expert availability should be carried out in order to obtain the highest-quality information (Martin et al. 2012). The objective of good expert elicitation is to eradicate connotations and irrationalities, and to structure the process in such a way that the expert can evaluate their knowledge and experience in a rational manner (O'Hagan et al. 2006).

As managers often believe that models do not result in better decisions than those supported by the subjective opinions of experts (Addison et al. 2013), expert-informed modeling can contribute to bridging the gap between researchers and decision-makers (Aizpurua et al. 2015). Expert knowledge can be incorporated into models at different stages of the procedure (Pearce et al. 2001).

4.5. Modeling

Environmental modeling aims at knowledge generation and communication to enhance the understanding of phenomena (Jakeman et al. 2006). The population impacts of oil contamination and the rate of recovery can be estimated using environmental models. Data for these models can be gathered and derived from field surveys, laboratory tests and the literature. In addition, expert knowledge can provide valuable information for modeling (Kuhnert et al. 2010). Modeling enables researchers to estimate the potential impacts of oil spills and utilize those results in ERAs, including cost-benefit and decision analyses (Thorbek et al. 2009, French McCay et al. 2004). This will lead to informed decisions in strategic planning or operational management (Jakeman et al. 2006). As a general rule, a developed model should be simple for addressing the question of interest (Thorbek et al. 2009). For more efficient use of environmental models in ERA, increased training in model use and interpretation and the active involvement of stakeholders is necessary.

If ERA models are applied to other areas than originally developed, researchers and managers must pay attention to the suitability of the model parameters and validated bounds (II). In Australia, an ERA model developed and validated under European farming conditions was used to assess the runoff impacts on the Great Barrier Reef from sugarcane farming in the catchment area (Holmes 2014). It was noticed years later that ERA was insufficient and the toxicological endpoints used in the modeling were inadequate. Consequently, a commonly used pesticide, diuron, posed a risk to marine ecosystems (Holmes et al. 2014).

5. Methods to support informed decision-making

5.1. Risk assessment using Bayesian networks

In Papers I, II and IV, we carried out risk assessments to estimate the mortality of Baltic herring eggs and juveniles, the decrease in migrating and breeding seabird populations, and the loss of threatened coastal species and habitat types caused by an oil spill in the northern Baltic Sea. In Papers II and IV, we used BNs as a modeling tool.

BNs originate in artificial intelligence studies, but, in addition to ecological and environmental research (Chen and Pollino 2012, Aguileira et al. 2011, Varis and Kuikka 1999), they are nowadays applied in various disciplines of research, such as economics (Neil et al. 2005), social sciences (Haapasaari and Karjalainen 2010), medicine (Forsberg et al. 2011) and engineering (Goerlandt and Montewka. 2015).

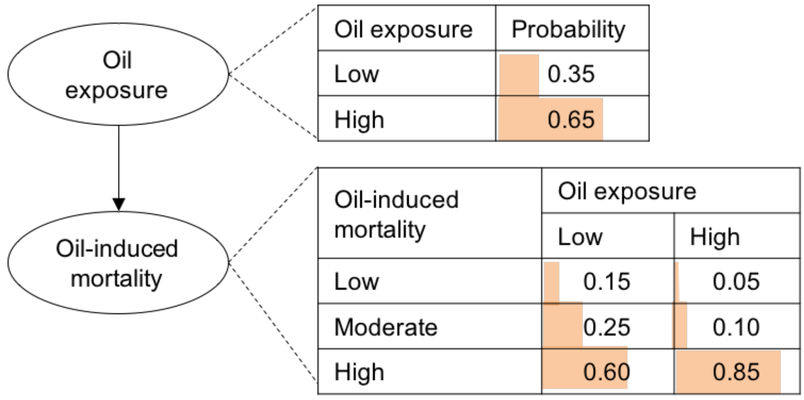

Figure 2. A simple example of a directed acyclic graph: the nodes correspond to variables and the arrow link directly dependent variables.

BN is an explicit description of the direct dependencies among a set of variables, in the form of a directed acyclic graph (DAG) and a set of conditional probability tables (CPT) (Fenton and Neil 2013). DAGs consist of a set of nodes and arrows: the nodes correspond to variables and the arrows link directly dependent variables (Fig. 2). Links between arrows encode assumptions that there are direct causal or influential dependencies between variables. Each node has an associated CPT that describes the probability distribution of a given variable (Fenton and Neil 2013). It is noteworthy that the graphical feature of a BN also tells us which variables are not linked, and hence captures assumptions about which pairs of variables are not dependent (Fenton and Neil 2013). The graphical structure makes it easy to discuss the model assumptions with stakeholders.

BNs can be generalized to influence diagrams (IDs) by including decision and/or utility functions. IDs can model and solve not only probabilistic inference problems, but also decision-making problems following expected utility criterion (Fenton and Neil 2013). Expected utility theory deals with the analysis of choices among risk assessments with uncertain outcomes.

In BNs, the probability distributions can be expressed in discrete form and solved analytically (II, III), and employing simulation-based Monte Carlo techniques, the distributions can be estimated by generating samples from these by simulation (II). Both of these methods share the idea of conditional dependence between variables and the updating of knowledge (Fig. 3).

Spatial aspects can be introduced to risk assessments by using BNs in geographical analysis (Jolma et al. 2014, Stelzenmüller et al. 2010). In Paper III, we combined 1) an oil spill simulation model, 2) an occurrence database of threatened species and habitat types and, 3) estimated losses and conservation values of species and habitat types, in order to estimate the spatially distributed risk of oil shipping.

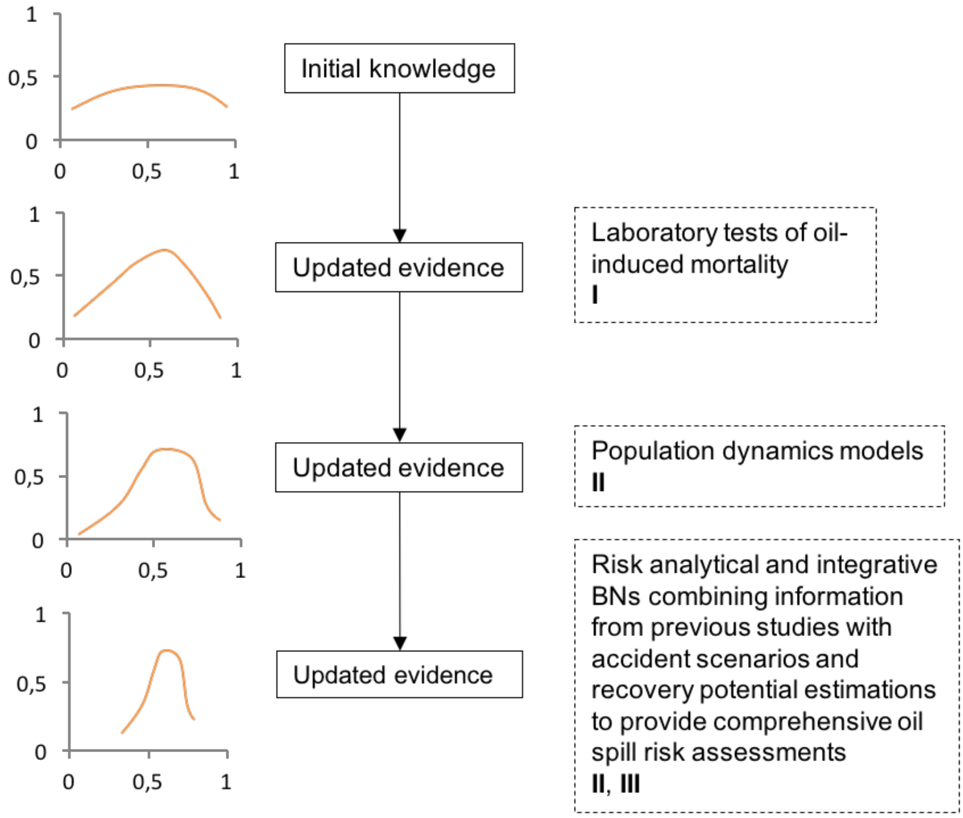

Figure 3. Updating prior knowledge with new evidence to achieve a better posterior understanding of an oil spill risk (Bayesian and hermeneutic theory). As knowledge accumulates, the probability distribution narrows (modified from Haapasaari 2012). The graphs are only illustrative, but the vertical axis (y axis) could describe the estimated probability and the horizontal axis (x axis) could describe the recovery of a population five years after an oil spill.

Bayesian methods can improve the understanding of oil spill risks if there are limitations in data availability or experiments cannot be repeated, but management decisions are expected to be made based on this deficient information. Uncertainty analysis, such as sensitivity analysis and information-gap methodologies, are suitable to study epistemic uncertainties (Regan et al. 2005). Sensitivity analyses study how the uncertainty in the model output can be apportioned to different sources of uncertainty in its inputs. Information-gap theory addresses the question how much uncertainty can be tolerated before a decision would change (Regan et al. 2005). The situation in oil spill risk assessment is similar to fisheries stock assessment, where the amount of available information from a specific stock might be limited and the resources to gather new data are insufficient (Pulkkinen et al. 2011). BNs can be used effectively to represent uncertainty in understanding and variability of ecosystem responses (McCann et al. 2006).

5.2. Map applications

Quantifying and mapping local-scale stressors in a standardized, comparable manner offers a powerful means of assessing both the spatial pattern and the temporal change of human pressures, such as oil spills, as well as their total impact on marine ecosystems across highly variable geographies (Halpern et al. 2014, Halpern et al. 2008).

Spatial decision support systems (SDSS) are computer-based solutions to support complex decision-making while solving spatial problems. An SDSS tool is built on following components: 1) database management system for accessing spatial and non-spatial internal and external data, information and knowledge, 2) modeling functions and 3) user interface designs for interactive queries, graphical display and reporting (Mulligan 2014, Brimicombe 2010). Coupling SDSS with a Geographical Information System (GIS) and environmental models is a successful strategy for exploring the decision space of an application domain (Brimicombe 2010). In addition, when creating Web-based decision tools, the participatory processes incorporated include equal access to all sectors and parties involved, and a high level of trust and transparency of decisions that occur (Mulligan 2014, Brimicombe 2010). Different GIS-based applications may work as tools for informative policy instruments, and provide information to managers on vulnerable species and habitat types, and the risks threatening them (III–IV).

In oil spill risk management, ecological data and knowledge should be integrated into tools that facilitate strategic and tactical decision-making. Often the decisions required are of spatial nature, such as defining sites with a need of safeguarding. For adequate contingency planning and efficient spill response operations, the spatial information must be available, usable and accurate. In Paper IV, we developed a spatial on-site accessible tool for real-time decision-making. We were able to produce a dynamic way in which maps are reproduced when background data change spatially or is updated with new information. The map application was designed to help decision-makers in choosing appropriate methods for oil spill response and on-shore cleaning.

Maps present complex spatial information in a manner that is easily interpreted. However, maps can be misleading for the same reasons as they are useful. They can be misinforming because they might imply that there is more information than actually exists. An unoccupied map view may indicate that no species or habitat types occur in that particular area. However, this might only be because no monitoring surveys have been carried out there. Furthermore, it is noteworthy that maps present what they have been designed to present. Models and maps are simplifications, and thus cannot represent reality completely.

5.3. Conservation prioritization scheme

Spatial conservation prioritization tools provide information on conservation value for ERM. Different studies propose alternative criteria for identifying conservation priorities based on ecology (Leiger et al. 2012, Ihaksi et al. 2011, Tortell 1992, Jensen et al. 1998). For oil spill risk management, it is suggested that protecting, for example, coastal national parks, is advisable in order to reduce the adverse effects of oil accidents on marine ecosystems (Leiger et al. 2012, Jensen et al. 1998). However, these are too large and difficult to safeguard, and this can make oil spill response less efficient.

We developed a method in Paper V to prioritize threatened species and habitat types in the AS and the GOF in order to give recommendations for oil spill response. The developed prioritization scheme acknowledges the conservation value, legislative status and oil-induced loss of populations and habitat types, their recovery potential, and the species- and habitat type-specific effectiveness of retention booms and protection sheets. The difference between the populations and habitat types five years after an accident with and without protection actions is examined, and the higher the difference, the more worthwhile the management actions are. In marine conservation, if an ecosystem is likely to persist without a particular action, then the action will have a low ecological benefit (Bottrill et al. 2008).

Our conservation prioritization scheme is similar to the conservation triage approach (Bottrill et al. 2008). The latter highlights the need to prioritize the allocation of limited resources to maximize conservation returns, relative to goals, under a constrained management budget. Conservation triage has been argued to promote defeatism when an ecosystem is deemed too difficult to save (Marris 2007). In addition, it is doubtful that the approach will result in the protection of only moderately threatened biodiversity assets (Mittmeier et al. 2005). However, these arguments fail to acknowledge the fact that the resources for marine conservation and oil spill risk management are often limited. In oil spill response, if an oil slick spreads towards shoreline, coastal ecosystems should be protected. However, in a large-scale accident, response authorities might face a situation where the total length of oil retention booms is limited, and every species and habitat type cannot be safeguarded. Thus, the protection order must be decided. Since the operational decision-makers are not ecologists, evaluation process must be assessed and presented in advance, as we did in Paper V. Based on our pre-defined goals, efficient resource allocation will be achieved. If the conservation prioritization scheme presented in Paper V would not be followed, the worst case scenario could be that an unsuccessful protection of common species would be attempted, and the restoration of threatened species having a directive status (i.e. the EU Habitats Directive Annex II and Annex IV species or the Bird Directive Annex I species) would be funded from the state budget after an oil spill.

The process of triage is implicitly applied on a daily basis by decision-makers (Bottrill et al. 2008, Marris 2007). If they are not transparent, they are likely to make inefficient choices (Bottrill et al. 2008). To support smart and explicit decision-making, ecological evidence must be provided for decision-makers in oil spill risk management through the use of advisory tools.

5.4. Use of knowledge in practical work: Metsähallitus Finnish Parks and Wildlife as a case study

There is an acknowledged need for marine conservation scientists and their professional associations to become more involved in advocating for science-based management and policy decisions (Rose and Parsons 2015). Problems may arise when scientists who enter the management arena fail to understand the differing priorities between science and policy. Marine scientists, for instance, seek biodiversity protection, and managers incorporate various factors in their decisions including human livelihood improvement (Rose and Parsons 2015, Minteer and Miller 2011). This imposes heavy pressure on communication, and can lead to misunderstandings and the misinterpretation of science. To avoid this, if possible, scientists should find more participatory ways in which to integrate their knowledge into the decision-making process.

During my thesis research, I worked as a planning officer for Metsähallitus Parks and Wildlife Finland (Metsähallitus) for two months, which was a collaboration between Metsähallitus and the Fisheries and Environmental Management Group (FEM Group). The aim of this was to 1) assess a suitable role for Metsähallitus in Finnish contingency planning, and 2) integrate the information and knowledge of oil spills risk provided by our research group and research network (FEM Group and Kotka Maritime Research Centre, respectively).

Metsähallitus is part of a state-owned enterprise, and is responsible for public administration services which include nature conservation and the management planning of protected areas. Metsähallitus' personnel consist mainly of conservation biologists who have much knowledge of coastal and terrestrial ecosystems and regional nature values, and have great experience in field work and data sampling. Metsähallitus has suitable resources and equipment for activities to support oil spill response. However, they have not been officially included in the national oil spill response preparedness plans in Finland.

6. Specific studies supporting the thesis

This section of the thesis presents the main study results and outputs of specific case studies, and supports the methodologies described in the previous chapter.

6.1. Impacts of oil spills on the early stages of Baltic herring

In Paper I, we estimated the kind of adverse effects oil exposure can cause on the early developmental stages of Baltic herring in the GOF. The larval stages suffer the harmful impacts of oil most, and if oil exposure occurs during the hatching period, these larvae are more sensitive to the impacts of oil than larvae exposed after hatching. Embryos in the late embryonic stage are the most tolerant (LC_{50} was 44% of the WAF solution). During the exposure, some malformations, such as curving of notochord, were noticed (Fig. 4).

Figure 4. Hatched larvae of Baltic herring which have been exposed to WAF during the embryonic stage. Oil exposure generates curving of the notochord, and here, the notochord has not straightened, which causes mortality.

It is estimated that the mortality of early stage larvae of North Sea herring (*Clupea harengus*) has an effect on the population dynamics of the stock (Fässler et al. 2011). When providing an integrative management advice which combines stock assessments with oil spill risk assessments (Rahikainen et al. 2014), the estimates of impacts of oil spills on the early stages of Baltic herring are required.

6.2. Impacts of oil spills on migrating seabirds

In the second paper, we applied a probabilistic method to assess the harmful effects that oil tanker accidents pose to migrating seabird populations. Based on modeling, oil spills can be harmful to the common guillemot and long-tailed duck populations in the GOF.

We demonstrated that BNs are functioning methods for estimating the impacts of an oil spill on seabird populations. Networks can be based on expert knowledge and completed with statistical data. The availability of experts around the GOF is good and their knowledge can be utilized easily. However, when using experts, the interviewing methods are remarkably important. In this thesis, the expert elicitation was conducted as a face-to-face interaction between the expert and the interviewer. This required the commitment of human resources. In addition, a questionnaire was used to ensure that each expert received the same background information and treatment. Furthermore, these in-depth interviews allowed the development of defined questions to which answers were written down.

These kinds of BNs can make use of the results from laboratory tests (I) and are easily connected to other management decision tools (e.g. Lehikoinen et al. 2015) to estimate the comprehensive oil spill risk and its management using quantitative methods.

6.3. Recovery potential of threatened species and habitat types

The recovery potential of species after an oil spill is an outcome of acute loss, recolonization efficiency, and reproductive capacity when a certain proportion of the species population has been lost due to oil exposure and subsequent mortality (II, V). In Paper V, we determined oil-induced mortality for all species as the probability of an individual dying after an oil spill. The estimates are based on expert knowledge and the literature, and both physiological factors and structural aspects are taken into account. From the mortality index, we derived an estimate for population loss. We defined recovery potential values using the species-specific estimations of fertility and recolonization (V). We described the recovery potential of coastal habitat types as the probability that a habitat type recovers to the state where it would have been without an oil spill.

Comprehensive assessments of recovery potential of threatened species and habitat types play a significant role in marine conservation and oil spill risk management. Quantified recovery values, including estimations of loss, give useful ecological information about the adverse effects of oil spills for preparedness planning.

6.4. Use of ecological knowledge in oil spills and exercises

In Papers IV and V, we developed a map application (Fig. 5) and a conservation prioritization scheme, respectively, to act as advisory tools in the spatial allocation of oil retention booms and coastal protection sheets. In an accident or training exercise situation, the spatial

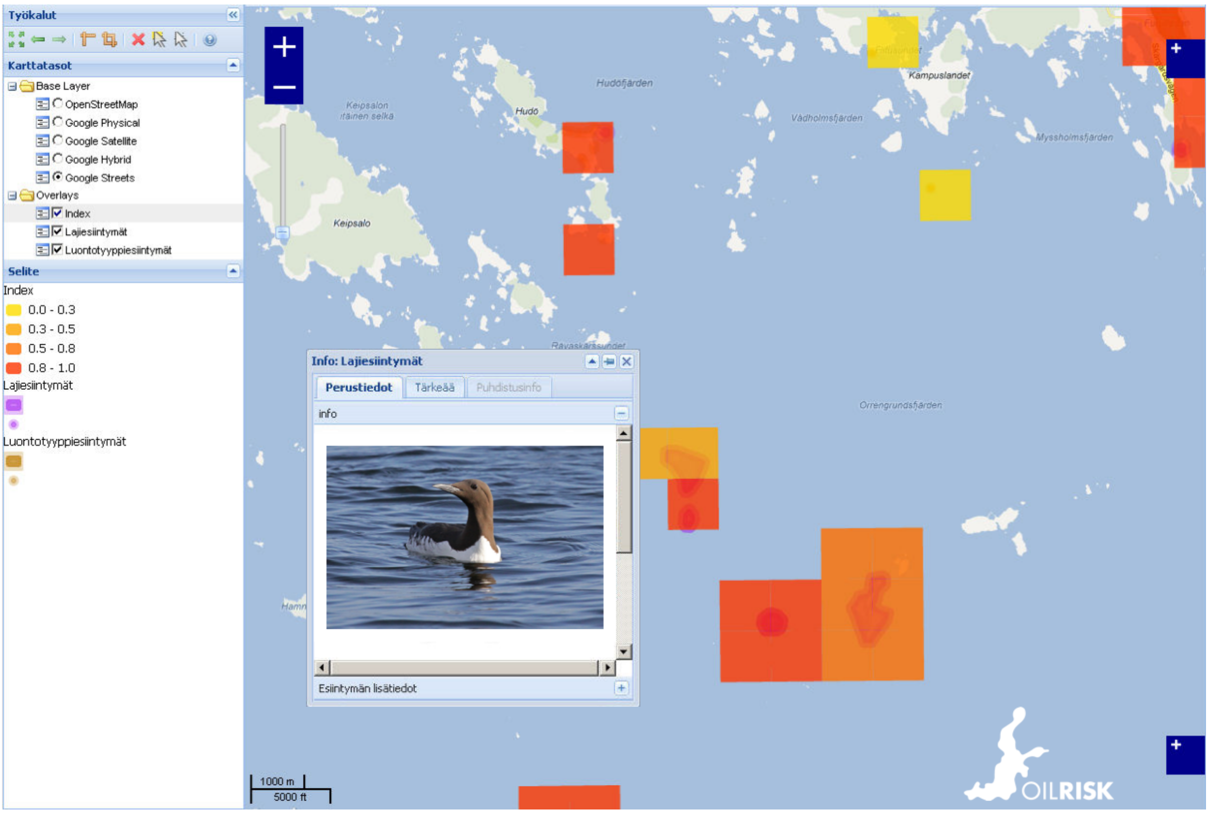

Figure 5. The full version of the map application with information boxes on species and habitat types. The boxes contain information about the ecology of threatened species and habitat types and practical oil spill risk management. The colored grids describe the prioritization index value in each cell. Currently, the application is available in Finnish.

information about vulnerable coastal ecosystems will be given to decision-makers to help them decide which areas should be protected first to minimize the adverse effects of the oil spill. The results showed that such interactive and intelligent tools are an efficient method of delivering ecological knowledge and information for oil spill response and oil spill risk management.

Spatial information available for the application includes the occurrences of threatened species and habitat types, and the index layer describing the conservation prioritization. In an accident situation, a layer indicating the drifting of the oil slick should be added to the overlays. Currently, this is not possible due to the different institution policies related to the software selection. The application was developed using open-source software and the application programming interfaces are not congruent. However, the information regarding conservation prioritization and occurrences of species and habitat types can be included in the national situation awareness system for environmental emergency response (BORIS) which contains oil drifting trajectories.

Detailed information about coastal ecosystems can be displayed in the map application, and precautionary information is given on how to deal with the species and habitat types in question during the operational phase. Recommendations for cleaning are also provided. Since the possible user group for the application is diverse, including rescue officers and ecologists who may not be computer system experts, the application is designed to be intuitive, that is, learnable and memorable to use. The design of the application was discussed with the response personnel during our work. Iterative discussions and training with the potential users were essential throughout the development. For the successful user experience, the simplicity of the interface and the speed of the system were called for. Furthermore, oil spill authorities expected that all the necessary information would be obtainable, even though they would not use it. Thus, the ecological information about the species and habitat types is available in the database.

6.5. New role of Metsähallitus Parks and Wildlife Finland in national preparedness planning

In order to carry out effective ERAs and protection plans, it is necessary to know how coastal ecosystems might be affected by oil spills, and where the occurrences of sensitive species and habitat types are located. In contingency planning, Metsähallitus will provide updated spatial ecological data for risk assessment and oil spill response, which will ensure that the required spatial information for the Finnish shoreline is up-to-date and this information is readily available for assessments and in the event that a spill should happen.

In the assessment report, other additional activities were described to increase the participation of Metsähallitus in oil spill response. Its suggested duties include 1) participation in the search, capture and transportation of oiled birds, 2) participation in oiled shoreline assessment surveys, and 3) sampling follow-up data on ecological consequences after oil spills. Furthermore, training of personnel and participation in oil spill exercises is considered essential in Metsähallitus' preparedness plans. In addition, Metsähallitus should take species and habitat types vulnerable to oil into account in their own management plans for marine protected areas (MPAs).

The main challenge to achieve the above mentioned activities is the possible lack of funding. The tasks were planned based on the existing knowledge of personnel, available resources and possessed equipment. However, the existing funds are limited to their current purposes. The Ministry of the Environment could allocate additional funds for Metsähallitus because their participation would improve the national preparedness.

7. Discussion

Here I discuss the novelties of the work for the conservation of marine ecosystems, summarize the most relevant findings of my thesis related to oil spill risk management, and discuss the work from a practical and methodological point of view. In oil spill management, risks need to be assessed, the expected utilities defined and the oil spill risk managed to achieve the expected goals. This thesis demonstrates methods to estimate risks and gives novel insights into oil spill risk management. In the future, methods to quantify utility functions based on ecological and socioeconomic aspects should be further improved.

7.1. Extrapolation based on the similarity of ecological functions

Predicting the adverse effects of oil spills on a single species may be difficult in comprehensive ecosystem assessments and even unnecessary in oil spill risk management. In these cases, functional groups can be used to reduce the diversity of species to operational entities for prediction and modeling (Lecklin et al. 2011, Thompson et al. 1996). Functional groups in ecology are defined as non-phylogenetic groupings of species, which perform similar ecological functions and share physiological similarities in an ecosystem. This can be in relation either to the contribution of species to ecosystem processes, or to the response of species to changes in environmental variables (Lavorel et al. 1997). It is noteworthy that species included in functional groups cannot be included on a strictly taxonomic basis (Lecklin et al. 2011).

In oil spill risk assessment, species can be classified by their response to disturbance (Lavorel et al. 1997). When estimating species' tolerance to specific environmental factors, disturbance-related attributes should be taken into account. These are categorized as a) life-history parameters, b) morphological types, and c) recolonization ability (Lavorel et al. 1997). However, it should be noted that functional groups may conceal subtle mechanisms that underlie the response of particular species to a specified disturbance (Lavorel 1997). Correlations among life-history parameters have arisen from differences in life-history strategies, and relationships between morphology and life-history strategies arise as a result of evolutionary processes (Pulkkinen et al. 2011). Life-history characteristics are traits that affect the recovery of a species, and can be seen as investments in growth and reproduction. Examples of life-history characteristics include length of lifespan and sexual maturity (Reece et al. 2014).

The Bayesian hierarchical meta-analytic approach models study-to-study heterogeneity explicitly and borrows strength across studies (Pulkkinen et al. 2011). With hierarchical modeling aiming at parameter estimation, the ecological responses of species can be linked to their biological attributes. This enables the efficient use of laboratory tests in modeling approaches, as the specifically tested responses of particular traits could be used in identification of broad correlation patterns (Lavorel 1997). Bayesian hierarchical meta-analysis could be a suitable tool for ecosystem risk assessment using functional groups as subjects, and these approaches should be further developed in oil spill risk management. This approach could improve the understanding of life-history parameters based on existing datasets and formalize available knowledge about the survival and recovery potential of species vulnerable to oil.

7.2. Uncertainty of population recovery

After an oil spill, the recovery potential of threatened species is fundamental, as they already probably suffer from low fecundity and small population size (Ihaksi et al. 2011). The recovery potential of species is assumed to be an outcome of acute loss, recolonization efficiency and reproductive capacity (V). The latter two are seen more significant to the recovery of populations than the oil-induced acute decrease in population size (Albers 2003). However, if there are no individuals after the accident, recovery is impossible despite the high reproductive potential (Ihaksi et al. 2011).

The importance of recolonization and reproduction for recovery varies with the features of the species in question (V, Ihaksi et al. 2011, Lecklin et al. 2011). Populations with a long lifespan, late maturity and relatively few young, like seabirds, are more affected by losses of breeding-age adults than losses of offspring (Votier et al. 2005, Esler et al. 2002). For organisms that produce large numbers of offspring, like invertebrates and fish, significant proportions of eggs or larvae have to be lost to affect recruitment (I). Short-lived species are probably more affected by years with reproductive failure, for example, the loss of the seed cohort may completely destroy the population of an annual plant species without a persistent seed bank in the soil (V, International Petroleum Industry Environmental Conservation Association 2002). For species that a) have good dispersal abilities, b) are able to migrate long distances, and/or c) have low reproductive capacity, recolonization has a larger effect on recovery estimates than a single year's reproduction.

When species abundance is low, population growth rate is assumed to be relatively fast in the presence of negative density dependence. This is a result of reduced intraspecific competition (Mace et al. 2008). However, population growth can be limited in small populations by the Allee effect: that is, individual survival and/or reproductive success can be reduced, leading to a smaller per capita population growth (Stephens et al. 1999). This is relevant after oil spills, when collapsed populations may not recover even if the oil is removed and their living environment restored. In addition, the metapopulation dynamics should be considered when recovery potential of species is assessed in oil spill risk management: this has an impact on recolonization estimates. Metapopulation dynamics describe the migration among local populations and the conditions of regional persistence of species with unstable local populations (Hanski 1998).

Kuparinen and Hutchings (2014) demonstrate through empirically-based models that the Allee effect increases the uncertainty in the recovery time. From the standpoint of oil spill risk management, the increased uncertainty and variability in the recovery time affects the assessment of extinction probabilities (Kuparinen and Hutchings 2014). Moreover, the longer the population remains at too low abundance, the higher the likelihood of extinction, due to reduced capacity to tolerate external stressors, such as oil spills (Kuparinen and Hutchings 2014). It is suggested that the most suitable way to avoid the adverse effects of the Allee effect is to ensure that the population size remains larger than the level at which the Allee effect is likely to present itself (Kuparinen and Hutchings 2014). For oil spill risk assessment, this means that the species suffering from the Allee effect and their quantified levels of Allee effect expression must be identified.

Not all combinations of life-history parameter values are equally probable, and strategies reflect trade-offs between the costs and benefits of reproduction (Pulkkinen et al. 2011). Because of this, the sensitivities of different reproductive strategies should be estimated and taken into consideration in oil spill risk management. Perennial plants recover more rapidly from oil pollution than annual plants (V); an important life-history strategy of plants for surviving disturbance is the existence of seedbanks (Nishihiro et al. 2006, Ozimek 2006, Pahl et al. 2003). The life-history stage at which the protective policy instruments should be applied should be assessed, the alternatives being, for example, before possible population collapse or during restoration. The stochastic simulation modeling approach demonstrated in Paper II is convenient for this.

The importance of facilitating movement has been acknowledged in spatial applications that prioritize actions to protect sites that minimize dispersal distance (Moilanen et al. 2009). In theory, a metapopulation is often stable since immigrants from one population are likely to recolonize habitats that have been left open by the extinction of another population (Doerr et al. 2011). When threatened species are studied, it is especially important to identify key habitat patches and understand what factors govern their occupancy so that their protection can be targeted and effective. In future assessments, it is important to recognize the processes that link and maintain the subpopulations in metapopulations. Furthermore, there is a need to assess the population dynamics to determine if metapopulation dynamics are relevant and then use that information to plan effective restorations.

7.3. Models supporting management design

Decision-making under uncertainty can be hard for humans as our minds have a limited capacity to handle conditional probabilities in more than one or two dimensions (Morgan and Henrion 1990). This means that it is difficult to take uncertainties into account because the connections between decisions do not operate in a deterministic manner (Helle et al. 2011). Nevertheless, the aim of decision-makers in oil spill management is to make inferences about risk in the face of uncertainty (Halpern et al. 2006).

Model-based ERAs provide results that are naturally consistent, transparent (if communicated properly) and free of linguistic confusion (Burgman 2005). The ability to quantify the effects of uncertainty in ERAs creates resilience in the ERM and provides a mechanism for describing realistic outcomes (Halpern et al. 2006). Deficiencies of ERA models include the fact that they assume decision-makers to be rational in the sense that they would follow the instructions of probability theory or decision theory (Burgman 2005). However, this is seldom the case. People tend to comply with strategies that are heuristic, that is, easy to adapt and use, sufficient for immediate goals, but not always optimal (O'Hagan et al. 2006). In addition, as models incorporate uncertainty, they leave the responsibility to consider the possibilities and consequences of incorrect decisions (which result in increased oil spill risk) to managers, since the outputs of ERAs do not translate directly into policy and management decisions (Guisan et al. 2013, Burgman 2005).

Environmental models can be used to structure and solve problems, integrate knowledge and uncertainty, and inform decision-makers by visualizing the results (Lehikoinen et al. 2014, Rahikainen et al. 2014, Brimicombe 2010). However, this is only possible if the models are chosen and deployed appropriately and, of course, employed perceptively (Brimicombe 2010). Modeling outputs can be visualized either as BNs or maps, and these are very effective means for communicating the outcomes of scenarios around the changes of the impacts of policy interventions (Mulligan 2014). Paper III indicates that spatial aspects of ecosystem components should be included in addition to ship accident hotspots. Oil spill risk is distributed unevenly among areas and the presence of threatened species and habitat types has an effect on the total oil spill risk. This supports the commitment to ecosystem-based management. Modeling is an important basis for policy support, since it makes explicit the understanding of processes and combines these with spatio-temporal datasets representing particular states (Mulligan 2014).

Decision-makers may object to the use of models when they believe that model outputs do not result in better-informed decisions than the subjective opinion of experts (Aizpurua et al. 2015, Addison et al. 2013). Thus, they potentially constrain the optimal use of available information (Aizpurua et al. 2015). However, involving managers in the development of models facilitates their understanding and simplifies the assimilation of ERAs in decision-making (Aizpurua et al. 2015). The best ERAs involve stakeholders and experts in an iterative process that improves the ERM (Burgman 2005).

Fisheries management is a field of applied science where scientists routinely advise managers and politicians about future catch allocations (Kuikka et al. 2014). It is an area of risk management which utilizes potential information sources to make scientifically sound estimates for effective decision-making (Kuikka et al. 2014). The use of science can contribute to fisheries management through improved confidence and increased transparency (Dickey-Collas et al. 2010). In the Baltic Sea, Bayesian analysis is applied in Baltic salmon (*Salmo salar*) management. This approach enables a) incorporating uncertainties into predictions about stock development, b) estimating the effects of alternative decisions on various aims of stakeholders and society, c) the possibility of using experts' knowledge in addition to data, and d) taking the precautionary approach into account as the legislation suggests (Kuikka et al. 2014). As noted in Papers II and III, the challenges of oil spill management are very close to those of fisheries management – only instead of stock development, uncertainties exist related to the environmental consequences of oil. Since Bayesian inference is applied to practical scientific advice in Baltic salmon management, this method should also be applied in oil spill management.

7.4. Accumulation and integration of knowledge

BNs are useful in many real-life data analysis and management questions as they consist of variables relevant to the problem. They provide a way to handle missing data, allow the combination of data with domain knowledge, facilitate learning about causal relationships between variables, provide a method for avoiding unnecessary overfitting of data, show good prediction accuracy, and can easily be combined with decision analytic tools to aid management (Uusitalo 2007, Jensen and Nielsen 2001). BNs are a transparent and consistent method for inductive inference, and are good at communicating uncertainty.

It has been noted that tests of chronic exposure are required to understand the impacts of oil on the marine ecosystem (National Research Council 2003). On the other hand, it is stated that acute tests of toxicity in the laboratory *per se* are inadequate for ecotoxicological risk assessment (Peterson et al. 2003). Thus, results from laboratory tests (e.g. I) should be combined with the outputs of population models (e.g. II). BNs can use different kinds of information as probability distributions: for example, posterior distributions from simulation models, expert knowledge, and statistical data (Helle et al. 2015, Lehikoinen et al. 2015, Fernandes et al. 2012, Uusitalo et al. 2012). In Paper II, we combined posterior distributions from a Monte Carlo Markov Chain (MCMC) simulation model with a BN in order to estimate the recovery potential of populations after oil spills with expert knowledge to estimate, for example, the oil exposure of birds based on their behavior. In the BN described in Paper III, we used statistical data to estimate the accident probability of oil spills.

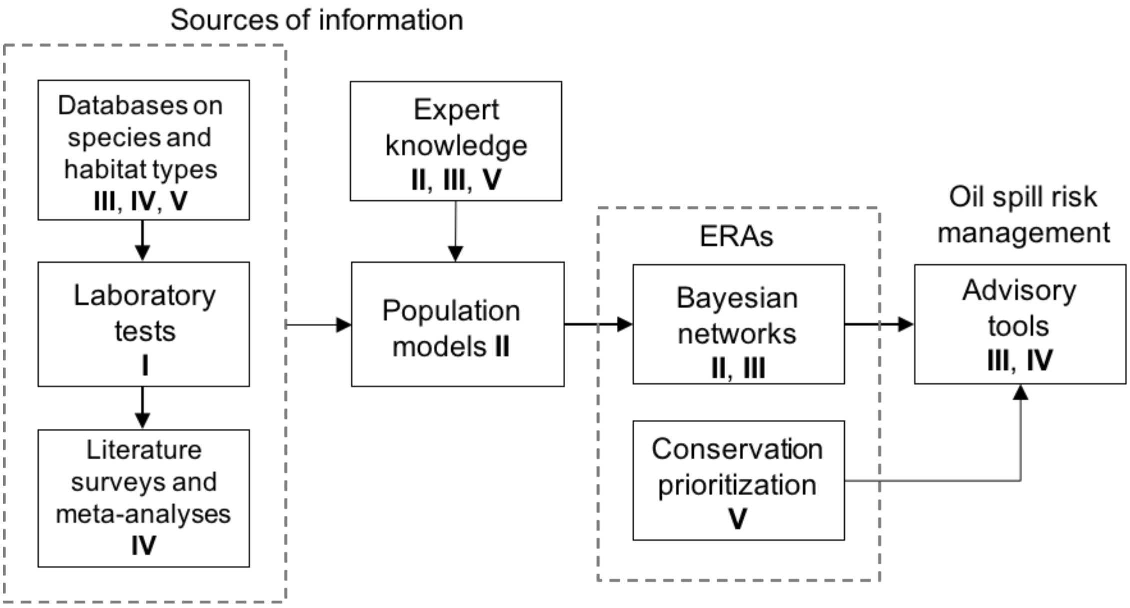

Figure 6. Integration of accumulated knowledge into ERAs and advisory tools: arrows describe information flows between methods.

As different information can be incorporated into population models whilst knowledge accumulates, they can be further developed to estimate the comprehensive oil spill risk (Fig. 6). BNs estimating the effects of oil spills on coastal and marine ecosystems should be connected to risk assessment and management decision tools. BNs are also suitable for ERA and ERM tools (III, Helle et al. 2015, Lehikoinen et al. 2015).

ERM perceives multiple actions in seas and their possible contradicting objectives. Managing ecosystems that are affected by several human actions including oil shipping requires methods that enable the analysis of the joint effects of different factors in one framework. Combining knowledge from experts, field surveys, laboratory experiments, data analysis and modeling, probabilistic models and spatial tools provides a suitable method for implementing ecosystem-based management. If BNs are created to estimate the harmful effects of oil spills on certain species and habitat types, as described in Papers II and III, the same models can be utilized in marine spatial planning when the sustainable reconciliation of human actions and nature values is being pursued.

ERAs do not only describe the quantitative risk value, but also help to identity, understand and quantify the complex interrelationships underlying even seemingly simple situations. The results of ERA depend on the choice of risk attitudes and people tend to be risk-averse on average (for the prospect theory see Kahneman and Tversky 1979). This has an effect on the preferred expected value. BNs help decision-makers to see how risks emerge, how they are connected, and how they can be controlled and mitigated to the best level possible (Fenton and Neil 2012). Occasionally, BNs might be seen as more of a hindrance than a help in ERAs, as they require mental effort to make the problem tractable: care has to be taken to identify cause and effect, the states of variables need to be carefully identified, and probabilities that reflect the best current knowledge need to be assigned (Fenton and Neil 2012). In Paper II, we found it rather difficult to define the structure of BN for common guillemots: causal inference and choosing the correct edge direction required attention. When assessing the causality of nodes between the oil-induced loss of adults and their offspring, the direction implied by the deduction caused an invalid BN compared to the direction of actual cause and effect. We were able to correct this when we studied the incoherence in distributions. BNs are workable as they allow this kind of examination.

7.5. Advisory tools

Since the objectives of ERA are to produce new information for ERM and policy design, which should rely on the extensive use of scientific evidence, communication is of great importance. Managers may object to the use of models if they consider them poorly communicated (Borowski and Hare 2007). Scientific information can be summarized within Web-based advisory tools and probabilistic frameworks communicating risks. The visualization of knowledge is an important function of advisory tools. A good piece of modeling, with uncertainty reduced to a minimum, will fail if the results cannot be adequately communicated (Brimicombe 2010).

Advisory tools can be graphical, as with BNs (e.g. III) or maps (e.g. III and IV) which are suitable for communicating the outcomes of risk scenarios to decision-makers and policy advisors (Mulligan 2014). Spatially explicit tools can help handle and communicate the outcomes of modeling (Mulligan 2014). Spatial risk assessments offer valuable information for the development of oil spill management strategies (III, Frazão Santos et al. 2013).

Noted objections of managers to the use of model outputs relate to the need for conceptual and technical expertise and the amount of resources and time needed to implement such procedures (Aizpurua et al. 2015). Thus, there is a need for easy-to-use tools. Advisory tools can transfer model outcomes into comprehensible suggestions for management actions. Participatory processes involving relevant stakeholders facilitate the development of usable tools and the adoption of evidence into decision-making, as in Papers III–V. The tools and frameworks described in this thesis enable better learning for decision-makers, and also provide subject-specific advice for the implementation of assessed management actions. If the results of ERAs are communicated using advisory tools instead of policy briefs, the information can be more profoundly appreciated by decision-makers (Mulligan 2014). With the tools and outputs from Papers III–V, we were able to dispense new knowledge and demonstrate novel approaches to the stakeholder groups, such as conservation biologists and response authorities.

7.6. Policy instruments in oil spill risk management

Policy instruments are translated into practical regulations which are mostly enforced by the environmental and maritime authorities involved in oil spill risk management. The aim is to restrict the harmful actions of citizens and companies toward nature. An indirect instrument may include the enactment of safer maritime legislation in order to avoid oil spills; a more direct method would be the prohibition of shipping near a vulnerable seabird nesting area. The selection of suitable management policy is difficult in situations where information gaps exist (Aven and Zio 2014). In oil spills, validated risk models providing the probabilities of accident occurrences and their environmental consequences are not available. This creates a situation where researchers are divided on what kind of actions to recommend to reduce risks. Moreover, the question arises as to the risk management policies that decision-makers should adopt in such cases.

As oil spill response and its success can be uncertain, preventive measures for oil accidents are often a better choice. Haapasaari et al. (2015) state that a proactive approach to safety management will save economic resources and avert environmental damage. However, widespread concern about the environmental effects of shipping activities remains toned down as long as the maritime industry supports the global economy (Wan et al. 2016). This can reduce the willingness to invest to the proactive management of irregular and uncertain oil spills. Furthermore, oil spills may occur offshore beyond the national jurisdiction, and thus, the shipping policies should be applied worldwide to be effective (Wan et al. 2016). A downside is that the enforcement of maritime legislation is slow and the outcomes can be inefficient as they are constrained by the international laws and rules. Based on this, more regional approaches should be introduced to protect the Baltic Sea ecosystem. Moreover, the oil spill risk can be controlled regionally.

Marine spatial planning (MSP) is a proactive policy instrument. It is a planning process that implements risk-averse ecosystem-based management and enables integrated, forward-looking and consistent decision-making (Stelzenmüller et al. 2010, Ehler and Douvere 2007). In addition, MSP is an instrument whereby relevant stakeholder groups can be involved in a transparent manner. Oil spill risk may be reduced with careful and efficient MSP. Ecosystem-based fairway planning directs maritime traffic to areas where the environmental consequences of oil spills will be less harmful (Soomere et al. 2012). Traffic separation schemes can be placed to the areas where the sensitive threatened species and habitat types are absent. In papers III and V we describe methods to identify the species and habitat types at risk. However, it is noteworthy that route planning is often based on navigational constraints, and the locations of coastal shipping lanes might be impossible to reposition. In MSP, planners can assess the cumulative effects of human pressures and seek to make actions more sustainable and proactively minimize the conflicts between shipping and other human actions. Paper III provides information on areas where the oil shipping causes the highest risk.

Climate governance has extended on a more regional level through changes initiated by different actors, such as business, local government, and civil society (Jordan et al. 2015). Moreover, less top-down and analytical frameworks, for example, emissions trading systems and collaboration between cities, populate the current landscape (Jordan et al. 2015). This is based on the argument that to become more effective, climate governance should become more diverse and multileveled (Stewart et al. 2013). Similar polycentric governance should be introduced in maritime traffic. In the Baltic Sea, a new direct policy instrument could be introduced to reduce accident probability. This novel and regional safety management procedure could aim to increase regional ship safety by controlling certain safety factors (e.g. personal or technical).

Port State Control (PSC) is an already working system of harmonized inspection procedures designed to target substandard ships, with the main objective to bring them into conformity with international maritime law and regulations. However, national PSC could be improved with regional Baltic Sea inspection practices, since the current practices are not working to assure the adequate safety of tanker ships. Suggestions for regional inspection practices are described below. Tankers entering the Baltic Sea for the first time could complete a safety checklist. This should be done in advance in the port of departure when planning the route. Masters could complete an electronic checklist on the Web, and provide information about safety factors (e.g. faults in bow thruster, sea chest heating during winter, crew competence matrix) and give pre-arrival information for pilots (e.g. gyro compass error). If any defects are noticed, ships will be vetted in the port of call. Furthermore, information about previous or current cargo is important when inspections are planned: if the cargo is oil, the Oil Discharge Monitoring Systems and Equipment (ODME) could be inspected to check whether oil has been discharged into the sea. In addition, spot checks would be possible. This Baltic Sea inspection practice could be upheld by HELCOM, since it is an international organization working in the area, and has experience in marine conservation and maritime transportation. If the tanker passes the inspection, it would receive a certificate of compliance which could remain valid for 12 months. A similar system is currently in operation in the United States, controlled by the United States Coast Guard. In addition to safety control, HELCOM would be able to collect more comprehensive information about the safety status of ships in the Baltic Sea. This could have a significant impact on future policy planning. Currently, information about ship defects is collected by shipowners and other private actors, such as the Oil Companies International Marine Forum (OCIMF). In Finland, the Finnish Transport Safety Agency collects only accident and near-miss incident reporting.

8. Conclusions and implications for management

Developing efficient measures to protect coastal and marine ecosystems and manage possible risks requires an interdisciplinary approach. In such an approach, ecological knowledge based on biology needs to be integrated with information about human activities which can be analyzed by social and economic sciences and controlled by environmental laws and policies. Interactive and intelligent technical solutions aid in interpreting the accumulated knowledge for decision-makers and policy advisors.

This thesis offers new insight into oil spill risk management in the Baltic Sea. It highlights the need for risk analytical marine conservation, both proactive and reactive. In addition, it demonstrates the benefit of knowledge-based marine conservation, and how information could be provided for decision-making. Moreover, the thesis emphasizes the importance of learning in oil spill risk management from previous studies and also from other disciplines.

The original five scientific papers contribute to answering the questions, 1) what are the methods for gathering knowledge on the impacts of oil on sensitive species, 2) how could ecological knowledge be given to the decision-makers, and 3) what are the improved approaches to the required ecological knowledge in oil spill risk management? Based on the synthesis of this thesis, suggestions can be made to improve oil spill risk management and the development of knowledge-induced policy planning.

To conclude, BNs are functioning tools for ERA and thus widely used in environmental modeling (Chen and Pollino 2012, Aguileira et al. 2011, Uusitalo 2007, McCann et al. 2006). Their adoption would enhance the current oil spill risk management in the Baltic Sea, since all the variables affecting oil spill risk could be considered in one framework in a transparent manner. This would improve risk communication to the public, and thus help in the public understanding of risk, which influences political decisions. BNs enable the use of different kinds of information and the consideration of uncertainty, which are advantages in an issue, such as oil spill risk management. This approach challenges the current policy framework. Most studies in the field of maritime risk assessment focus on calculated estimates based on measured data. Furthermore, uncertainty treatment is lacking from several existent decision-making applications (Goerlandt and Montewka 2015).

In oil spill response, the prioritization of coastal and marine species and habitat types should be made based on their conservation value, oil-induced loss, recovery potential and protection effectiveness (V). Moreover, a need for better knowledge about the factors affecting the recovery potential of threatened species is evident. Therefore, more comprehensive assessments should be carried out taking into account the Allee effect, life-history strategies and the metapopulation dynamics. This requires the gathering of more ecological information.

8.1. Linking science to policy

One important factor in policy innovation is the extent to which new policies achieve significant and lasting effects on the problems they seek to address (Hildén et al. 2015). As the evaluation systems are lacking, policy-makers seem to know relatively little about what the instruments they might introduce will achieve (Mickwitz and Birnbaum 2009). The role of policy-support tools is to provide scientific support and help connect scientific knowledge to the understanding of change in landscapes of operational decision-making in the policy domain (Mulligan 2014). As SDSSs are intended to assist decision-making around a specific issue, policy-support tools assist in the design of broader policies. The use of science could be improved in policy planning if ERAs and ERMs were further developed as policy-support tools.

This thesis has shown that probabilistic risk assessment tools can be developed and that they make it possible to provide justified risk-averse planning. The probabilistic method applied in the Paper III will be of high interest to marine spatial planners who have to cope with uncertainties typical for ERA and ERM.

8.2. Challenges ahead

This thesis has concentrated on the spawning and nursery grounds of commercially important fish species (I), migrating seabirds (II) and threatened species and coastal habitat types (III–V). It is evident that in future assessments, other ecosystem components, such as hotspot occurrences of keystone species should also be taken into account. In addition, common ecosystem interactions and cascading effects should be considered. This calls for the combination of oil spill ERAs with ecosystem models.

Researchers, decision-makers and representatives of NGOs should be networking from the early stages of ERA and ERM (Hildén et al. 2004). The iterative information flow between them should be transparent, involving and flexible (Hildén et al. 2004). As this thesis indicates, a new culture of open information and utilization of open-access tools is required to support knowledge-based oil spill risk management. In addition, the public should be involved, as they will define the expected utilities. This requires participatory processes and successful communication between the producers and users of knowledge and the target groups of policy instruments.

Knowledge of coastal and marine ecosystems in needed for efficient conservation decisions in oil spill risk management (Uusitalo et al. 2015, Burgman 2005). This demands sustained funding for research. Funding for scientific research comes predominantly from the government and private operators. Other important funding sources are EU and foundations. However, research funding may be decreased by national politics, which would weaken the evidence to support decision-making. Thus, other sources for funding should be pursued. The polluter pay principle is implemented in oil spills. The principle can be understood as an overarching rule of environmental responsibility encompassing the pollution prevention and the liability, that is clean-up costs of damage to the environment. Protection and indemnity insurance (P&I insurance) covers the shipowner’s liability towards third parties for oil pollution that arises from the operation of ship. If a polluter cannot be identified or a litigation process delays the compensation, national and international Oil Pollution Compensation Funds (OPCFs) reimburse the costs of oil damage and oil spill response. The funds are secondary sources of compensation and they will recover the payment back from the responsible party or the International Oil Pollution Compensation Fund and its Supplementary Fund. The operations of OPCFs are funded through the accumulated oil protection fees which are collected from shipowners for shipped oil. The polluter pay principle would imply that the shipping industry should also fund the research related to oil spill risk assessment. The state-managed OPCFs could grant funds to research in order to achieve the independence of researchers. The outcomes of Papers III and V demonstrate the coastal ecosystem at risk and could act as a justification for the increased oil protection fees.

Acknowledgements

Firstly I wish to thank my supervisors Sakari Kuikka and Mika Rahikainen. I am grateful to Sakari for giving me a great opportunity to work in the Fisheries and Environmental Management (FEM) Group. His way of recognizing people's strengths and trusting their abilities has helped me as I have developed to become an independent researcher. I am thankful to Mika for all the encouragement and patience he has given me during these past years. He has set an excellent example of an inquisitive, committed and scholarly researcher. He also shows the right amount of hang loose attitude.

I am very grateful to Mark Dickey-Collas for examining my thesis and acting as an opponent in the defense. I sincerely thank Mikael Hildén and Charles H. Peterson for pre-examining my thesis. Their comments and views have been extremely helpful, and I appreciate the time and expertise they have put into the pre-examination.

I thank all my co-authors for their valuable input and for sharing their extensive knowledge with me. Inari Helle has been my closest coworker in most of the papers. I am much obliged to Inari for her help and insight. I wish to thank Juha Karjalainen who was my supervisor during my Bachelor's and Master's studies, and inspired me to pursue a career as a researcher. I also want to thank Ari Jolma and Samu Mäntyniemi for teaching me in many things.

I wish to thank Miina Karjalainen, Elina Leskinen, and Laura Uusitalo for being the members of my thesis advisory committee. The meetings have been a great opportunity for me to pull my ideas together and receive educational comments about my work.

I want to thank all the current and former members of FEM Group. I have been a part of the group for many years and I have had a possibility to work with such a talented, committed and wonderful people. Special thanks to Kirsi Hoviniemi who has helped me in several matters during the years, and to Päivi Haapasaari, Anna Kuparinen, and Jarno Vanhatalo who all have provided food for thought in many research-related issues. I also want to thank Suvi Ignatius, Mirka Laurila-Pant, and especially Annukka Lehikoinen for sharing a room with me in Kotka.

I have also worked as a part of the Kotka Maritime Research Centre (Merikotka). I want to thank all the researchers and personnel who have created such a multidisciplinary and incomparable team. Special thanks to the various kinds of Merikotka sport clubs.

I want to thank all the coworkers in the Finnish Environment Institute (SYKE) and in the Finnish Inventory Programme for the Underwater Marine Environment (Velmu). People working in the SYKE Pollution Response group has helped me in the issues related to oil spills and the researchers in Velmu has produced the information about marine ecosystems. Many species and habitat type experts have contributed to the work during the years, for which I am grateful.

Finally, I am thankful to my family and friends, especially to my husband Sakari for his endless love. Thanks to our dog Topi who is one of a kind. During my thesis work, he always knew when it was time to take a brake and go for a walk. I am grateful to my parents for their all-embracing support, and to my brother and sister, Mikko and Pauliina for growing up together and sharing a lot. I would like to thank my group of friends for always being there. Eeva, Katja N, Katja S and Niina give me joy, love and stories to remember. Special thanks to my dear and talented friend Henri for designing the cover of my thesis. Lastly, I want to thank Sanna for being a member in the MR Appreciation Club.

The work presented in this thesis has mainly been conducted as part of the following research projects: OILRISK (Applications of ecological knowledge in managing oil spill risk) and IBAM (Integrated Bayesian risk analysis of ecosystem management in the Gulf of Finland). Additional funding was received from the Centrum Balticum Protection Fund for the Archipelago Sea, for which I express my gratitude.

Bibliography

Acevedo M.A., Sefair J.A., Smith C., Reichert B. & Fletcher Jr. R.J. 2015. Conservation under uncertainty: optimal network protection strategies for worst-case disturbance events. Journal of Applied Ecology 52: 1588–1597.

Addison P.F., Rumpff L., Bau S.S., Carey J.M., Chee Y.E., Jarrad F.C., McBride M.F. & Burgman M.A. 2013. Practical solutions for making models indispensable in conservation decision-making. Diversity and Distributions 19: 490–502.

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

Aizpurua O., Cantú-Salazar L., San Martin G., Biver G., Brotons L. & Titeux N. 2015. Reconciling expert judgement and habitat suitability models as tools for guiding sampling of threatened species. Journal of Applied Ecology 52: 1608–1616.

Albers P. 2003. Petroleum and individual polycyclic aromatic hydrocarbons. In: Hoffman D. J., Rattner B. A., Burton G. A. Jr. & Cains J. Jr. (Eds.). Handbook of Ecotoxicology. CRC Press, Boca Raton, pp. 342–408.

Alenius P., Myrberg K. & Nekrasov A. 1998. The physical oceanography of the Gulf of Finland: a review. Boreal Environment Research 3: 97–25.

Alonso-Alvarez C., Pérez C. & Velando A. 2007. Effects of acute exposure to heavy oil from the Prestige spill on a seabird. Aquatic Toxicology 84: 103–110.

Andrejev O., Myrberg K., Alenius P. & Lundberg P. 2004. Mean circulation and water exchange in the Gulf of Finland – a study based on three-dimensional modeling. Boreal Environment Research 9: 1–16.

Assmuth T. & Hildén M. 2008. The significance of information frameworks in integrated risk assessment and management. Environmental Science & Policy 11: 71–86.

Aven T. 2010. On how to define, understand and describe risk. Reliability Engineering & System Safety 95: 623–631.

Aven T. & Zio E. 2014. Foundational Issues in Risk Assessment and Risk Management. Risk Analysis 34: 1164–1172.

Barron M., Carls M., Heintz R. & Rice S. 2004. Evaluation of fish early life-stage toxicity models of chronic embryonic exposures to complex polycyclic aromatic hydrocarbon mixtures. Toxicological Sciences 78: 60–67.

Boersma P., Kareiva P., Fagan W., Clark J. & Hoekstra J. How Good Are Endangered Species Recovery Plans? BioScience 51: 643–649.

Borowski I. & Hare M. 2007. Exploring the gap between water managers and researchers: Difficulties of model-based tools to support practical water management. Water Resource Management 21: 1049–1074.

Bottrill M., Joseph L., Carwardine J., Bode M., Cook C., Game E., Grantham H., Kark S., Linke S., McDonald-Madden E., Pressey R., Walker S., Wilson K. & Possingham H. 2008. Is conservation triage just smart decision making? Trends in Ecology and Evolution 23: 649–654.

Boufadel M., Sharifi Y., Van Aken B., Wrenn B. & Lee K. 2010. Nutrient and oxygen concentration within the sediments of an Alaskan beach polluted with the *Exxon Valdez* oil spill. Environmental Science and Technology 44:7418–7424.

Briggs K.T., Yoshida S.H. & Gershwin M.E. 1996. The influence of petrochemicals and stress on the immune system of seabirds. Regulatory Toxicology and Pharmacology 23: 145–155.

Brimicombe A. 2010. GIS, Environmental Modeling and Engineering, Second Edition. CRC Press, Boca Raton.

Brown E., Baker T., Hose J., Kocan R., Marty G., McGurk M., Norcross B. & Short J. 1996. Injury to the early life history stages of Pacific herring in Prince William Sound after the Exxon Valdez oil spill. American Fisheries Society Symposium 18: 448–462.

Brunila O.-P. & Storgård J. 2012. Oil transportation in the Gulf of Finland in 2020 and 2030. Publications from the Centre for Maritime Studies University of Turku A61.

Burgman M. 2005. Risks and Decisions for Conservation and Environmental Management. Cambridge University Press, Cambridge.

Cadiou B., Riffaut L., McCoy K., Cabelguen J., Fortin M., Célinaud G., Le Roch A., Tirard C. & Boulinier T. 2004. Ecological impact of the Erika oil spill: determination of the geographic origin of the affected common guillemots. Aquatic Living Resources 17: 369–377.

Calow P. 2009. Who should value nature? Integrated Environmental Assessment and Management 4: 369–370.

Calow P. & Forbes V. 2003 Peer Reviewed: Does Ecotoxicology Inform Ecological Risk Assessment? Some argue that ecotoxicology is too simplistic to do the job effectively. Environmental Science and Technology 37: 146A–151A.

Carls M., Holland L., Larsen M., Collier T., Scholz N. & Incardona J. 2008. Fish embryos are damaged by dissolved PAHs, not oil particles. Aquatic Toxicology 88: 121–127.

Cerqueira M.C., Cohn-Haft M., Vargas C.F., Nader C.E., Andretti C.B., Costa T.V.V., Sberze M., Hines J.E. & Ferraz G. 2013. Rare or elusive? A test of expert knowledge about rarity of Amazon forest birds. Diversity and Distributions 19: 710–721.

Chen S.H. & Pollino C.A. 2012. Good practice in Bayesian network modelling. Environmental Modelling & Software 37: 134–145.

Chen I., Hill J., Ohlemüller R., Roy D. & Thomas C. 2011. Rapid Range Shifts of Species Associated with high Levels of Climate Warming. Science 333: 1024–1026.

Christensen N.L., Bartuska A.M., Brown J.H., Carpenter S.D., Antonio C., Francis R., Franklin J.F., MacMahon J.A., Noss R.F., Parsons D.J., Peterson C.H., Turner M.G. & Woodmansee R.G. 1996. The report of the ecological society of America committee on the scientific basis for ecosystem management. Ecological Applications 6: 665–691.

Cohen A., Nugegoda D. & Gagnon M.M. 2001. Metabolic Responses of Fish Following Exposure to Two Different Oil Spill Remediation Techniques. Ecotoxicology and Environmental Safety 48: 306–310.

Culbertson J., Valiela I., Pickart M., Peacock E. & Reddy C. 2008. Long-term consequences of residual petroleum on salt marsh grass. Journal of Applied Ecology 45: 1284–1292.

Davis H.K., Moffat C.F. & Shepherd N.J. 2002. Experimental Tinting of Marine Fish by Three Chemically Dispersed Petroleum Products, with Comparisons to the Baer Oil spill. Spill Science & Technology Bulletin 7: 257–278.

Dean T.A., Stekoll M.S. Jewett S.C. Smith R.O. & Hose J.E. 1998. Eelgrass (*Zostera marina* L.) in Prince William Sound, Alaska: Effects of the Exxon Valdez oil spill. Marine Pollution Bulletin 36: 201–210.

Dickey-Collas M., Nash R., Brunel T., van Damme C., Marshal T., Payne M., Corten A., Geffen A., Peck M., Hatfield E., Hintzen N., Enberg K., Kell L. & Simmonds J. 2010. Lessons learned from stock collapse and recovery of North Sea herring: a review. ICES Journal of Marine Science 4: 26.

Doerr V.A.J., Barrett T. & Doerr E.D. 2011. Connectivity, dispersal behaviour and conservation under climate change: a response to Hodgson et al. The Journal of Applied Ecology 48: 143–147.

Ehler C. & Douvere F. 2007. Visions for a Sea Change. Report of the First International Workshop on Marine Spatial Planning. Intergovernmental Oceanographic Commission and Man and the Biosphere Programme. IOC Manual and Guides No. 48, IOCAM Dossier No. 4. Paris, UNESCO.

Elliott M. 2013. The 10-tenets for integrated, successful and sustainable marine management. Marine Pollution Bulletin 74: 1–5.

Esler D., Bowman T., Trust K., Ballachey B., Dean, T., Jewett, S. & O'Clair, C. 2002. Harlequin duck population recovery following the Exxon Valdez oil spill: progress, process and constraints. Marine Ecology Progress Series 241: 271–286.

Fenton N. & Neil M. 2013. Risk Assessment and Decision Analysis with Bayesian Networks. CRC Press, Boca Raton.

Fernandes J., Kauppila P., Uusitalo L., Fleming-Lehtinen V., Kuikka S. & Pitkänen H. 2012. Evaluation of Reaching the Targets of the Water Framework Directive in the Gulf of Finland. Environmental Science and Technology 46: 8220–8228.

Finch B., Wooten K. & Smith P. 2011. Embryotoxicity of weathered crude oil from the Gulf of Mexico in mallard duck (Anas platyrhynchos). Environmental Toxicoly 30: 1885–1891.

Fässler S., Payne M., Brinel T. & Dickey-Collas M. 2011. Does larval mortality influence population dynamics? An analysis of North Sea herring (*Clupea harengus*) time series. Fisheries Oceanography 20: 530–543.

Thorbek P., Forbes V., Heimbach F., Hommen U., Thulke H.-H., Van den Brink P., Wogram J. & Grimm V. 2009. Ecological models for regulatory risk assessments of pesticides: Developing a strategy for the future. SETAC Press, Florida.

Forsberg J.A., Eberhardt J., Boland P.J., Wedin R. & Healey J.H. 2011. Estimating Survival in Patients with Operable Skeletal Metastases: An Application of a Bayesian Belief Network. PLOSOne 6, e19956.

Frazão Santos C., Michel J., Neves M., Janeiro J., Andrade F. & Orbach M. 2013. Marine spatial planning and oil spill risk analysis: Finding common grounds. Marine Pollution Bulletin 74: 73–81.

French McCay D., Rowe J.J., Whittier N., Sankaranarayanan S. and Etkin D.S. 2004. Estimation of potential impacts and natural resource damages of oil. Journal of Hazardous Materials 107: 11–25.

Fulton E.A., Smith, A.D.M., Smith, D.C. and van Putten, I.E. (2011). Human behaviour: the key source of uncertainty in fisheries management. Fish and Fisheries 12, 2–17.

Game E., Kareiva P. & Possingham H. 2013. Six Common Mistakes in Conservation Priority Setting. Conservation Biology 27: 480–485.

Gaston A. 2004. Seabirds – a natural history. Black Publishers Ltd., London.

Gaston K. & Fuller R. 2008. Commonness, population depletion and conservation biology. Trends in Ecology and Evolution 23: 14–19.

Gilks W., Thomas A. & Spiegelhalter D. 1994. A language and program for complex Bayesian modeling. The Statistician 43: 169–177.

Goerlandt F. & Montewka J. 2015. A framework for risk analysis of maritime transportations systems: A case study for oil spill tankers in a ship-ship collision. Safety Science 127: 77–85.

Goldstein B., Osofsky H., Lichtveld M. 2011. The Gulf Oil Spill. The New England Journal of Medicine 364: 1334–1348.

Gonzàlez J., Viñas L., Franco M., Fumega J., Soriano J., Grueiro G., Muniategui S., López-Mahía P. Prada D., Bayona J., Alzaga R. & Albaigés J. 2006. Spatial and temporal distribution of dissolved/dispersed aromatic hydrocarbons in seawater in the area affected by the Prestige oil spill. Marine Pollution Bulletin 53: 250–259.

Guisan A., Tingley R., Baumgartner J. et al. 2013. Predicting species distributions for conservation decisions. Ecology Letters 16: 1424–1435.

Haapasaari P. & Karjalainen T.P. 2010. Formalizing expert knowledge to compare alternative management plans: sociological perspective to the future management of Baltic salmon stocks. Marine Policy 34: 477–486

Haapasaari P. 2012. Addressing human-induced uncertainty in fisheries management: social scientific and interdisciplinary solutions using Bayesian belief networks. Doctoral dissertation, University of Helsinki, Helsinki.

Haapasaari P., Mäntyniemi, S. and Kuikka S. 2012. Baltic Herring Fisheries Management: Stakeholder Views to Frame the Problem. Ecology and Society 17: 36.

Haapasaari P., Helle I., Lehikoinen A., Lappalainen J. & Kuikka S. 2015. A proactive approach to marine safety policy making for the Gulf of Finalnd: seeking best practices. Marine Policy 60: 107–118.

Halpern B., Regan H., Possingham H. & McCarthy M. 2006. Accounting for uncertainty in marine reserve design. Ecology Letters 9: 2–11.

Halpern B.S., Walbridge S., Selkoe K.A., Kappel C.V., Micheli F., D'Agrosa C., Bruno J.F., Casey K.S., Ebert C., Fox H.E., Fujita R., Heinemann D., Lenihan H.S., Madin E.M.P., Perry M.T., Selig E.R., Spalding, M., Steneck R. & Watson R. 2008. A global map of human impact on marine ecosystems. Science 319: 948–952.

Halpern B., Longo C., Scarborough C., Hardy D., Best B., Doney S, Katona S., McLeod K., Rosenberg A. & Samhouri J. 2014. Assessing the health of the US west coast with a regional-scale application of the Ocean Health Index. PLOSOne 9: e98995.

Hanski I. 1998. Metapopulation dynamics. Nature 396: 41–49.

Hario M., Rintala J. & Nordenswan G. 2009. Dynamics of wintering long-tailed ducks in the Baltic Sea – the connection with lemming cycles, oil disasters, and hunting. Suomen Riista 55: 83–96. [In Finnish with English Summary]

Helle I., Ahtiainen H., Luoma E., Hänninen M. & Kuikka S. 2015. A probabilistic approach for a cost-benefit analysis of oil spill management under uncertainty: A Bayesian network model for the Gulf of Finland. Journal of Environmental Management 158: 122–132.

Helle I., Lecklin T., Jolma A. and Kuikka S. 2011. Modeling the effectiveness of oil combating from an ecological perspective - A Bayesian network for the Gulf of Finland; the Baltic Sea. Journal of Hazardous Materials 185: 182–192.

Helsinki Commission. 2010. Integrated wildlife response planning in the Baltic Sea. HELCOM Recommendation 31E/6. Helsinki Commission, Helsinki.

Helsinki Commission. 2001. Restricted use of chemical agents and other non-mechanical means in oil combating operations in the Baltic Sea. HELCOM Recommendation 22/2. Helsinki Commission, Helsinki.

Henkel J. R., Sigel B. J. & Taylor C. M. 2012. Large-Scale Impacts of the Deepwater Horizon Oil Spill: Can Local Disturbance Affect Distant Ecosystems through Migratory Shorebirds? BioScience 62: 676–685.

Hietala M. & Lampela K (Eds). 2007. Oil pollution preraredness on the open sea – Final report of the working group. Suomen ympäristö 41: 7–39. [In Finnish with English Summary]

Hildén M., Furman E. & Kaljonen M. 2004. Views on planning and expectations of SEA: the case of transport planning. Environmental Impact Assessment Review 24: 519–536.

Hildén M., Jordan A. & Rayner T. 2014. Climate policy innovation: developing an evaluation perspective. Environmental Politics 23: 884–905.

Holmes G. 2014. Australia's pesticide environmental risk assessment failure: The case of diuron and sugarcane. Marine Pollution Bulletin 88: 7–13.

de la Huz R., Lastra M., Junoy J., Castellanos C. & Viéitez J. 2005. Biological impacts of oil pollution and cleaning in the intertidal zone of exposed sandy beaches: Preliminary study of the "Prestige" oil spill. Estuarine Coastal and Shelf Science 65: 19–29.

Hyvättinen H. & Hildén M. 2004. Environmental policies and marine engines – effects on the development and adoption of innovations. Marine Policy 28: 491–502.

Hänninen M. & Kujala P. 2012. Influences of variables on ship collision probability in a Bayesian belief network model. Reliability Enginering and System Safety 102: 27–40.

Hänninen J., Vuorinen I., Helminen H., Kirkkala T. & Lehitlä K. 2000. Trends and Gradients in Nutrient Concentrations and Loading in the Archipelago Sea, Northern Baltic, in 1970–1997. Estuarine, Coastal and Shelf Science 50: 152–171.

Ihaksi T., Kokkonen T., Helle I., Jolma A., Lecklin T. & Kuikka S. 2011. Combining Conservation Value, Vulnerability, and Effectiveness of Migitation Actions in Spatial Conservation Decisions: An Application to Coastal Oil Spill Combating. Environmental Management 47: 802–813.

Incardona J., Carls M., Day H., Sloan C., Bolton J., Colloer T. & Scholz. N. 2009. Cardiac arrhytmia is the primary response of embryonic Pacific herring (*Clupea pallasi*) exposed to crude oil during weathering. Environmental Science & Technology 43: 201–207.

Incardona J., Collier T. & Scholz N. 2003. Defects in cardiac function precede morphological abnormalities in fish embryos exposed to polycyclic aromatic hydrocarbons. Toxicology and applied Pharmacology 196: 191–205.

Jakeman A.J., Letcher R.A. and Norton J.P. 2006. Ten iterative steps in development and evaluation of environmental models. Environmental Modelling & Software 21: 602–614.

Jensen F. & Nielsen T. 2001. Bayesian networks and decision graphs – statistics for engineering and information science. Springer Verlag, New York.

Jensen, J. R., Halls J.N. & J. Michel. 1998. A systems approach to Environmental Sensitivity Index (ESI) mapping for oil spill contingency planning and response. Photogrammetric Engineering and Remote Sensing 64: 1003–1014.

Jolma A., Lehikoinen A., Helle I. & Venesjärvi. 2014. A software system for assessing the spatially distributed ecological risk posed by oil shipping. Environmental Modelling & Software 61: 1–11.

Jordan A., Huitema D., Hildén M., van Asselt H., Rayner T., Schoenefeld J., Tosun J., Forster J. & Boasson E. Emergence of polycentric climate governance and its future prospects. Nature Climate Change 5: 977–982.

Juhasz A. & Naidu R. 2000. Bioremediation of high molecular weight polycyclic aromatic hydrocarbons: a review of the microbial degradation of benzo[*a*]pyrene. International Biodeterioration & Biodegradation 45: 57–88.

Kahneman D. & Tversky A. 1979. Prospect Theory: An analysis of Decision under Risk. Econometrica 47: 263–292.

Kennish M. 1997. Practical handbook of estuarine and marine pollution. CRC Press, Boca Raton.

Kokkonen T., Ihaksi T., Jolma A. & Kuikka S. 2010. Dynamic mapping of coastal values to support the prioritization of coastal oil combating. Environmental Modelling & Software 25: 248–257.

Korpinen S., Meski L., Andersen J.H. & Laamanen M. 2012. Human pressures and their potential impact on the Baltic Sea ecosystem. Ecological Indicators 15: 105–114.

Kostka J., Prakash O., Overholt W., Green S., Freyer G., Canion A., Delgardio J., Norton N., Hazen T. & Huettel M. 2011. Hydrocarbon-Degrading Bacteria and the Bacterial Community Response in Gulf of Mexico Beach Sands Impacted by the Deepwater Horizon Oil Spill. Applied and Environmental Microbiology 77: 7962–7974.

Krueger T., Page T., Hubacek K., Smith L. & Hiscock K. 2012. The role of expert opinion in environmental modelling. Environmental Modelling & Software 36: 4–18.

Kuhnert P.M., Martin T.G. & Griffiths S.P. 2010. A guide to eliciting and using expert knowledge in Bayesian ecological models. Ecology Letters 13: 900–914.

Kuikka S., Vanhatalo J., Pulkkinen H., Mäntyniemi S. & Corander J. 2014. Experiences in Bayesian Inference in Baltic Salmon Management. Statistical Science 29: 42–49.

Kuparinen A., Keith, D. & Hutchings. J. 2014. Allee effect and the uncertainty of population recovery. Conservation Biology 3: 790–798.

Lavorel S., McIntyre S., Landsberg J. & Forbes T. 1997. Plant functional classification: from general groups to specific groups based on response to disturbance. Trends Ecology & Evolution 12: 474–478.

Lecklin T., Ryömä R. & Kuikka S. 2011. A Bayesian network for analyzing biological acute and long-term impacts of oil spill in the Gulf of Finland. Marine Pollution Bulletin 62: 2822–2835.

Lee R. 2003. Photo-oxidation and Photo-toxicity of Crude and Refined Oils. Spill Science & Technology Bulletin 8: 157–162.

Lehikoinen A. & Virkkala R. 2015. North by north-west: climate change and directions of density shifts in birds. Global Change Biology, In press.

Lehikoinen A., Helle I., Klemola E., Mäntyniemi S., Kuikka S. and Pitkänen H. 2014. Evaluating the impact of nutrient abatement measures on the ecological status of coastal waters: a Bayesian network for decision analysis. International Journal of Multicriteria Decision Making, 4: 114–134.

Lehikoinen A., Hänninen M., Storgård J., Luoma E., Mäntyniemi S. & Kuikka S. 2015. A Bayesian Network for Assessing the Collision Induced Risk of an Oil Accident in the Gulf of Finland. Environmental Science and Technology 49: 5301–5309.

Leiger R., Aps R., Kotta J., Orviku Ü., Pärnoja M. & Tönisson H. 2012. Relationship between shoreline substrate type and sensitivity of seafloor habitats at risk to oil pollution. Ocean & Coastal Management 66: 12–18.

Leppäranta M. & Myrberg K. 2009. Physical Oceanography of the Baltic Sea. Springer-Verlag, Berlin.

Lopes A., da Rosa-Osman S.M. &Piedade M.T.F. 2009. Effects of crude oil on survival, morphology, and anatomy of two aquatic macrophytes from the Amazon floodplains. Hydrobiologia 636: 295–305.

Lundqvist C. & Granek E. 2005. Strategies for successful marine conservation: integrating socio-economic, political, and scientific factors. Conservation Biology 19: 1771–1778.

Mace G.M., Collar N.J., Gaston K.J. et al. 2008. Quantification of extinction risk: IUCN's system for classifying threatened species. Conservation Biology 22: 1424–1442.

Marris E. 2007. Conservation priorities: What to let go. Nature 450: 152–155.

Martin T., Burgman M., Fidler F., Kuhnert P., Low-Choy S., McBride M. & Mengersen K. 2012. Eliciting Expert Knowledge in Conservation Scinece. Conservation Biology 26: 29–38.

Marty J. (Eds.) 2014. Risk Assessment for Marine Spills in Canadian Waters. Final Study Report from WSP Canada, Transport Canada, 172 p.

Mazaheri A., Montewka J., Nisula J., & Kujala P. 2015. Usability of Accident and Incident Reports for Evidence-Based Risk Modeling - A case study on ship grounding reports. Safety Science 76: 202–214.

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

Mickwitz P. & Birnbaum M. 2009. Key insights for the design of environmental evaluations. New Directions for Evaluation 122: 105–112.

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

Minteer B. A. & Miller T. R. 2011. The New Conservation Debate: Ethical foundations, strategic trade-offs, and policy opportunities. Biological Conservation. 144: 945–947.

Mittmeier R., Gil P., Hoffman M., Pilgrim J., Brooks T. Mittmeier C., Lamoreux J. Fonseca G. 2005. Hotspots revisited: Earth's biologically richest and most endangered terrestrial ecoregions. CEMEX, Nuevo Leon.

Moilanen A., Wilson K.A. & Possingham H.P. 2009. Spatial Conservation Prioritization: Quantitative Methods and Computational Tools. Oxford University Press, New York.

Moles A. 1998. Sensitivity of 10 aquatic species to long-term crude oil exposure. Bulletin of Environmental Contamination and Toxicology 61: 102–107.

Morales-Caselles C., Jiménez-Tenorio N., Canales M., Sarasquete C. & DelValls T. 2006 Ecotoxicity of sediments contaminated by the oil spill associated with the tanker prestige using juveniles of the fish *Sparus aurata*. Archives of Environmental Contamination and Toxicology 51: 652–660.

Morgan M. & Henrion M. 1990. Uncertainty: a guide to dealing with uncertainty in quantitative risk and policy analysis. Cambridge University Press, Cambridge.

Mouillot D., Graham N., Villéger S., Mason N. & Bellwood D. 2013. A functional approach reveals community responses to disturbances. Trends in Ecology and Evolution 28: 167–177.

Mulligan M. 2014. Models Supporting Decision-Making and Policy Evaluation. In: Wainwright J. & Mulligan M. (Eds.) Environmental Modelling: Finding Simplicity in Complexity, 2nd Edition. John Wiley & Sons, Chichester.

Yim N.H., Kim S.-H., Kim H.-W. & Kwahk K.-Y. 2004. Knowledge based decision making on higher level strategic concerns: system dynamics approach. Expert Systems with Applications 27: 143–158.

National Research Council. 2003. Oil in the Sea III: Inputs, Fates and Effects. National Academic Press, Washington.

Neil M., Fenton N. & Tailor M. 2005. Using Bayesian networks to model expected and unexpected operational losses. Risk Analysis 25: 963–972.

Nishihiro J., Nishihiro M. & Washitani I. 2006. Assessing the potential for recovery of lakeshore vegetation: species richness of sediment propagule banks. Ecological Research 21: 436–445.

O'Hagan A. & Oakley J.E. 2004. Probability is perfect, but we can't elicit it perfectly. Reliability Engineering and System Safety 85: 239–248.

O'Hagan A., Buck C.E., Daneshkhah A., Eiser J.R., Garthwaite P.H., Jenkinson D.J., Oakley J.E. & Rakow T. 2006. Uncertain Judgements: Eliciting Expert Probabilities. John Wiley and Sons, Chichester.

Ort B. S., Cohen C. S., Boyer K. E., Reynolds L. K. Tam S. M. Wyllie-Echeverria S. 2014. Conservation of eelgrass (*Zostera marina*) Genetic Diversity in a Mesocosm-Based Restoration Experiment. PLOS One 9: 89316.

Ozimek T. 2006. The possibility of submerged macrophyte recovery from a propagule bank in the eutrophic Lake Mikolajskie (North Poland). Hydrobiologia 570: 127–131.

Pahl J., Mendelssohn I., Henry C. & Hess T. 2003. Recovery trajectories after in situ burning of an oiled wetland in coastal Louisiana, USA. Environmental Management 31: 236–251.

Pearce-Higgins J. & Green R. 2014. Birds and Climate Change: Impacts and Conservation Responses. Cambridge University Press, Cambridge.

Pearce J., Cherry K., Drielsma M., Ferrier S. & Whish G. Incorporating expert opinion and fine-scale vegetation mapping into statistical models of faunal distribution. Journal of Applied Ecology 38: 412–424.

Petersen M., Kyster-Hansen H., McDaniel J., Cardebring P., Meyer-Rühle O., Sirén, J., Räsänen J., Nyberg J., Kajander S., Saurama A. & Wolek, M. 2011. Baltic Transport Outlook 2030. Tetraplan A/S, Copenhagen, 37 p.

Peterson C., Anderson S., Cherr G., Ambrose R., Anghera S., Bay S., Blum M., Condon R., Dean T., Graham M., Guzy M., Hampton S., Joye S., Lambrinos J., Mate B., Meffert D., Powers S., Somasundaran P., Spes R., Taylor C., Tjreerdema R. & Adams E. A Tale of Two Spills: Novel Science and Policy Implications of an Emerging New Oil Spill Model. BioScience 62: 461–469.

Peterson C.H. 2001. A synthesis of direct and indirect or chronic delayed effects of the Exxon valdez oil spill. Advances in Marine Biology 39: 1–103.

Peterson C., Rice S., Short J., Esler D., Bodkin J., Ballachey B. & Irons D. 2003. Long-Term ecosystem Response to the Exxon Valdez Oil Spill. Science 302: 2082–2086.

Pettersson H. 2007. Wave climate in the northern Baltic proper and in the Gulf of Finland 2006. In: Olsonen R. (Ed.) FIMR monitoring of the Baltic Sea environment – Annual report 2006. Meri Report Series of the Finnish Institute of Marine Research 59: 13–18.

Pezeshki S., Hester M., Lin Q. & Nyman J. 2000. The effects of oil spill and clean-up on dominant US Gulf coast marsh macrophytes: a review. Environmental Pollution 108: 129–139.

Piatt J. & Ford R. 1996. How many birds were killed by the Exxon Valdez oil spill? American Fisheries Society Symposium 18: 712–719.

Piatt J., Lensink C., Butler W., Kendziorek M. & Nysewander D. 1990. Immediate impact of the Exxon Valdez oil-spill on marine birds. Auk 107: 387–397.

Pita C., Pierce G. J. & Theodossiou I. 2010. Stakeholders' participation in the fisheries management decision-making process: Fishers' perceptions of participation. Marine Policy 345: 1093–1102.

Pulkkinen H., Mäntyniemi S., Kuikka S. & Levontin P. 2011. More knowledge with the same amount of data: advantage of accounting for parameter correlations in hierarchical meta-analyses. Marine Ecology Progress Series 443: 29–37.

Rahikainen M., Helle I., Haapasaari P., Oinonen S., Kuikka S., Vanhatalo J., Mäntyniemi S. and Hoviniemi K.-M. 2014. Towards Integrative Management Advice of Water Quality, Oil Spills, and Fishery in the Gulf of Finland: A Bayesian Approach. AMBIO 43: 115–123.

Rahikainen M, Hoviniemi K.-M., Mäntyniemi S., Vanhatalo J., Helle I., Lehtiniemi M., Pönni J., Kuikka S. Impacts of eutrophication and oil spills on the Gulf of Finland herring stock. Marine Ecology Progress Series, Submitted.

Ralls K. & Taylor B. 2000. Better policy and management decisions through explicit analysis of uncertainty: New approaches from marine conservation - Introduction. Conservation Biology 145: 1240–1242.

Reece J., Urry L., Cain M., Wasserman S., Minorsky P. & Jackson P. 2014. Campbell Biology. Benjamin Cummins, San Francisco.

Regan H., Ben-Haim Y., Langford B., Wilson W.G., Lundberg P., Andelman S.J. & Burgman M.A. 2005. Robust decision-making under severe uncertainty for conservation management. Ecological Applications 15: 1471–1477.

Rose N. A. & Parsons E. C. 2015. "Back off, man, I'm a scientist!" When marine conservation science meets policy. Ocean & Coastal Management 115: 71–76.

Rydén L., Migula P. & Andersson M. (Eds.) 2003. Environmental science: understanding, protecting and managing the environment in the Baltic Sea region. Baltic University Press, Uppsala.

Ryzhik I. V. 2012. The metabolic activity of cells of *Fucus vesiculosus* Linnaeus, 1753 (*Phaeiohyta: Fucales*) from the Barents Sea conditions of oil pollution. Russian Journal of Marine Biology 38: 96–99.

Sánchez F., Velasco F., Cartes J., Olaso I., Preciado I., Fanelli E., Serrano A. & Gutierrez-Zabala J. 2006. Monitoring the Prestige oil spill impacts on some key species of the Northern Iberian shelf. Marine Pollution Bulletin 53: 332–349.

Sarkar S., Justus J., Fuller T., Kelley C., Garson J. & Mayfield M. 2005. Effectiveness of Environmental Surrogates for the selection of Conservation Area Networks. Conservation Biology 19: 815–825.

Scholz D., Michel J., Shigenaka G. & Hoff R. 1992. Biological resourses. In: Hayes M., Hoff R., Michel J., Scholz D. & Shigenaka G. Introduction to coastal habitats and biological resources for spill response, report HMRAD 92-4. National Oceanic and Athmospheric Administration, Seattle.

Short J. 2003. Long-term effects of crude oil on developing fish: Lessons from the Exxon Valdez oil spill. Energy Sources 25: 509–517.

Skinner D.J.C., Rock, S.A. & Pollard S.J.T. 2014. A review of uncertainty in environmental risk: characterizing potential natures, locations and levels. Journal of Risk Research 17: 195–219.

Soomere T., Berezovski M., Quak E. & Viikmae B. 2011. Modelling environmentally friendly fairways using Lagrangian trajectories: a case study for the Gulf of Finland, the Baltic Sea. Ocean Dynamics 61: 1669–1680.

Stelzenmüller V., Garnacho L. & Rogers S. 2010. Assessment of a Bayesian Belief Network-GIS framework as a practical tool to support marine planning. Marine Pollution Bulletin 60: 1743–1754.

Stephens P., Sutherland W. & Freckleton R. 1999. What is Allee Effect? Oikos 87: 185–190.

Thompson K., Hillier S.H., Grime J.P., Bossard C.C. & Band S.R. 1996. A functional analysis of a limestone grassland community. Journal of Vegetation Science 7: 371–380.

Tortell P. 1992. Coastal zone sensitivity mapping and its role in marine environmental management. Marine Pollution Bulletin 25: 88–93.

Uusitalo L. 2007. Advantages and challenges of Bayesian networks in environmental modelling. Ecological Modelling 203: 312–318.

Uusitalo L., Kuikka S., Kauppila P., Söderkultalahti P. & Bäck S. 2012. Assessing the roles of environmental factors in coastal fish production in the northern Baltic Sea: a Bayesian network application. Integrated Environmental Assessment and Management 8: 445–455.

Uusitalo L., Lehikoinen A., Helle I., Myrberg K. 2015. An overview of methods to evaluate uncertainty of deterministic models in decision support. Environmental Modelling & Software 63: 24–31.

Van der Laag L., Renooij S., Witteman C., Aleman B. & Taal B. 2002. Probabilities for a probabilistic network: a case study in oesphageal cancer. Artificial Intelligence in Medicine 25: 123–148.

Varis, O. & Kuikka, S. 1999. Learning Bayesian Decision Analysis by Doing: Lessons from Environmental and Natural Resources Management. Ecological Modelling 119: 177–195.

Votier S., Hatchwell B., Beckerman A., McCleery R., Hunter F., Pellatt J., Trinder M., Birkhead T. & Coulson T. 2005. Oil pollution and climate have wide-scale impacts on seabird demographics. Ecology Letters 8: 1157–1164.

Wan Z., Zhu M., Chen S. & Sperling D. 2016. Pollution: Three steps to a green shipping industry. Nature 530: 275–277.

Wiedmer M., Fink M., Stegeman J., Smolowitz R. Marty G. & Hinton F. 1996. Cytochrome P-450 induction and histo-pathology in pre-emergent pink salmon from oiled spawning sites in Prince William Sound. American Fisheries Society Symposium 18: 509–517.

Wirtz, K. W. & X. Liu. 2006. Integrating economy, ecology and uncertainty in an oil-spill DSS: The Prestige accident in Spain, 2002. Estuarine Coastal and Shelf Science 70:525–532.

Wood M. & Heaphy N. 1991. Rehabilitation of oiled seabirds and bald eagles following the Exxon Valdez oil spill. Proceedings of the 1991 Oil Spill Conference, March 4-7 1991, San Diego, pp. 263–266.

Yamada K., Eltih J., McGarthy M. & Zerger A. 2003. Eliciting and integrating expert knowledge for wildlife habitat modeling. Ecological Modelling 165: 251 - 264.

Zhang D., Yan X.P., Yang Z.L., Wall A. & Wang J. 2013. Incorporation of formal safety assessment and Bayesian network in navigational risk estimation of the Yangtze River. Reliability Engineering & System Safety 118: 93–105.

The Zoological Society of London. 2015. Living Blue Planet Report: Species, habitats and human well-being. The Zoological Society of London, London.

Abbreviations used in the text

AIS	Automatic Identification System
AS	Archipelago Sea
BN	Bayesian network
BORIS	The national situation awareness system for environmental emergency response
BSAP	Baltic Sea Action Plan
CPT	Conditional probability table
DAG	Directed acyclic graph
ENSI	Enhanced Navigation Support System
ERA	Environmental risk assessment
ERM	Environmental risk management
EU	European Union
FinBIF	Finnish Biodiversity Information Facility
FSA	Formal Safety Assessment
GES	Good Environmental State
GIS	Geographical Information System
GOF	Gulf of Finland
GOFREP	The mandatory ship reporting system in the Gulf of Finland
HELCOM	Baltic Marine Environment Protection Commission (Helsinki Commission)
IBA	Important Bird Area
ID	Influence diagram
IMO	International Maritime Organization
MARPOL	International Convention for the Prevention of Pollution from Ships
MCMC	Monte Carlo Markov Chain
MPA	Marine Protected Area
MSFD	Marine Strategy Framework Directive
MSP	Marine spatial planning
NGO	Non-Governmental Organization
OCIMF	Oil Companies International Marine Forum
ODME	Oil Discharge Monitoring Systems and Equipment
OPCF	Oil Pollution Compensation Fund
PAH	Polycyclic aromatic hydrocarbons
P&I	Protection and indemnity insurance
PSC	Port State Control
PSSA	Particularly Sensitive Sea Area
SDSS	Spatial decision support system
SOLAS	International Convention for the Safety of Life at Sea
UNFCCC	United Nations Framework Convention on Climate Change
VTS	Vessel Traffic Services
WAF	Water-accommodated fraction

Explanations for some concepts

Allee effect	A decline in individual fitness at a low population size or density that can result in critical population thresholds below which populations crash to extinction.
Bayesian network	Graphical models that represent relationships among uncertain variables, in which probabilities may be estimated subjectively and updated using Bayes' theorem.
Contingency plan	A partial process of disaster preparedness planning devised for an outcome other than in the usual expected plan and it is developed to explore and prepare for any eventuality.
Deterministic model	A model in which there is no representation of random variability in data nor description of uncertainty of results.
Epistemic uncertainty	Reflects incomplete knowledge including measurement error, systematic error, natural variation, model uncertainty and subjective judgment.
Habitat type	Terrestrial or aquatic unit that consists of an aggregation of habitats having equivalent structure, function and responses to disturbance.
Functional trait	Defines species in terms of their ecological roles – how they interact with the environment and other species.
LC_{50} value	Lethal threshold concentration at which 50% of individuals die within a specific time.
Life-history parameter	Traits that affect the life table of an organism and can be defined as various investments in growth, reproduction and survival.
Markov Chain Monte Carlo	Useful in analyzing large and complicated data sets with some form of hierarchical structure, such as life-history parameters in natural populations. It can be used in conjunction with Bayesian prior distributions to estimate parameters from data.
Policy instrument	Describes the methods used by governments to achieve a desired policy effect.
Preparedness plan	Disaster preparedness planning includes the identification of risks, developing practices as well as implementing action plan to have the best possible readiness in case of disaster.
Recovery potential	An individual's, species' or habitat type's potential to recover its pre-disturbance state.
Risk analysis	Evaluation and communication of the nature and extent of uncertainty.
Stochastic model	A model in which some of the parameters are drawn from statistical distributions or in which there is some other description of uncertainty.