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Magnus Sparrevik

Methods for Sustainable Management of Contamination Sources in Urban Coastal Areas

Doctoral thesis

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Department of Industrial Economics
and Technology Management

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Trondheim, April 2012

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To raise new questions, new possibilities, to regard old problems from a new angle, requires creative imagination and marks real advance in science

Albert Einstein

Abstract

Managing contaminated urban coastal areas is an important issue in today's society. Harbor areas are transformed from industrial sites and shipyards, to housing areas with high environmental requirements. The use of coastal areas for aquaculture and fishing activities increases the need for cleanup of previous contamination sources. High environmental standards when handling dredged material from harbors and industrial activities are also required in our society today.

Many countries including Norway have based the management for the coastal zone environment on use of health and ecological risk assessments (HERA). This means that there should be no adverse risk to human or ecological life due to exposure of contaminants from sediments or water. This single criteria framework based on the precautionary principle is inherently conservative and may introduce costly and resource consuming remediation methods with isolated focus on sediment disposal instead of beneficial use.

In order to take more balanced management remedial decisions this thesis promotes a shift in management of contaminated areas from use of HERA alone to a multicriteria focused approach incorporating sustainable values.

The main contributions of this thesis are:

- Knowledge on how the present Norwegian management system deviates from a holistic risk governance concept.
- Wider understanding of how social factors as risk perceptive values are influencing management processes.
- A life cycle impact assessment model (LCIA) for marine sediment contamination allowing use of life cycle assessments for contaminated sediment problems.
- A multicriteria involvement model (MIP) to promote participatory involvement processes
- An integrative stochastic multicriteria decision model (SMCA) supporting sustainable decisions in contaminated sediment management.

Preface

This thesis is submitted to the Norwegian University of Science and Technology (NTNU) for partial fulfillment of the requirements for the degree of philosophiae doctor.

The thesis consists of a summary and five research papers. The summary gives an introduction to the topic, describes research methodology and discusses and evaluates the answers to the research questions. The complete description of the research outcome is given in the papers in appendix A. The reader is recommended to study both parts to get a full understanding of the topic and research answers presented in this work.

The doctoral work has been performed at the Department of Industrial Economics and Technology Management, NTNU, Trondheim, with Professor Annik Magerholm Fet as main supervisor and with co-supervision from Professor Jan Hovden and Professor Gijs Breedveld, University of Oslo.

The thesis has been financed by the Norwegian Geotechnical Institute as a part of the internal research fund for education.

The work has also been supported by two research projects from the Norwegian Research Council; Sediment and Society (184928/S40) and Opticap (182720/I40). The thesis is also a part of the risk management activities at the International Centre of Geohazards.

Through an internal sabbatical there has also been cooperation with Environmental Laboratory, U.S Army Research and Development Centre Concord, Massachusetts, USA.

Oslo, October 5 2011

Magnus Sparrevik

Acknowledgements

Working with this thesis has been a privilege since it has involved multiple contacts with stakeholders, governmental authorities and research colleagues in Norway and abroad. Without their contribution this collaborative research work could not have been performed. Even though many persons have been involved, there are some people whose contribution I would like to mention particularly.

First I would like to thank my supervisors; Professor Annik Magerholm Fet; Professor Jan Hovden and Professor Gijs Breedveld for their support. Even though I think I have been quite self-sustained you have supported me all the way through the PhD. The competence on how to reflect and discuss scientific work is something I have learned from you and it will be important in my future research career.

I would sincerely like to thank Dr Igor Linkov and colleagues for the support and advice during the sabbatical in Boston, USA. Dr Linkov's work has been an inspiration for the thesis and the productivity in this cooperation has been far beyond expectations.

Very important has been the support from colleagues and contributors in the "Sediment & Society" research project. Among all contributors I would especially thank Dr David Barton, Dr Amy Oen and Gerard Jan Ellen for giving substantial contributions to the papers in the thesis.

Equally important are the contributions from the "Opticap" research project, and especially Dr Tuomo Saloranta, Dr Espen Eek, Dr Gerard Cornelissen, Dr Bjørn Vidar Vangelsten, Dr Morten Schaanning, Dr Hans Peter Arp and Dr Sarah Hale.

A special gratitude goes to Tim Gregory, NGI for providing illustrations to the thesis.

I am also deeply grateful to NGI for financing the work. The possibility for taking a PhD within a normal employment position is a unique opportunity and is the envy of colleagues at other places.

Gratitude goes also to the Department of Industrial Economics and Technology Management NTNU for the administration of the PhD.

Last, but not least I would like to thank my wife for making it possible for me to travel on several tours to Trondheim and conducting the USA sabbatical.

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Abbreviations

AC	Activated Carbon
AHP	Analytical Hierarchy Process
ANOVA	Analysis Of Variance
CAD	Confined Aquatic Disposal
CBA	Cost-Benefit Analysis
CDF	Cumulative Distribution Function
CE	Contingent Evaluation
CEA	Cost-Effectiveness Analysis
CSR	Corporate Social Responsibility
EIF	Environmental Impact Factor
EVPI	Expected Value of Perfect Information
HERA	Health and Ecological Risk Assessments
HIFC	High Inner Fjord Capping
HOFC	High Outer Fjord Capping
IFC	Inner Fjord Capping
IRGC	International Risk Governance Council
LCA	Life Cycle Assessments
LCI	Life Cycle Inventory
LCIA	Life Cycle Impact Assessment
MAUT	Multi Attribute Utility Theory
MAVT	Multi Attribute Value Theory
MCA	Multicriteria Analysis
MCDA	Multicriteria Decision Analysis
MIP	Multicriteria Involvement Process
NR	Natural Recovery
NTNU	Norwegian University of Science and Technology
OFC	Outer Fjord Capping
PAF	Perceptive Affecting Factors
PAF	Potential Affected Species
PAH	Polycyclic Aromatic Hydrocarbons
PCA	Principal Component Analysis
PNEC	Probable No Effect Concentrations
RA	Risk Analysis
SEM	Structural Equation Modeling
SMCA	Stochastic Multicriteria Analysis
SQG	Sediment Quality Guideline values
SSD	Species Sensitivity Distribution
TCDD	2,3,7,8-Tetrachlorodibenzo-p-dioxin
WFC	Whole Fjord Capping
WFD	Water Framework Directive
WQG	Water Quality Guideline values
WTP	Willingness to Pay

1. Introduction

1.1 Problem Outline and Research Context

Managing contaminated urban coastal areas is an important issue in today's society. Harbor areas are transformed from industrial sites and shipyards, to housing areas with high environmental requirements. The use of coastal areas for aquaculture and fishing activities increases the need for remediation of contamination from previous industrial activities. High environmental standards when handling dredged material from harbors and industrial activities are also required in our society today necessitating management of large quantities of potentially contaminated dredged material[1].

In 2001 the Ministry of environment concluded in the white paper nr 12[2] on a strategy for management of contaminated sediments in Norway. In 2006 the plan was followed by white paper nr 14[3]. Here 17 fjords and harbor areas in Norway have been prioritized for further measures due to sediment contamination, and costs of NOK 800 million – 2 billions are estimated to accomplish the remediation goals. A few projects are in the final stage of execution today, but most of the work is still in the planning phase. In an international perspective the focus on river basin management and coastal area protection is increasing. The EU Water Frame Directive[4] addresses the need for protection of water flowing into the recipient basins including management of contamination sources.

Many countries including Norway have based the management for the coastal zone environment on use of health and ecological risk assessments (HERA). This means that there should be no adverse risk to human or ecological life due to exposure of contaminants in sediments or water. The difficulties with HERA, which is inherent precautionary is the degree of conservatism leading to the risk of imposing very strict guidelines[5]. This can lead to obvious problems in the management process such as:

- Introduction of costly and resource consuming remedial methods with limited environmental benefit
- Creating a disproportional difference between the objectively estimated risk and the public perceptive values
- Giving isolated focus on sediment disposal instead of beneficial use

By focusing more on other aspects than health and ecological risk a more balanced decision can be taken[6]. This demand for broadening the view on contaminated sediment management and development in a sustainable direction is the background for the PhD work as described in this thesis.

This thesis promotes a shift in management of contaminated areas from use of single criteria decisions in terms of health and ecological risk reduction to a more sustainable multicriteria focused approach, Figure 1. Here, the definition of sustainability often described as the triple bottom line definition or “pillars” of sustainability; concern for environmental, economic and social aspects is used[7]. The decision methodology proposed therefore reflects these three basic aspects. The thesis also recognizes that the present management process is risk driven. Risk assessments encompasses the analysis and evaluation of risk[8] and serves as the foundation of the HERA framework. This thesis is however especially focused on management aspects, and therefore investigates how risk assessments can be included in a broader risk governance perspective. This incorporates the totality of actors, rules and conventions about analyzing, communicating and managing risk[9] .

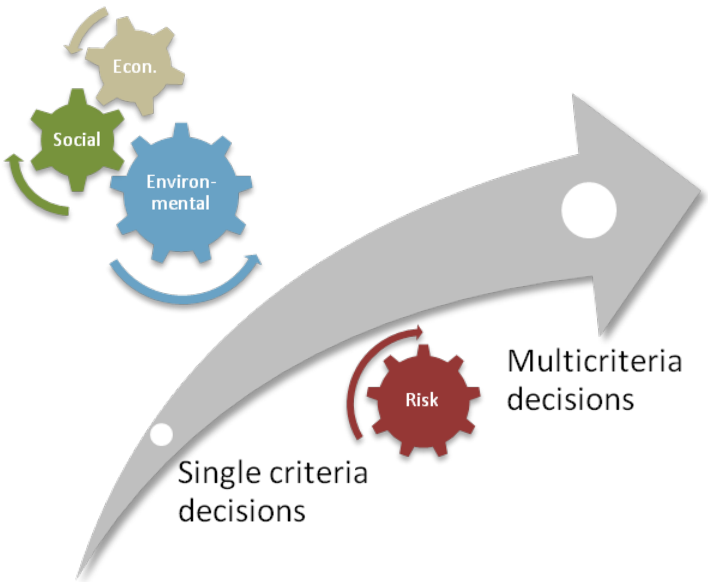

Figure 1 Research context for the thesis

Sediments as a source of contamination in urban coastal areas have been given the primary focus in the thesis since it is especially focused within the regulatory framework. The findings are however often of a more general nature and may be applicable to other multi-dimensional environmental decision problems requiring a sustainable approach.

The thesis has focused on the Norwegian risk based management system for contaminated sediments[10] as a model for a single criteria decision framework. The Norwegian management system is however representative for other countries and other applications. It is therefore possible to generalize results since the thesis focuses on the methodological aspects, although Norwegian data have been used to illustrate and evaluate the proposed models.

Three harbor areas have been used as study objects for the thesis:

- Oslo harbor which has been subjected to a remediation project removing 300.000 m^3 of sediments from the inner harbor area disposing them in a confined aquatic disposal site (CAD). This project has been selected in the thesis due to the substantial social unrest connected to the selected disposal solution.
- Bergen harbor is one of the next upcoming harbors for remediation. Here several aspects have to be balanced when selecting appropriate strategies for further management. For example, the harbor area is included in the UNESCO list of world heritage sites, requiring a balance between ecological objectives of sediment quality and the selection of technologies not depreciating the historical value of the harbor area.
- The Grenland fjord is subjected to contamination due to previous industrial activities. The magnitude of the area (58 km^2) implies that only capping of the seabed either through natural recovery processes or by active methods is feasible to reduce the fluxes of contamination in to the food chain. Potentially high cost, resource use and social concern calls for careful considerations to assess feasible management strategies.

The research has been conducted in cooperation with three external research projects supported by the Research Council of Norway:

- Sediment & Society (ID 184928/S40) whose overall objective is to recommend an integrated management strategy for stakeholder involvement that can be implemented within the existing national management framework for contaminated marine sediments.
- Opticap (ID 182720/I40) whose objective is to increase the knowledge of materials and methods suitable for capping of contaminated seabed to reduce contaminant spreading.
- International Centre of Geohazards (ICG) which is an international arena for conducting scientific and technological research on identification, assessment and mitigation of geohazards.

In addition the research has been conducted in close cooperation with researchers both in Europe and USA. Especially cooperation with the Environmental laboratory, US Army Engineer and Development Center (ERDC) and TNO in the Netherlands has been crucial for this research.

The research is conducted as an integrated part of research activities at the Environmental Management & Corporate Social Responsibility group at the Department of Industrial Economy and Technology Management, NTNU.

1.2 Research Questions

To propose a research question and to choose research methods in a way that offers the best chance to obtain useful answers is an important task in modern research[11]. This thesis recognizes weaknesses within a HERA based single criteria management system and therefore hypothesize a multicriteria decision framework to be a preferable alternative. The research question proposed for the thesis is therefore:

"*How can sustainable contaminated sediment management be facilitated and what are the feasible multicriteria decision models to be used?"*

This complex question can be investigated and discussed from various conceptual angles. The question contains a subjective goal to promote a paradigm shift in management methodology and an objective goal to investigate the feasibility of using multicriteria decision methodology to achieve this objective. In order to investigate the research question more in detail four specific research questions have been proposed:

- What are the gaps between HERA driven management of contaminated sediments and a holistic risk governance concept?
- How are risk perceptive values affecting social preferences and how to involve stakeholders in the management process?
- How can environmental impact be estimated holistically?
- How to structure a multicriteria decision model incorporating sustainable values?

The following section gives an overview of the performed research and structures the results in relation to the proposed research questions.

1.3 Outcome of Thesis

The research work has resulted in five publications published in, or submitted to peer-review journals, see Table 1. Three papers are presented in Environmental Science & Technology, which is ranked as level 2 (higher level) in the Norwegian classification system for registered publications, see http://dbh.nsd.uib.no/.

In addition thesis material is presented in three secondary papers, see Appendix B. The secondary papers are connected to thesis work, but are not central to the research questions, and therefore included for informatory purposes only.

Table 1 Publications included in the thesis

ID	Author	Title	Journal
P1	Sparrevik & Breedveld	From Ecological Risk Assessments to Risk Governance. Evaluation of the Norwegian Management System for Contaminated Sediments	IEAM[1]
P2	Sparrevik et al.	Evaluation of Factors Affecting Stakeholder Risk Perception of Contaminated Sediment Disposal in Oslo Harbor	ES&T[2]
P3	Sparrevik et al.	Use of Life Cycle Assessments to Evaluate the Environmental Footprint of Contaminated Sediment Remediation	ES&T
P4	Sparrevik et al.	Use of Multicriteria Involvement Processes to Enhance Transparency and Stakeholder Participation at Bergen Harbor, Norway	IEAM
P5	Sparrevik et al.	Towards Sustainable Decisions in Contaminated Sediment Management by use of Stochastic Multicriteria Analysis	ES&T (submitted)

1 Integrated Environmental Assessment and Management; Society of Environmental Toxicology and Chemistry
2 Environmental Science & Technology; American Chemical Society

In the first publication[12], the Norwegian model for management of contaminated sediments is compared against a generic system for risk governance encompassing both knowledge, legally prescribed procedures, and social values[13]. Important deviations are discussed and improvements are suggested.

The second publication[14] describes an analysis of the importance of risk perceptive values among stakeholders for the management decision in a much debated sediment remediation case in Oslo harbor Norway. Perceptive affecting factors (PAF) are identified and items to be addressed within a successful stakeholder involvement model are discussed.

In the third paper[15] environmental impacts, not only relating to the contamination itself, but also to energy and resource use connected with the remediation effort are investigated. The life cycle assessment (LCA) methodology is adapted to marine conditions and the environmental footprint of different sediment capping materials for remediation of the contaminated Grenland fjord in Norway are evaluated.

The fourth paper[16] describes a participatory “bottom-up” multicriteria involvement process (MIP) for involvement of stakeholders in contaminated sediment management and evaluates the model for future sediment remediation in Bergen harbor, Norway. The proposed methodology builds on the quantitative principles of multicriteria decision analysis, but also incorporates group interaction and learning through qualitative participatory methods.

Finally, in paper five[17] an integrative “top-down” decision model incorporating social, environmental and economical values in the decision by use of stochastic multicriteria analysis is proposed. This model incorporates the different analytical approaches like risk analysis (RA), life cycle assessment (LCA), multicriteria analysis (MCA) and economic valuation methods. The model is evaluated for the Grenland fjord remediation project.

The papers are highly interconnected and respond in plural to the research question and the sub questions as shown in Figure 2.

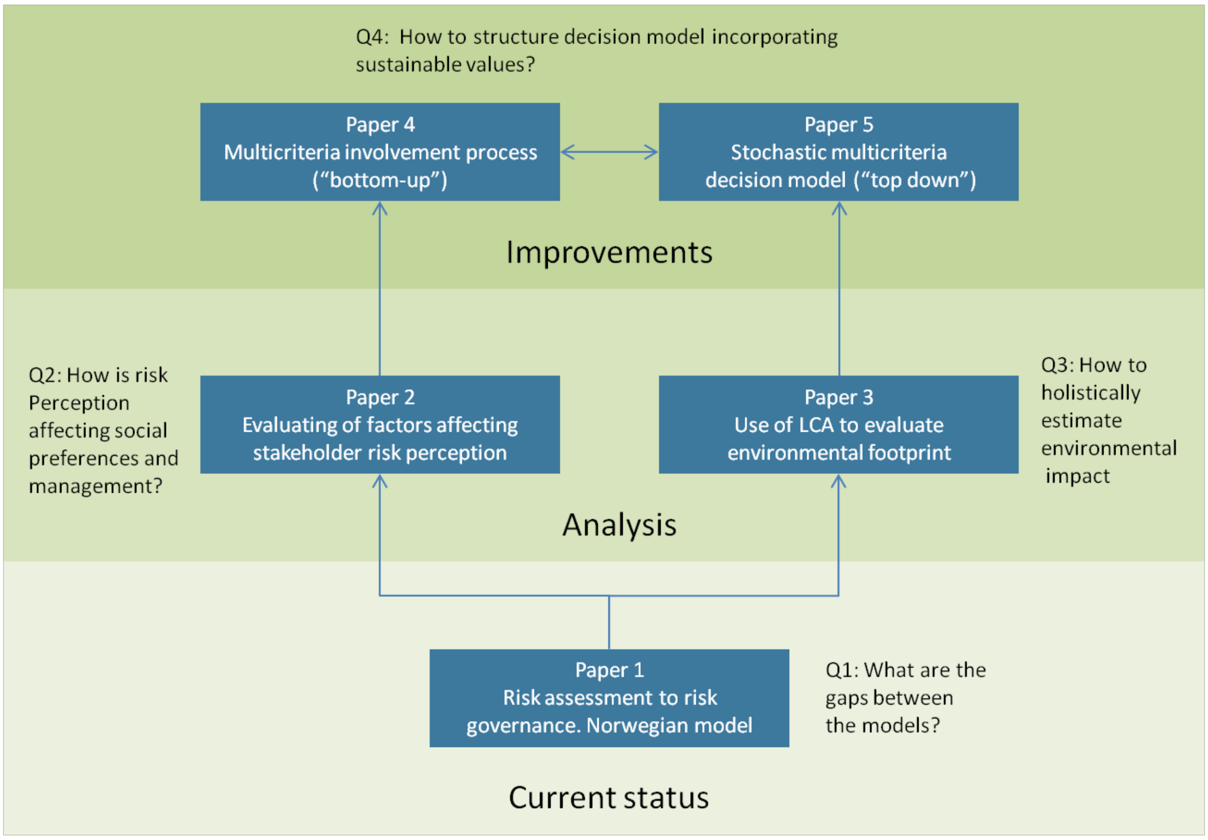

Figure 2 Structure of thesis and relation between the papers and the research questions

The work with the thesis is divided into three distinct phases; The thesis starts with an evaluation of the *current status* of the Norwegian management system. Weaknesses, compared to the holistic model are identified and subsequent research needs are outlined (paper 1, research question 1). In the second part of the work, research is conducted to further *analyze* specific topics as a foundation for proposing integrative models (paper 2 and 3, research question 2 and 3). In the final part of the work, *improvements* in the form of multicriteria decision models for contaminated sediment management are proposed (paper 4 and 5, research question 4).

1.4 Thesis Structure

The thesis is structured to present the conducted work in a logical and informative way. Theory is integrated within the thesis to give background for the discussion and evaluation of research results. Each chapter in the results section starts by outlining the theoretical foundations, continues with an explanation of the results from the corresponding papers and ends with a conclusion on how the papers have responded to the connected research question.

In all, the thesis consists of the following five sections:

- This *introductory section* (Chapter 1) gives a context for the work, poses the research questions and outlines the outcome of the thesis by briefly describing the content of the research papers and how they are linked internally and to the research questions.
- The *methodology section* (Chapter 2) introduces different methodological approaches used in this research and explains the research design and subsequent research methods used in the work.
- The *results section* (Chapter 3 to 6) is organized in chapters according to the different work phases explained in Figure 2; Chapter 3 relates to evaluation of the current status described in paper 1. Chapter 4 and 5 relate to the analysis of social and environmental issues described in paper 2 and 3. Chapter 6 relates to improvements of models described in paper 4 and 5.
- In the *discussion and conclusion section* (Chapter 7) the work is discussed in relation to the proposed research questions and selected research methodology. Major findings are summarized and discussed in relation to applicability in contaminated sediment management and need for future research is proposed.
- Key references are listed in the *reference section* (Chapter 8).

The papers are attached as Appendix A.

2. Research Methodology

2.1 Different World Views in Research

Before starting to discuss different research methods used in the thesis it is important to look at different views of epistemology, i.e. the relationship between the researcher and the research in the study. This world view is important since it will guide the researcher into the detailed research design. Creswell and Plano Clark[18] define epistemology in the terms of objectivism, constructivism and subjectivism;

Objectivism or positivist and post-positivist theory is characterized by a “top down” approach where the researcher works from a theory or hypothesis and uses data to add or contradict the theory. The researcher is objective, collecting and analyzing data. This is a common way of working in natural science with a long tradition of experimental work based on fundamental nature laws.

Constructivism is characterized by a different more “bottom-up” view. Here the views of the participants are important to construct broader themes, and theory is generated by the research to interconnect the themes. In constructivism the researcher still remains objective, but may use a variety of research methods to collect data.

In *subjective* research the researcher no longer remains objective in the process. In advocatory and participatory research, which is one kind of subjective research, the participant is actively engaged in the process. In pragmatic research which is another form of subjective research, both biased and unbiased perspectives are used to address the research question[19].

The research described in this thesis clearly bears the sign of all three epistemologies. The three harbor areas studied are central in theory building indicating a “bottom-up” approach typical for constructivism. Themes interconnect and theory is generated through studies of the management process within these harbor areas. However, both paper 4 and 5 are subjective, actively seeking use of new methodology to promote a shift from single criteria health and ecological focus towards sustainable multicriteria based decisions. At the same time there are also signs of objectivism in some parts of the thesis. The new LCA impact assessment modules developed in the thesis are based on experimental work and is an example of use of objective research methodology.

Use of research design and methods should therefore mainly be found within the constructive and subjective research view aided by methods typical for objective research.

2.2 Research Design and Methods - an Overview

Based on the epistemology of the research, different research design and research methods are selected to address the proposed research question. Here the word design, refers to the plan that links the world view of the researcher to the specific methods[19,20]. The methods which are used as a part of the research design are more specific, and may be defined as techniques for data collection and analysis[19,21].

The objective view is closely connected to collection of quantitative data in experiments or controlled measurements. The obvious value with experiments is that the controlled settings may reduce bias and give reproducible results. The weakness is that the controlled environment makes it difficult to transform results to the real world[22]. In natural science this problem is often overcome by combining laboratory experiments with field experiments.

The constructive view is still influenced by the objectiveness, even though the research question makes it difficult to perform the research as experiments. The typical design is survey based, where the researcher seeks correlation between research data and theory to test assumptions and to build new theory data[18]. Typical research methods may include observations where the researcher observes phenomena in its natural setting, surveys using quantitative closed questions for data processing and textual analysis.

In *subjective research* the researcher leaves the objective sphere and becomes a part of the work process. Three of several relevant research designs in this field are grounded theory, mixed methods research and action research:

In grounded theory the main aim is to develop theories based on a systematic analysis of research data[23]. The process may be iterative and stops where no more data will add information to the theory building. Since the theory is build up from a specific set of data, the generalization of the theory to other research field should be made with care and may require validation by use of more constructive research methods. The research methods used in grounded theory research design may vary, but typically contains qualitative methods like interviews and observations.

Mixed methods research design typically combines the use of qualitative and quantitative data. The most common approach to mix the data is a triangulation design where complementary but different data are collected on the same topic. The strength of the quantitative method with possibilities for trend and statistical analysis is combined with the strength of interaction and adaptation that is possible with qualitative methods[18]. The obvious strength of mixed method design is of course that by using both quantitative and qualitative data a broader picture of the result will be achieved encompassing the multi disciplinary view of a problem.

The final research design discussed among the subjective research methods is action research. In this design, research is performed in collaboration with the participants. The objective is to address real life problems holistically and action research looks at diversity as enrichment for the process. The workability of solving problems through the research action is important[24]. Of all the subjective research approaches action research is probably the one requiring most collaboration and interaction since the researcher will initiate the measures leading to changes in the process itself [23]. Action research may utilize several research methods, but case studies[22] are central.

2.3 Research Design for the PhD

As indicated in Figure 3, the research described in the thesis encompasses a variety of research methods in the different parts of the study.

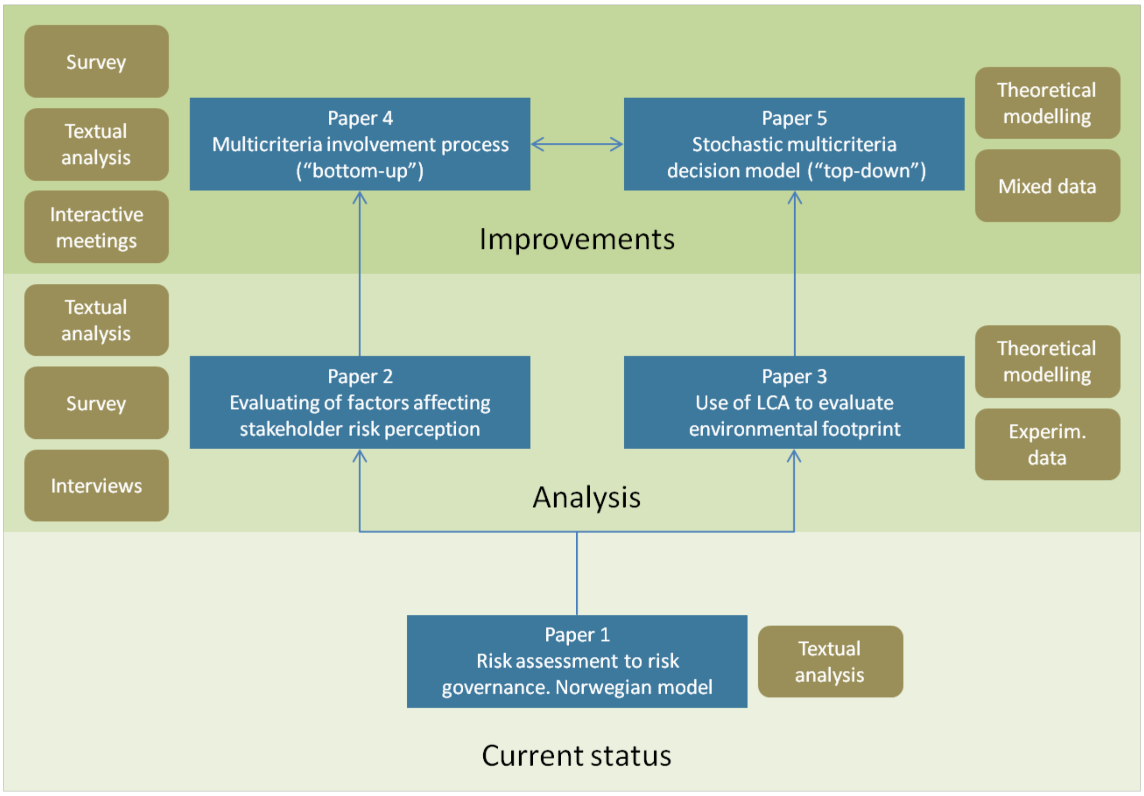

Figure 3 Research methods used in the work

By looking at the work in the methodological dimension given in the left and right side of the figure, the choice of methods qualifies this project for a mixed methods research design using methods typical for objective, constructivistic and subjective epistemology. However, by looking on how the different papers are related in work phases, the picture gets slightly different. Here the contour of an overall case becomes stronger looking at the problem in a reflective and analytical angle, finally proposing improved decision models for future use. The research context promoting the management shift from single to multiple criteria decisions, focusing on workability and methods in the process is clearly subjective; perhaps more in the line of the action research definition. The idea of using mixed methods in action research is not a contradiction. Greenwood and Levin [24] states that action research can have a mixed method research design as long as the mixing of methods is contextually determined. However, there are no feedback loops in the thesis to evaluate the effect of introduced methodological changes to the management process which is also an important aspect of action research[25]. A relevant conclusion is therefore to say that the research design is a subjective and case-based type of research supported by the use of mixed methods.

3. Evaluation of Current Status

3.1 Risk Assessments and Risk Governance

Assessing impacts from contaminated sediments has traditionally relied on comparing the chemical status of sediment samples with predefined sediment quality guideline values (SQG) through a screening process, where exceedance above background values require further evaluations and/or remedial measures, Figure 4. When SQG's are established the regulatory framework for screening of contaminated sites is simple and transparent avoiding ambiguity for decision makers whether further management is required or not.

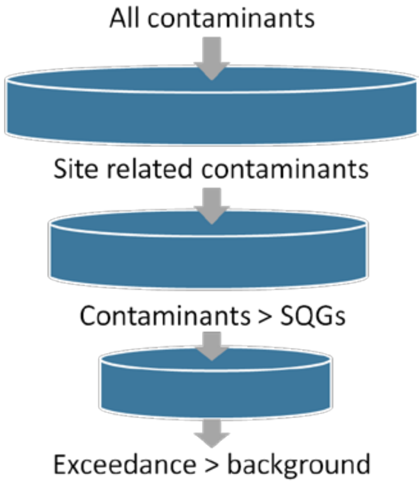

Figure 4 Use of SQG contaminant screening process. Adapted from [26]

To understand the inherent precautionary principle behind SQG it is necessary to briefly discuss how these values are determined, Figure 5.

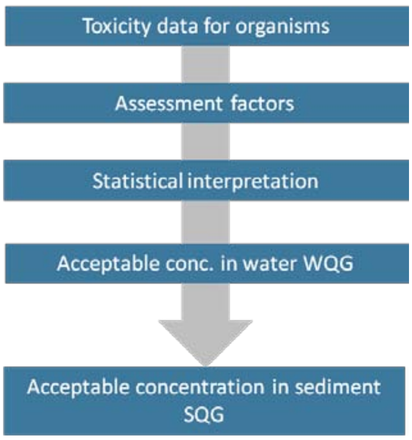

Figure 5 Assessment of Sediment Quality Guideline values (SQG)

SQG for sediments are normally estimated from toxicity data derived from laboratory or/and field studies of acute and chronic toxicity of various chemicals for aquatic organisms. The objectives with the toxicity data are to estimate predicted no-effect concentrations (PNEC) for the chemicals in the water. If toxicity data exist for terrestrial organisms they are used directly to estimate PNEC values for sediments and soil. Toxicity tests estimate the concentrations of a chemical which affects half of the exposed organisms. This effective concentration (EC50) for all the investigated species forms the basis for estimating acceptable concentrations in water. In cases where toxicity data is present for organisms in different trophic levels, a species sensitivity distribution curve (SSD) is established. PNEC values are then calculated probabilistic as the level where 95% percent of the species are protected in an ecosystem. In case of lacking data, PNEC values are calculated based on available data applying assessment factors to compensate for lack of data material. Finally the SQG for water are converted into SQG by multiplying with the sediment/water distribution coefficient (K_d).

In Norway SQG have been developed classifying sediment in five classes from background concentrations (class I) to severely affected (class V)[10]. In this system the upper limits for class II and class III are the PNEC for chronic respectively acute (intermittent) exposure from the compound and have been used for establishing acceptance criteria for screening coastal areas. The SQG are used within the risk assessment framework as a screening level (tier 1) to see whether remedial measures are necessary. If the SQG are exceeded, a full HERA is recommended (tier 2). In tier 2 risks are estimated for both exposure to the ecosystem and human health. In addition the framework requires site specific calculation of contaminant spreading both as an input to the risk assessment and in case there are site specific objectives.

The Norwegian risk assessment model also opens up for a third tier. This may be collection of more site specific data, use of additional non-chemical data or use of more extensive models to predict the effect of contamination. This third tier has similarities to the sediment triad used in US and Canada management systems for contaminated sediments[27,28]. This system describes a decision weighing system where several lines of evidence (LOE) as chemical analysis, toxicity studies on relevant species and alteration studies assessing state of the site and potential contamination-related impacts are combined.

Both the Norwegian and the triad decision models incorporate an embedded subjective weighing of evidence, either qualitatively or quantitatively[29]. This duality of risk with an objective and subjective part is naturally also present for other actors subjected to the results of the risk assessments[30].

One way to expand the process of contaminated sediment management is to incorporate it into a broader risk governance perspective. Risk governance includes the totality of actors, rules, conventions, processes and mechanisms concerned with how relevant risk information is collected, analyzed and communicated and how a management decision is taken. One of the main aspects in risk governance is the acceptance and understanding of the duality of risk in an objective and subjective part[30]. A transparent decision model will therefore have to balance the socio-economic and political considerations with scientific evaluations into a governance framework.

Perhaps the most comprehensive conceptual risk governance framework is described by the International Council of Risk Governance[9,13], Figure 6.

Figure 6 *Conceptual model of the risk governance framework. Adapted from IRGC[13]*

This conceptual framework consists of four stages promoting a holistic view on understanding and managing risks;

The *pre-assessment* serves as the baseline for the risk assessment and management, giving guidance on both the dimension of the risk, the relevance and interests of the stakeholders and public as well as the existing foundations such as laws, regulations and other relevant guidelines. The framework defines stakeholders as socially organized groups who are or will be either affected by the risk or have strong interests in the issue. Public is defined as individuals, non-organized groups or media that are experiencing the outcome of the event or is a part of the opinion on the issue[9]. The pre-assessment is important since it allows the duality of risk to be reflected early in the policy making phase.

The second step is *risk appraisal*. This step identifies and assesses important information about the risk to be used in the subsequent characterization and evaluation steps. The risk appraisal contains both the conventional scientific *risk assessment* based on the identification of hazard, exposure, vulnerability and probability of occurrence. The framework also contains a *concern assessment* which encompasses the associations and the perceived consequence that the stakeholder may associate with the hazard. This assessment will identify the potential bias between the scientific (objective view) and the perceptive (constructivist view) allowing both sides to be reflected in the evaluation and management of risk.

The third step in the framework consists *of risk characterization and evaluation*. This step encompasses both the process of characterizing the risk according to the findings in the appraisal phase, as well as an evaluation of the tolerability and acceptability. For the scientific evaluation this normally means evaluation towards predefined acceptance criteria, whereas this model also includes an evaluation towards social values based on the result of the concern assessment.

The final stage of the framework is the *risk management phase*. This phase comprises the identification of risk reducing measures and the decision making process as well as the design, implementation and monitoring the effect of these measures. *Communication* is central in the process, indicating that communication with stakeholders is required to build trust in all phases of the framework.

3.2 Evaluation of the Norwegian Framework

A comparison between the Norwegian HERA based management framework and the idealized IRGC risk governance frame work is conducted by reviewing the management plans for contaminated fjord areas[12]. Plans have been prepared for 29 areas, targeting 17 of them for further remedial actions. The work has systematically reviewed to what degree each of the four stages in the IRGC governance framework are incorporated in the 17 produced management plans.

The results, as presented in Figure 7, show a strong focus on ecological risk assessments especially the use of sediment quality guidelines (SQG), whereas concern assessment incorporating risk perceptive values and public acceptance is scarcely addressed. In 70% of the plans the management strategy is also formed by using an ad-hoc process, meaning that recommendations for remedial actions are not based on a systematic evaluation, weighing and prioritization of the obtained data.

The majority of the plans are therefore more of an instrument to evaluate the present situation and evaluate the results from the risk analysis than a fully developed management plan including the totality of all actors present in the management process. Even if the different data to support structured management decisions based on multiple criteria are in place, most of the recommendations for implementation of sediment management for the different locations are based on ad-hoc processes.

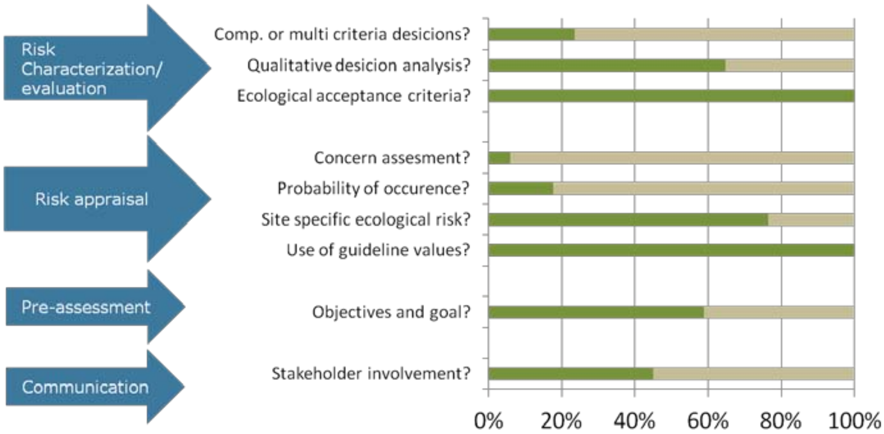

Figure 7 *Incorporation of risk governance elements in the management plans of 17 Norwegian fjords* [12]

3.3 What are the Gaps to a Risk Governance System?

The review of the Norwegian management model in relation to a conceptual risk governance framework confirms the Norwegian model as strongly driven by health and ecological risk assessments and use of SQG. This is visible in the pre-assessment phase where HERA methodology has served as the basis for the risk appraisal phase and the characterization evaluation performed, Figure 8.

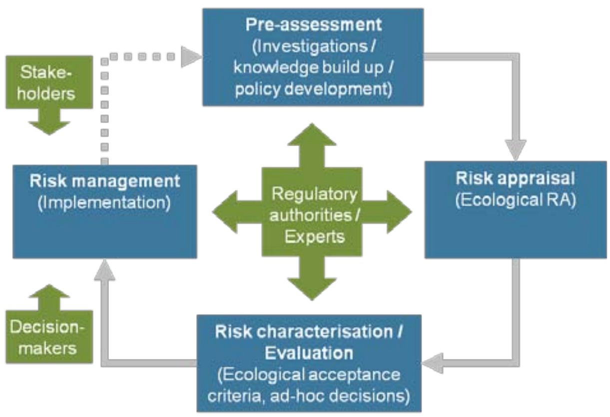

Figure 8 *Work process for the existing management system of contaminated sediments in Norway. Adapted from IRGC* [13]

From a process oriented view, the management system is dominated by the view of regulatory authorities and experts on ecological risk assessments, developing the management system and performing the assessments.

This focus on HERA expert competence has framed the system and contributed to encapsulate it from developing into a governance system incorporating the broader picture of risk including risk perceptive values. The involvement of stakeholders and decision makers in the analyzed management plans, and the communication during the process may be categorized as advisory, mainly influencing the management phase.

This lack of involvement, concern assessment and judgment of the acceptability and tolerability of risk together with the strong focus on expert driven HERA are therefore the main gaps as compared to a risk governance system.

To develop a sustainable management system, encompassing environmental, economical and social interests, a stronger focus on concern assessment, risk evaluation and multicriteria evaluations is required, allowing stakeholders and decision makers to participate more up front in the management process.

4. Analysis of Social Aspects in Management

4.1 Social and Risk Perceptive Factors

Whereas ecological risk assessments evaluate hazards from contaminated sediments to be related to toxic effects for humans and the ecosystem, certain members of society may use a more intuitive assessment of the risk involved. The distinction between this statistically estimated risk and public acceptability was early identified and addressed as risk perception[31]. Previous research has documented that risk perception may differ significantly from statistical estimations and is affected by social acceptability[32]. We are, for example, more willing to accept risk we believe we can control than risks that are forced upon us of an unknown nature. Later research has nuanced this view, suggesting that risk perception depends both on rational and intuitive arguments[33]. As with other environmental issues, the involvement of the public in sediment management has become more evident and should be addressed. As a result of such involvement it is necessary to consider risk also in contaminated sediment management in a much broader context than earlier[12].

Assessing risk of contaminated sediments to humans and the ecosystem is a complex process involving use of both experimental data and expert knowledge. Due to the long term nature of the risk, potential health effects of human contamination exposure are minor and difficult to separate from other health risk related problems in an urban environment. This makes sediment contamination more of a societal than a personal problem for people. Extensive knowledge in the field of chemistry, toxicology and biology is required to be able to understand the fundamental assumptions behind HERA. On the other hand SQG are presented very visual to public, for example by use of illustrated maps, Figure 9.

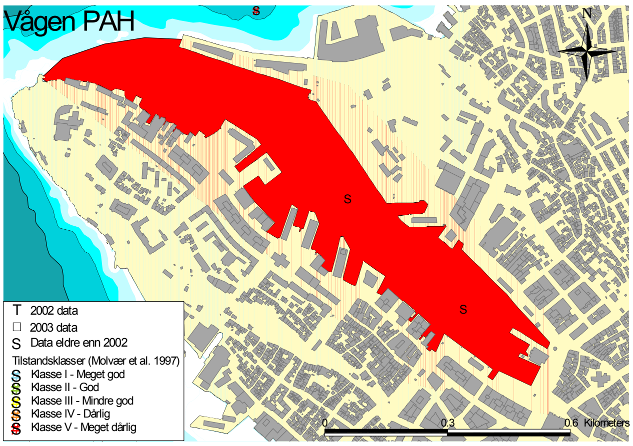

Figure 9 Presentation of PAH contaminant status for Vågen Bergen

The visual message presented in such a figure, may very well overshadow the inherent assumptions in the classification system underlying the visual presentation. Even though sediment experts can frame the risk level in a wider setting, use of SQG values certainly leaves room for risk perceptive values. Since SQG is used in Norway not only for screening purposes, but also as remedial guidelines and within remedial permits, it is plausible to assume that a HERA framework actually could have management related impacts with respect to social and risk perceptive factors.

4.2 Risk Perceptive Values in Oslo Harbor

The evaluation of the Norwegian management model[12], points to weaknesses in the Norwegian management model with respect to stakeholder involvement and a lack of focus on concern assessment and understanding of risk perceptive values.

When a conflict emerged in the Oslo harbor sediment remediation project it is a logical choice to see whether risk perceptive values have been an important aspect within this conflict.

Figure 10 Example of civil protest actions against the Oslo harbor project (source: www.stopp-giftdumpingen.org)

The major conflict area in the project is related to the disposal of 300.000 m^3 contaminated sediments after dredging. Two principally different solutions have been evaluated during the planning phase. One solution involved the transportation of the dredged material on barges to a land disposal site, situated approximately 80 km from the harbor. This site, NOAH Langøya, is a national disposal facility for hazardous waste. The second option was to construct a confined aquatic disposal site (CAD) at Malmøykalven 3 km from the dredging area. This site, a 70 meter deep sea-basin 3 km from the dredging area, has previously been used for uncontrolled disposal of dredged material. The choice to dispose dredged contaminated sediments in a confined aquatic disposal (CAD) site rather than at a land disposal site received a lot of societal attention, attracted large media coverage and caused civil protest actions against the project, Figure 10.

As described in paper 2 a mixed method approach has been used to investigate how risk perceptive affective factors (PAF), socio-demographic aspects and participatory aspects influences the various stakeholders' preferences for the two different disposal options. Data has been collected to reflect the views of the stakeholders involved in the project rather than the general public opinion. Stakeholders are defined here as people, organizations or groups who are affected by the issue and who have the power to make, support or oppose the decision or who have the opportunity to provide relevant knowledge to the decision making process[34]. Interviews and analysis of documents have been used as support for the survey. Evaluation of results has been performed using the validating quantitative data model[18]. Statistical analyses have been conducted to assess whether it is possible, based on the survey data material, to identify and relate any of the PAFs to the perceived risk of the CAD. The study has used exploratory factor analysis based on the principal component method (PCA) to identify underlying factors based on the survey model questions. PCA as well as subsequent analyses of variance (ANOVA) and reliability testing has been performed to evaluate the data. Structural equation modeling (SEM), normally used in psychological research, has been used to identify structural relationship between the identified factors. More details are given in the method section of the paper[14].

The findings as summarized in Figure 11 support the view that perceived risk and underlying PAFs are indeed vital for the choice of preferred remedial solution and therefore may be an important factor to address when selecting disposal solutions in contaminated sediment management. Risk perceptive factors such as *transparency* in the decision making process and *controllability* of the disposal options are identified as important for risk perception. The stakeholders' preferences for disposal solutions are with the exception of *education* and *risk aversion* not impacted by socio-demographical and participatory aspects.

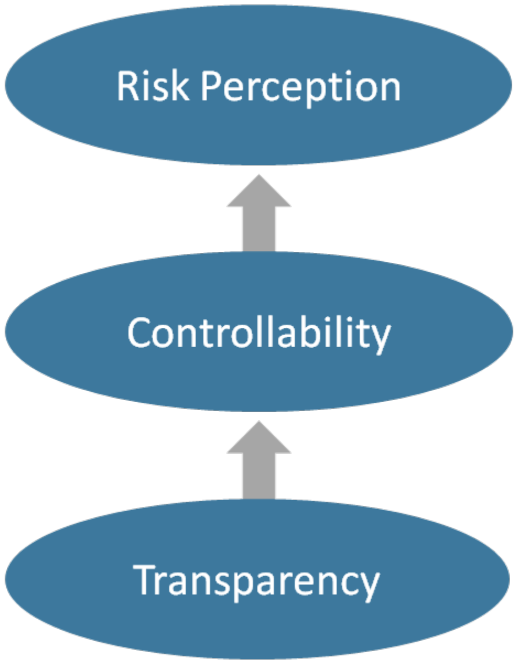

Figure 11 Structural relationship between the PAF's transparency and controllability, with risk perception

4.3 How are Risk Perceptive Values Affecting Management?

The results of the study, as presented in the second paper, identify a structural relationship between the selection of management alternative and risk perceptive affective factors among stakeholders. This answers the second research question, emphasizing the fundamental need to address transparency, openness and information in sensitive environmental decisions in order to progress to more technical questions. This confirms the need to assess and evaluate the perceptive sides of risk as suggested by the IRGC risk governance model[13]. As discussed in paper 1, the present Norwegian management system presently lacks appropriate instruments and requirements to perform such evaluations.

This case study also supports the view that there is no sharp distinction in risk perception between experts and other parties involved. Non-expert stakeholders may be very well informed, adopting their alternative expert opinion based on the various information sources available.

Further research on methods that allow for more open and transparent stakeholder involvement processes are therefore highlighted in the study as a focus area for development of decision models.

5. Analysis of Environmental Impact

5.1 Life Cycle Assessment and Adaptations

As earlier discussed, selection of sediment management alternatives for contaminated sediments is often based on human and ecological risk assessment (HERA) frameworks[35]. Whereas HERA is suitable for assessing whether the contaminated sediments constitute an unacceptable human and environmental risk, it does not address environmental consequences aggregated over the whole life cycle of the remediation project and from intended future site use. Even though high-end remediation alternatives may reduce the risk associated with sediment contamination to acceptable values, the material production and technology necessary for implementing these alternatives, as well as the energy and equipment use they necessitate, may result in environmental hazards presently not quantified by traditional HERAs. One common way to determine the relative environmental impact between product systems occurring over the whole life cycle is by use of life cycle assessments (LCA). In this method the inputs, outputs and the potential environmental impacts of a product system are compiled and evaluated throughout the product's life span[36,37].

An LCA consists of four steps[38]; goal and scope definition, life cycle inventory (LCI), life cycle impact assessment (LCIA) and interpretation. Goal and scope definition is important since it determines the content and methodological choices for the subsequent steps. The inventory aggregates the various inputs and outputs into cumulative numbers, whereas the impact model converts these numerical data into potential effects as environmental and human harm and resource depletion. Finally, the impact results are interpreted and uncertainty and sensitivity in the results are addressed.

In LCIA of contaminated sites, impacts have normally been referred to as primary, secondary and tertiary effects[39], Figure 12. For contamination in urban coastal areas, as is the topic for this thesis, primary effects may originate from the contamination source through uptake in sea food and human exposure and local ecotoxicological effects on the benthic faunas as well as other local impacts of the remedial operation. Secondary impacts are the effects related to the use of resources and energy during the remediation. Tertiary aspects of the remediation may include increased recreational use of the area or increased commercial fishing.

The marine setting and introduction of sediments as a source of contamination has implications on the choice of methodology for converting chemical and physical data into information about environmental effects in the LCIA. Marine aquatic toxicity, which is important for this thesis, is scarcely addressed in available impact models for toxicity[40].

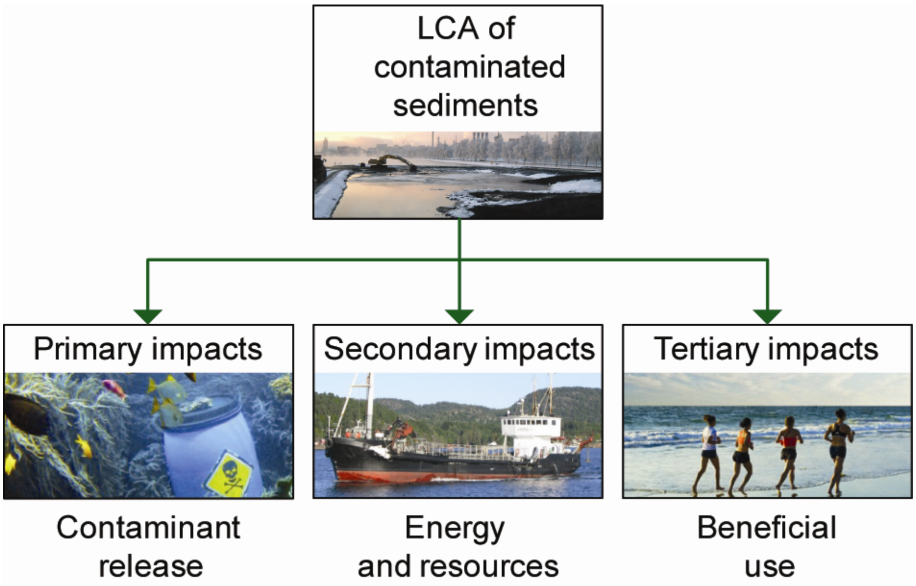

Figure 12 Dividing LCA in primary, secondary and tertiary impacts. Adapted from Lesage[39]

Sediments, if included in the models, are normally seen as a sink and not as a source for marine contamination. The ReCipe impact model[41] which utilizes USES-LCA[42] is at present the only readily available impact assessment method that includes a marine release compartment and is therefore selected for the work on LCIA within this thesis, Figure 13. The categories to the right in the figure show the damage categories that are adapted or added to the generic model.

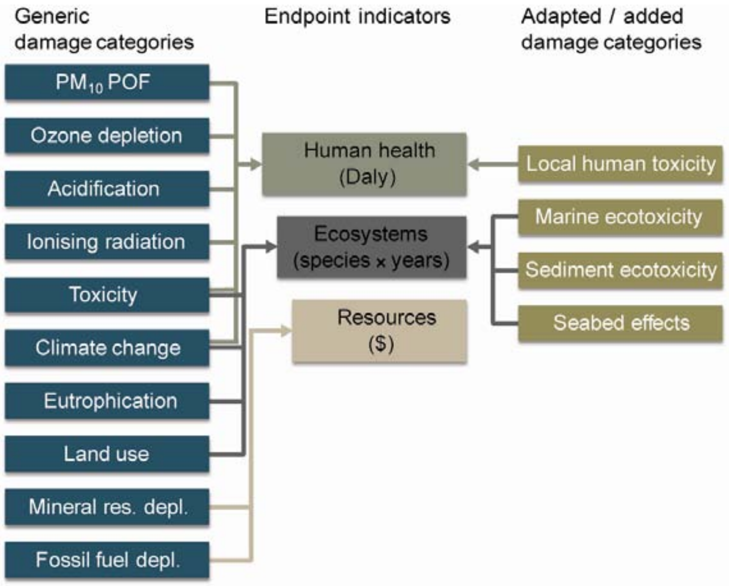

Figure 13 Combination of the generic and adapted or added damage categories into endpoint indicators for the ReCipe impact model used in the study[15]

In this thesis, an endpoint method is used in the impact assessment in order to achieve maximal agreement with the comparative and management-oriented objectives of the thesis. More information regarding the selected model and the performed adaptations is given in the paper[15].

5.2 LCA for the Grenland Fjord

The adapted LCIA model has been used to assess life cycle impacts for a potential sediment remediation project in the Grenland fjord, Norway which is contaminated by polychlorinated dibenzo-*p*-dioxins and -furans (PCDD/Fs), Figure 14.

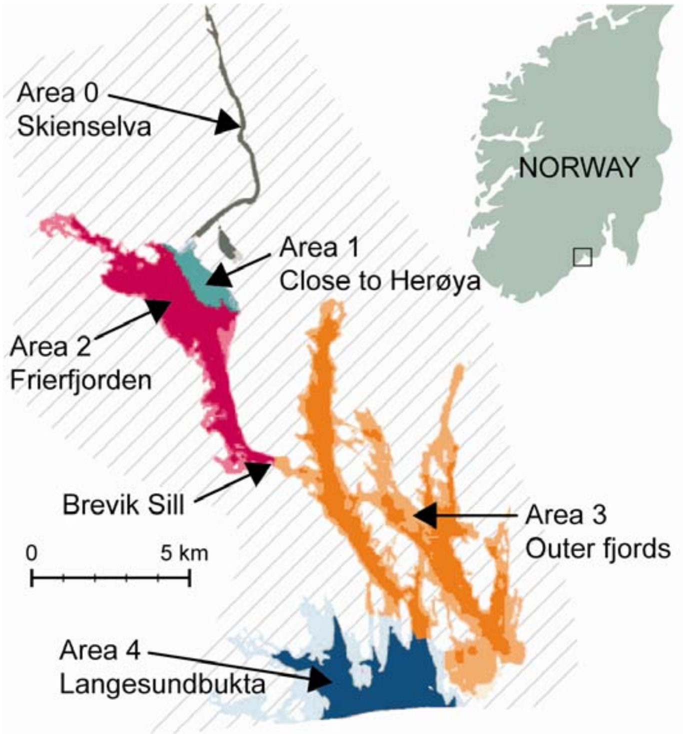

Figure 14 Bathymetric map of the horizontal compartment division in the model application to the Grenland fjords. Adapted from[43]

Capping of the contaminated sediments has previously been proposed to mitigate risk above the HERA-derived threshold values in fish and shellfish[43]. In this case study the effect of capping the sediments in the most contaminated inner area of the fjord (areas 1 and 2) has been evaluated. LCA methodology has been used to investigate the environmental footprint of different active and passive thin-layer capping alternatives as compared to natural recovery. The investigated materials consist of locally dredged clay, limestone from a nearby area and activated carbon (AC). The AC comes from two different sources; anthracite based AC from mines and bio mass based AC from coconut shells.

The results of the study as presented in detail in the third paper[15] show that capping is preferable to natural recovery when analysis is limited to effects related to the site contamination (primary effects). Incorporation of impacts related to the use of resources and energy during the implementation of a thin layer cap (secondary effects) increases the environmental footprint by over one order of magnitude, making capping inferior to the natural recovery alternative, Figure 15. Use of biomass-derived activated carbon, especially when crediting carbon dioxide sequestration during the production process, reduces the overall environmental impact to that of natural recovery.

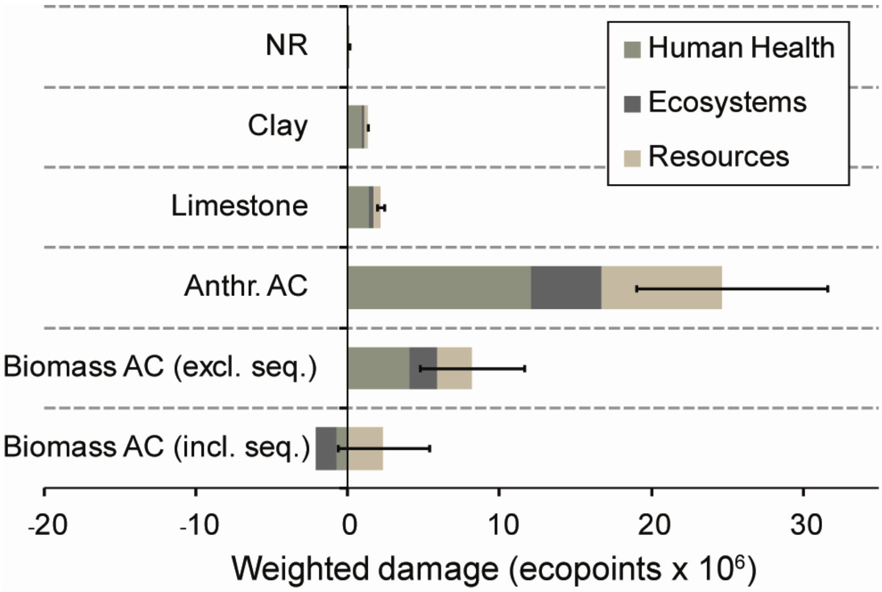

Figure 15 Normalized and weighted environmental footprint for alternative capping materials used to cap the contaminated inner fjord [15]

5.3 Can LCA be of Value in Sediment management?

The results from this study indicates that LCA is a valuable tool for assessing the environmental footprint of sediment remediation projects and can be used for problem formulation and prioritization and optimization of remedial alternatives from a life cycle perspective. The use of LCA in contaminated sediment management gives priority to remedial solutions with limited raw material and energy use. LCA may be especially relevant for addressing beneficial sediment and alternative energy uses, such as the use of biomass-derived AC instead of coal based AC as discussed in the paper. LCA can therefore be the answer to the third research question, which is asking for holistic evaluation of environmental impacts from sediment management. LCA can also be an important input for risk management option evaluation as described in the IRGC Risk Governance framework[13].

There are however many issues that need to be carefully considered in implementing LCA for sediment management. The results presented in Paper 2 shows how the environmental risk factors associated with sediment contamination have been extended to incorporate effects associated with the implementation of sediment management alternatives. The difference between traditional HERA results and results from the LCA are however substantial[44], and the LCA can therefore only be attempted for comparative assessment of remedial alternatives found to be acceptable through HERA. The question of relevant scale and focus is important for both LCA and HERA. In general, HERA considers the local scale and focuses on risk of specific stressors, while LCA operates on a global or regional scale, normalizing and weighting impacts for relative comparison. As for other specific LCA applications[45], the results from this study emphasize the necessity of including a local compartment to the impact assessment model for future LCA applications in coastal areas to reach an acceptable resolution in the impact assessment. Even so, based on the standardized normalization and weighting procedures applied in the study, the damage from primary aspects are assessed as relatively minor compared to the secondary aspects. From a life cycle perspective, contaminant levels have to be substantially higher to justify commonly accepted remediation practices, which may contradict public values. Therefore, instead of basing the weighting on standardized damage categories more focus may be given to the perspective of the decision maker, thus giving higher focus to local (primary) effects than global (secondary) effects in the LCA.

In addition, both LCA and HERA do not explicitly consider many social related factors important in the selection of sediment management alternatives. One way to address this is to assess the tertiary effects related to the remediation[46]. Examples of such effects would be increased recreational use of the area or increased commercial fishing after lifting the dietary advisory. This approach would, however, require a more developed system for monetization of social and economical impacts[47]. Establishing a more complex cause and effect related weighting systems may, on the other hand, reduce the transparency of the study and increase the use of controversial criteria which is undesirable[48].

The results presented in the paper therefore indicate a need for further research towards developing integrated framework for sustainable sediment management based on plural methodology, combining LCA and other methodology to avoid controversial weighing procedures and extensive use of monetization.

6. Improvement of Management Models

6.1 Available Multicriteria Models

The results from paper 2 suggest that emerging environmental challenges as coastal area management call for more effective stakeholder involvement processes for environmental management. A structured stakeholder involvement process could help in overcoming disagreements and result in better management alternatives[49]. Examples of qualitative involvement processes include focus groups with facilitated communication between parties to reach consensus[50]. Co-operative discourse methods are described by Renn[51] involving establishment of development criteria and alternatives using value trees elicited by stakeholders and experts in round table meetings. Group Delphi is another systematic, interactive forecasting method which relies on a panel reaching consensus through sequential use of questionnaires and intermittent discussions.

Multicriteria analysis (MCA) has been proposed as a method to enhance stakeholder involvement in sediment management and to facilitate decision making of complex problems[52-54]. The purpose of MCA in these studies has been to support evaluation and selection among management alternatives in an interactive process involving decision makers, stakeholders and scientists. Methodologically, MCA requires developing hierarchy of criteria and metrics to compare management alternatives and subsequent elicitation of weights to quantify relative importance of criteria, as well as scoring of alternative performance based on these criteria. The MCA approach overcomes the limitations of unstructured individual and group decision-making by providing decision transparency and focusing discussion on assessing the weights and scores. Thus MCA may be valuable in quantitative decision making; however, focus on participatory aspects in the involvement processes for sediment management is also warranted[14].

Several additional methods can complement HERA in data acquisition to promote a more high level holistic perspective (Table 2). Use of LCA can supplement HERA to create an enhanced systems approach to sediment management. By means of LCA the environmental impact is assessed. The information can then be used to evaluate the total environmental impact aggregated in time and space to find the total environmental impact. However, perceived benefits of remediation not readily reflected in market prices, are scarcely addressed in current approaches to LCA[55].

Survey-based contingent valuation (CV) of such non-market benefits has been used in environmental economics for many years[56], but only recently applied to contaminated sediments[57]. However, quantification of non-market values are difficult. Contingent valuation of non-market environmental benefit is subjected to discussion for providing hypothetical rather than actual revealed willingness to pay for remediation[58].

While the above methods are valuable for data aggregation in contaminated sediment management, prioritization of targets for improvements is often conducted based on comparing a monetary valuation of benefits against remediation cost. One traditional way of performing this evaluation is through cost-benefit analysis (CBA), where alternatives are ranked according to the net additive value of their benefits and costs. In cases where costs may need to be compared against a single not-easily monetized impact, cost-effectiveness analysis (CEA) can be used as an alternative to CBA[59]. Traditionally, CEA has also been used in Norwegian contaminated sediment management by evaluating remediation costs against the reduced health risk from exposure to sediment contamination[57].

Table 2 Comparison of different data-aggregation and evaluation methods relevant for complex environmental decisions

Decision phase	Method	Strength	Weaknesses
Aggregation	Human and ecological risk assessments (HERA)	Quantitative estimates of absolute risk	Resources used to reduce risk are not addressed
	Life cycle assessments (LCA)	Holistic perspective for impact assessment	Global focus with insufficient resolution for site assessment
	Contingent valuation (CV)	Monetization of non-market benefits of environmental quality	Stated hypothetical responses regarding stakeholder preferences
Evaluation	Cost-effectiveness analysis (CEA)	Avoids monetization of controversial topics	Only single criteria may be evaluated
	Cost-benefit analysis (CBA)	Quantitative well known decision method	Requires monetization
	Multicriteria analysis (MCA)	Aggregation of multiple data without monetization	Subjective weighing of criteria is necessary

Regardless of the metrics selected, both CEA and CBA have been criticized for their use in environmental projects because their monistic approach simplifies utility to a single value dimension[58,60]. Multicriteria analysis (MCA), in which both quantitative and qualitative criteria may be combined without the need to reduce parameters to a single unit, has been proposed as a method to overcome these traditional problems of CBA and CEA in public decision making[61]. MCA, however, has its own criticisms.

MCA recognizes that choices and decisions involve multiple subjective values and therefore require extensive use of deliberative methods and expertise in cognitive science to determine the values from decision-makers[62]. CBA and CEA often rely on a combination of large sample sizes and simulation-based probabilistic scientific models which it is claimed to provide more representative, replicable and less subjective values[63].

As a result of this research, the value of using MCA within a multicriteria decision framework is recognized. Different models are however required depending on the objectives of use. Two management models are proposed in this thesis. One for a “bottom-up” advisory settings and one for “top-down” integrative decisions.

6.2 The Multicriteria Involvement Process

In order to enhance the value of participatory stakeholder involvement in environmental management, a multicriteria involvement process (MIP) which builds on the quantitative principles of MCA, and incorporates group interaction and learning through qualitative participatory methods is proposed in the fourth paper[16], Figure 16.

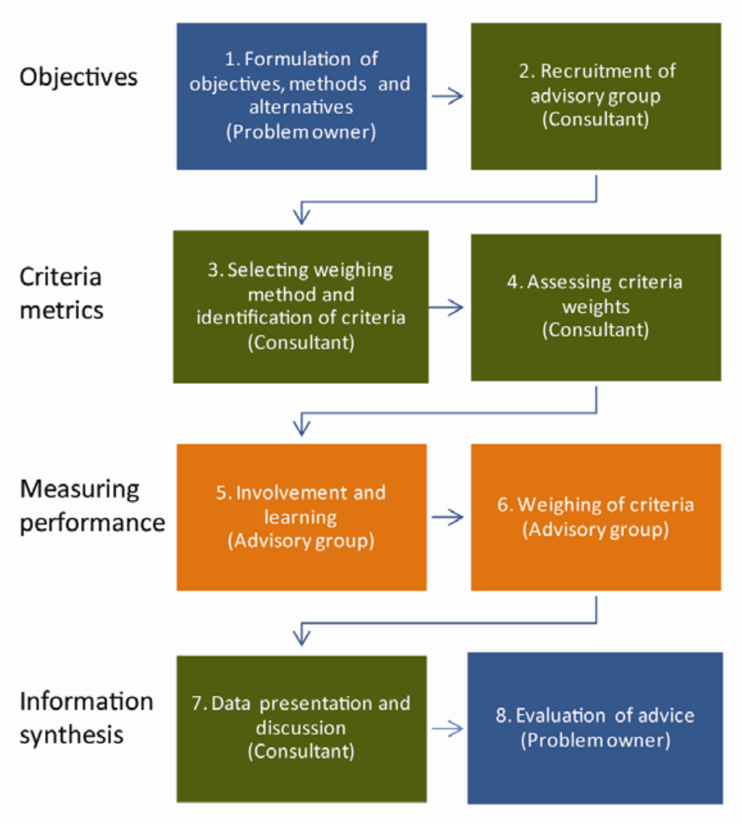

Figure 16 Schematic overview of the proposed multicriteria involvement process

The process bears resemblance to earlier proposed MCA processes for sediment management[64-67]. However, the process proposed here also addresses recruitment and includes an involvement and learning step inspired by deliberative decision making using citizens' juries[68,69]. In many ways use of the MIP would help to reduce the gaps on the involvement side identified in the first paper[12]. Use of MIP on appropriate advisory groups would give a possibility of conducting concern assessment, risk evaluation and option evaluation as required within the IRGC risk governance framework[13] in a practical manageable way.

The application of the MIP is illustrated for a sediment remediation case in Bergen harbor, Norway, by conducting the process for three different advisory groups including local residents, local stakeholder and non-resident sediment experts, Figure 17. A comparison of individual versus group consensus-based ranking of alternatives is presented in the paper.

Figure 17 Interactive meetings with Bergen residents as an integrated part of the involvement process (source: http://sedimentandsociety.ngi.no)

The harbor area of Bergen is contaminated due to previous industrial activities such as naval shipyards and manufacturing industries, earlier releases of municipal sewage and urban run-off from diffuse sources. The objectives with the MIP in this case is to provide valuable advice to the problem owner on how advisory groups perceive hypothetical remediation alternatives distilled from the recommendations laid out in the management plans.

Five alternatives for remediation of the contaminated sediments in Bergen harbor have been suggested based on the problem formulation performed with the problem owner, Figure 18 (MIP step 1):

- *Alternative 1* constitutes of natural recovery (NR) due to deposition of clean sediments on top of the contaminated sediments.
- *Alternative 2* consists of an active reduction of the contaminant flux by capping the inner fjord basin (Cap).
- *Alternative 3*, is similar to alternative 2 but in addition to capping, hot-spots are assumed to be dredged and material stored in near shore disposal facilities (Cap+NS)
- *Alternative 4 and 5* are similar to alternative 3 but dredged material is assumed to be disposed in local (LD) and national (ND) land-based waste disposal facilities respectively (Cap+LD and Cap+ND).

The analytical hierarchy process (AHP) has been used to pair-wise compare criteria. A hierarchical decision tree is used by organizing criteria in three levels, reflecting the different pillars of sustainable development; environmental, societal and economical aspects[70]. Under each criterion, sub-criteria are added. More information about methods and discussion about MCA methodology is given in the methodological section of the paper[16].

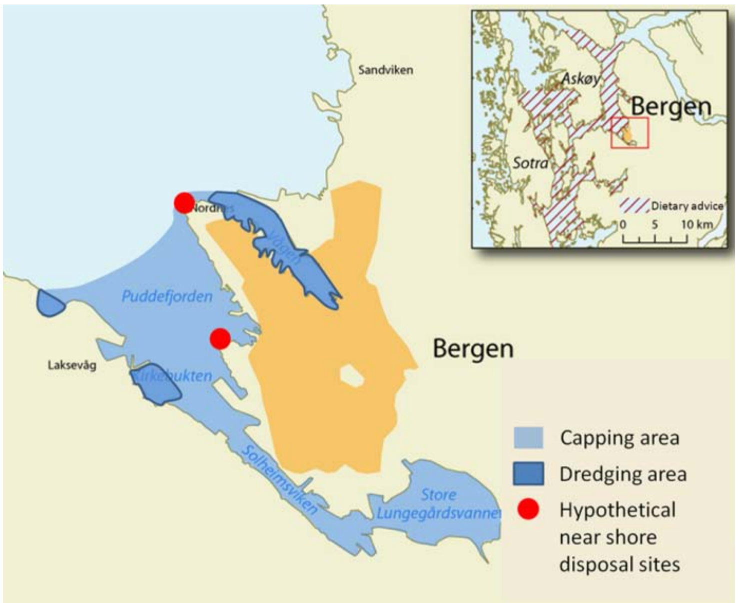

Figure 18   Proposed hypothetic remedial alternatives Bergen harbor

The preferences on remedial alternatives as a result of the MIP are presented in Figure 19. Her we can see a relative consensus among the different advisory groups around two specific alternatives, sediment capping or sediment capping with dredging of high contaminated areas with subsequent disposal of contaminated sediments in near shore disposal facilities.

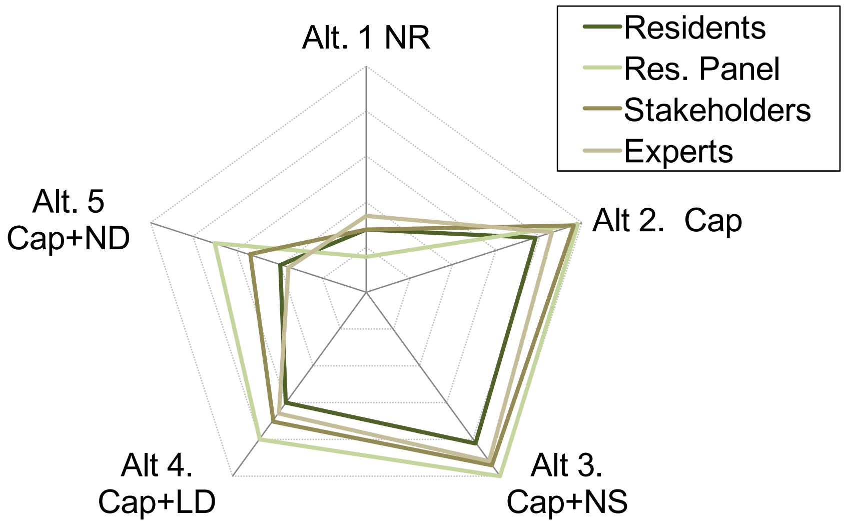

Figure 19 Results of preferential alternatives through the MIP

In addition important information is obtained by analyzing the weighing of criteria across the groups. The averaged criteria weights are higher than normal for the reduction in human and environmental risk. This observation can explain the low score on a natural recovery scenario, since this alternative is less effective in reducing the human and environmental risk on the short term than the other remediation alternatives.

The study also indicates that in order for stakeholders or residents to be able to embrace a complex decision such as selection of remediation alternatives, the involvement process with lateral learning, combined with MCA providing structure, robustness and transparent documentation is a preferable choice.

6.3 Integrative Model Based on Stochastic Multicriteria Analysis

In the fifth paper[17] the use of an MCA framework to prioritize the selection of contaminated sediment remedial strategies on an integrative level is advocated. The "top-down" method proposed differs from the MIP by emphasizing totality, flexibility and statistic reliability over user friendliness and information exchange, which is more relevant in an advisory setting. The objective with this model is to promote a value neutral decision methodology as an alternative to traditional CEA and CBA analysis. The results of the MIP can however be used as input for the proposed integrative framework.

Due to the complexity inherent in contaminated sediment management problems as discussed above, decision methods supporting the sustainable management of contaminated sediments will require use of multiple methodologies to be conclusive. This can be achieved by combining different analytical approaches like; risk analysis (RA) evaluating human and ecological risk; life cycle assessment (LCA) for evaluation of environmental impact holistically; and multicriteria analysis (MCA), including economic valuation methods for social and economical aspects, Figure 20.

Figure 20 Integrative high level decision model for contaminated sediment management

The necessity of using different methods is further confirmed by looking at how contaminated sediment risk is classified in the first paper[17]. The conceptual governance model[13] divides problems based on the characteristics of the risk, *simple, uncertain, complex and ambiguous*. The first study classifies contaminated sediments as a mainly uncertainty based risk problem, but uncertainty and ambiguity characteristics is also present. This diverse classification confirms the complexity and the need to use different methodology for conclusive decision making.

The thesis propose to use stochastic multicriteria analysis (SMCA) based on the PROMETHEE II outranking algoritm[71] to implement this integrative strategy. SMCA allows the probabilistic simulation of uncertainty across a large number of criteria[64] and is less sensitive to correlations than MCA using internal normalization

(MAUT/MAVT). More description of SMCA and use of outranking is given in the method section of the paper[17].

In the study described in the fifth paper SMCA has been used to select the preferable magnitude of clean-material sediment capping for the dibenzo-*p*-dioxins and -furans (PCDD/Fs) contaminated Grenland fjord in Norway. The positive utility of health risk reductions and socio-economic benefits from removing seafood health advisories has been evaluated against the negative utility of remedial costs and life cycle environmental impacts. Though MCA is intrinsically value plural, the method has also been used to mimic results from CEA and CBA by manipulation of weights.

Six distinct scenarios based on differing magnitudes of the capping operation are investigated, Figure 21. The first scenario is one of natural recovery (NR), in which natural re-sedimentation with clean material from the watershed will over time reduce the PCDD/F fluxes to background concentrations. The next two scenarios describe situations where only the highest contaminated areas in the inner fjord (HIFC) or outer fjord (HOFC) are capped with a layer of 5 cm of locally dredged clay. As a comparison, the effect of capping the entire contaminated area of the inner fjord (IFC), outer fjord (OFC), or whole fjord simultaneously (WFC) is analyzed.

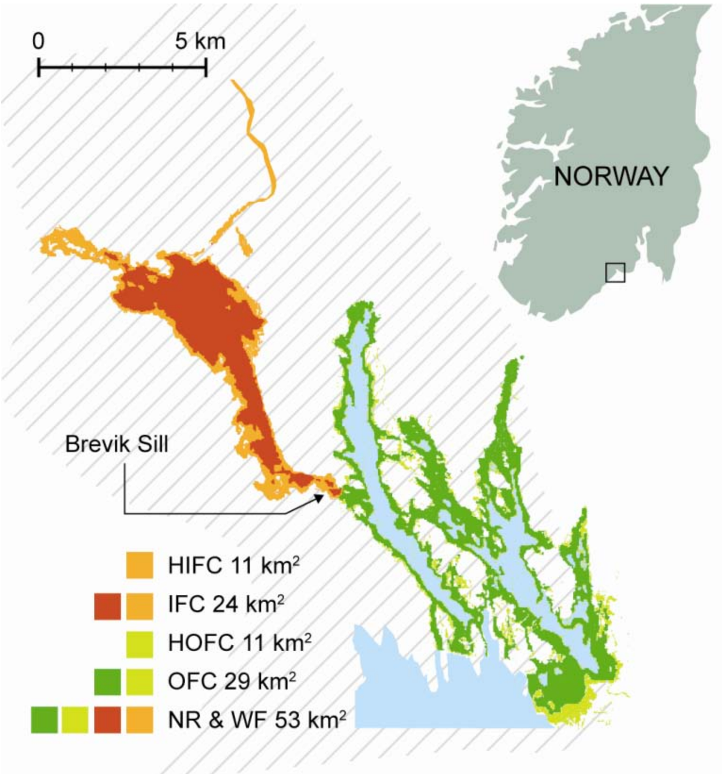

Figure 21 Overview of the six selected remedial alternatives for the Grenland fjord and subsequent estimated capping areas. Adapted from [43]

In the paper, utility curves are defined for four specific criteria relevant sustainable decisions incorporating social, environmental and economic objectives[7]. This includes positive utilities as health risk reduction and socio-economic benefits and negative utilities as monetary remediation cost and life cycle environmental impacts;

- *Health risk reduction* is defined as the expected number of years in relation to area before dietary restrictions can be removed.
- *The socio-economic benefit* of alternatives is estimated from the contingent valuation method estimating local people's willingness to invest in remedial measures to remove the dietary advisory.
- *Environmental impact* is assessed based on the environmental footprint of different capping alternatives
- *Cost* is estimated from previous studies and includes dredging and disposal of hot-spot areas combined with capping of the inner fjord and capping only of the outer fjord.

Criteria metrics for each alternative have then been estimated and Monte Carlo simulation using the probabilistic distributions of criteria has been conducted. More thorough discussion on alternatives, criteria metrics and simulation techniques is presented in the methodological section of the paper[17].

Table 3 gives mean values for the net flow of each capping alternative as a result of the simulations. The net flow expresses the dominance of one alternative to another within the range of ±1. Values close to one for an alternative indicate a strong dominance to other alternatives. In the case of net flows close to zero only weak dominance is observed, indicating only weak preference of one alternative over another alternative.

Table 3 Mean net flows from the SMCA analysis

Decision methodology	NR	HIFC	HOFC	IFC	OFC	WFC
Cost-effectiveness	0.03	**0.08**	-0.05	-0.07	-0.04	0.05
Cost-benefit	0	0.12	**0.33**	-0.16	0.06	-0.35
Value plural	0.02	**0.20**	0.12	-0.04	-0.12	-0.18

The results as given in Table 3 show a preference for capping of the highest contaminated areas in either the inner or outer fjord, depending on the decision methodology. Strongest preference is found for capping hot spots in the outer fjord (HOFC) based on cost-benefit principles. This means that all weighing preference is given to socio-economic benefit and monetary cost, neglecting other criteria. From a cost-effectiveness standpoint considering only health risk reduction and monetary cost, capping contaminated hot spots in the inner fjord (HIFC) proves most beneficial. A value-plural weighting balancing weights across all four criteria, shows a marginal preference between the two dominant alternatives (HOFC and HIFC) for capping highly contaminated areas in the fjord.

The study also analyses the robustness of the results looking at the stochastic dominance between cumulative distributions of net flow (CDF). This discussion is further expanded in the results section of the paper[17] and concludes with first order dominance for most alternatives. This means that the probability of ranking the alternative first is always highest and it should be preferred by all decision makers, whether risk adverse or risk seeking.

The robustness of the Grenland fjord results are further confirmed by looking at the results of simulations of the expected value of perfect information (EVPI) of each criterion (Partial EVPI)[72]. Evaluation of partial EVPI allows decision-makers and researchers to identify which criteria are significant for decision-making in terms of further data collection. Figure 22 shows Partial EVPI on criteria expressed by average net flow of highest scoring alternative(s).

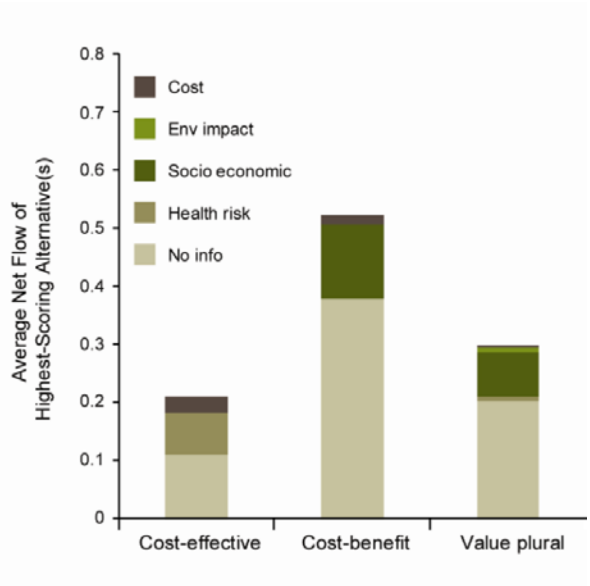

Figure 22 Analyzing the partial EVPI for the Grenland case

Perfect information about health risk will give the highest increase in expected value in the cost-effectiveness scenario. In this case, weak second order dominance is observed and generation of better information (reduced uncertainty) may yield more robust results and changes in the rank order (as illustrated by the larger proportion of average net flow ascribed to “health risk” in the cost-effectiveness approach). Collection of information about the socio-economic criterion is preferential in the other scenarios as well, but because there is little ambiguity in rankings in the benefit-cost and value-plural approaches, additional information on socio-economic benefits is not expected to impact the overall decision if these weighing sets are used (illustrated by the smaller proportion of average net flow ascribed to “socio-economic” benefits in these two approaches).

6.4 How to Structure a Sustainable Decision Model?

The results from the research presented in this thesis propose two models to incorporate sustainable values in contaminated sediment management. This will in plural address the fourth question about structuring decision models incorporating sustainable values. These will in addition be valuable methods to reduce the gaps to a more risk governance based approach of management.

The first model based on a participatory multicriteria involvement processes (MIP) is intended to enhance participatory involvement processes and to provide valuable advice for a problem owner in a contaminated sediment remediation process. The evaluation of the Bergen harbor case study indicates the feasibility of this method for these processes. The results show that using MCA as an integral step in the MIP adds structure and robustness to the involvement process and provides good documentation of criteria to be further addressed by the problem owner. However, there may also be challenges when using MCA in an advisory open end process. Firstly, there are considerations to make regarding the selection of MCA methodology. In this case, the analytical hierarchy process (AHP) is selected for its user friendliness. In other cases the use of different more mathematically robust methods may be required[73]. Secondly, this study shows that the quantitative scoring is perceived as problematic and has been questioned especially by stakeholders and experts. The interactions and the qualitative information gained from the different advisory group discussions, as suggested in the MIP, are therefore important in order to reduce misunderstandings and misinterpretations. Finally, it is important to remember that the emphasis on method and process should be balanced using both quantitative and qualitative methods as proposed in the MIP.

The second integrative model is intended for use in principle decision making, where focus on accuracy and uncertainties are important aspects. The results from the Grenland fjord case show that sustainable management decisions in contaminated sediment management requires use of multiple methods for data aggregation and evaluation[62] to be conclusive. Stochastic multicriteria analysis using outranking algorithms (SMCA), as a composite method, is proven to be a flexible and robust tool in this respect, giving valuable advice for high level decisions under uncertainty. The method can be used to mimic traditional cost-effectiveness (CE) and cost-benefit (CB) analysis decision methods or in weighting schemes that represent pluralistic evaluation. This gives the decision maker the opportunity to see how robust the results are using different decision analysis methods. The results of the study point to several obstacles to be addressed in future research. The study has assumed that a decision with respect to contaminated sediment management will have to incorporate multiple criteria reflecting sustainable values. This implies a holistic view of the decision maker requiring use of new and perhaps more controversial methodologies than normally practiced in a risk based management of contaminated sediments[10]. Another important aspect is the information lost when synthesizing data within the SMCA. As opposed to a CEA or CBA which is fully quantitative, this methodology gives only a relative rank order between alternatives. This calls, as demonstrated in this case, for a more qualitative interpretation of the results and a sufficient analysis of the robustness of the results compared to fully quantitative methods.

7. Discussion, Conclusions and Further Work

This thesis has looked at the Norwegian contaminated sediment case in a holistic matter and is proposing two decision models to facilitate sustainable management. Both models can be valuable contributions to promote a shift in management from single criteria risk assessments to a more plural based methodology. This chapter discusses the research contribution, methodological aspects and applicability of the performed work. In addition this chapter gives suggestions for future research.

7.1 Research Contribution

Several contributions to research can be synthesized from the thesis work, which in total respond to the research questions proposed in Chapter 1.2 of this thesis.

Research question 1

The Norwegian HERA management framework has in the thesis been related within a broader risk governance concept showing gaps in concern assessment, involvement and lack of structure in decision analysis as major issues. The gaps with respect of concern assessment and lack of structure in decision making is expected since qualitative subjective expert judgments embedded in the HERA have been reported before[29]. The thesis progresses further than earlier research by showing how an expert driven process affects the whole contaminated sediment management process in practice. This structured way of analyzing risk based management is novel and can be used in other cases to understand where and why conflicts may arise.

This also answers research question 1, which addresses the gaps between present management framework and the holistic risk governance model.

Research question 2

The importance of identifying, addressing and managing risk perceptive values through a structured involvement process has been demonstrated in the thesis through the Oslo harbor study. Whereas impacts of risk perceptive values on management have been demonstrated before in other areas[74-76], this study gives new knowledge on how risk perceptive factors are affecting an on-going sediment remediation case, which is valuable. By use of qualitative and quantitative methods it has been possible to isolate factors relating to the involvement process as highly important and it has been possible to demonstrate that structured involvement early in the management process is vital to reduce the gaps in risk perceptive values among stakeholders. The quantitative survey has been conducted with a limited population within an on-going conflict, which may bias the results. The mixing of methods has therefore been important to allow for validation of the quantitative results with results from interviews.

The study answers research question 2, by identifying what perceptive values are important for the management process and by giving advice on how to conduct the involvement process.

Research question 3

Also in the field of holistic environmental impact assessments this research has made contributions. Use of LCA for land based site remediation has been reported before[46,77]. Adapting LCA to marine conditions is, however, a novel contribution of this thesis. Whereas previous research often has concentrated on secondary remedial aspects[78], this work has also introduced the aspect of active versus passive remediation strategies allowing a combined evaluation of the totality of impacts. Increased use of LCA inspired by this work may have significant impact on selection of remediation strategies in the future. LCA is therefore a valid methodology to answer the research question 3.

The difference between traditional HERA results and results from the LCA are however substantial[44], and the LCA can therefore only be attempted for comparative assessment of remedial alternatives found to be acceptable through HERA. In addition, both LCA and HERA do not explicitly consider many social related factors important in the selection of sediment management alternatives and will require other methodology to be conclusive.

Research question 4

In the thesis, means of effective stakeholder processes has been discussed. The expansion of the participatory component to previously proposed multicriteria decision frameworks[66,79] for contaminated sediment management is novel. The multicriteria involvement process (MIP) with lateral learning, combined with MCA providing structure, robustness and transparent documentation has shown success in Bergen harbor and may be a valuable practical contribution to future management practices. However, it is important to remember that the emphasis on method and process should be balanced to be able to conduct these types of processes in practice.

The MIP is the first part of the answer to the fourth research question on how to structure a decision model. The developed model is particularly suited for participatory process early in the management phase ("bottom-up" approach).

The participatory approach is however less suitable for principal decisions. In a formal setting, direct stakeholder participation may in fact also be a source of conflict due to the exposure of direct weighting positions in a decision making process[61]. The proposed second integrative model is intended for use in principle decision making, where focus on accuracy and uncertainties are important aspects. One of the important aspects of this model is the importance of using different decision methodologies in plural. This view has been advocated as a necessity for complex environmental decisions previously[44,62] and is effectively concretized, developed and demonstrated in the thesis.

Use of SMCA will therefore be the second answer to the fourth research question and is particularly valuable for integrative decisions (“top-down” approach).

As opposed to existing fully quantitative cost-benefit (CBA) or cost-effectiveness (CEA) analysis this methodology gives only a relative rank order between alternatives. This calls, as demonstrated in this case, for a more qualitative interpretation of the results and a sufficient analysis of the robustness of the results compared to fully quantitative methods.

Overall research question

Implementing the proposed decision models giving focus to findings presented in the research will be a step towards sustainable decisions in contaminated sediment management, as questioned for in the overall research question. Although there are substantial advantages to use a multicriteria approach in complex decisions, there are also difficulties in relation to use. It will, for example, require more openness in the decision making process and complementary competence among experts and consultants[61,62]. For a successful implementation these challenges has to be overcome.

7.2 Methodological Aspects

Proper design of the research is fundamental in modern research[19]. According to the description of research design given in chapter 2, the research described in the thesis is best defined as subjective and case based supported by use of mixed methods.

The selection of a management case using study objects or individual cases consisting of conducted or planned sediment remediation projects has been an important methodological selection. A case based study gives many opportunities to study the effects of using health and ecological risk based management systems from variety of angles[22], which is a strength of the design. By using different individual cases in a multiple case design it is possible to work with different aspects within the management case in a relatively short period of time. This gives the thesis a depth that had been difficult to achieve by following only one single study object in a single case design.

The “field strategy” has proven advantageous since it gives the possibility to study methodological questions within a realistic setting avoiding use of hypothetical data sets. This increases the applicability of the results in a policy perspective and builds confidence that the models developed are not only theoretical abstractions, but actually robust methodology that can be applied in future management cases.

The weakness with case studies is naturally the validity of generalization of the results. In particular social factors and risk perceptive values are not static but are affected by a variety of different aspects. This means that repeating studies in order to reduce uncertainties in the outcome is in practice impossible compared to an objective experimental design. However, sustainability as a subject is a multidimensional concept involving economic security, social well-being, and environmental quality. Collaboration between the natural and social (environmental) sciences is therefore necessary to understand the complex nature of the problem[80]. To base such research only on experimental design is therefore difficult, and a case based design as selected in this thesis is therefore a preferential choice.

The research has used research methods typical for objective, constructive and subjective epistemology. This is preferential since a variety of methods also allow the case to be discussed from different angles, using conceptually different methodologies as promoted in multidisciplinary research[81]. Combining typical social science oriented methods as interviews and surveys with ecological impact modeling and statistical decision analysis enriches the case compared to a single disciplinary approach. At the same time working with multidisciplinary requires detailed knowledge of several research disciplines. It is necessary to build the work on already established principles and it is difficult to develop a detailed understanding of all aspects of the problem, which often is required in PhD work[82]. It is therefore a risk that multidisciplinary research can be superficial and inadequate. The thesis has dealt with this risk by actively seeking cooperation with different research disciplines to allow for satisfactory scientific depth of each topic, utilizing the nature of the cases to focus on specific issues and by using methodology in a contextual way to promote the subjective research objective. Even though research background always will govern the direction of the work, the implemented measures will help to overcome the challenges of multidisciplinary research[83].

7.3 Practical Applications

Execution of a contaminated sediment remediation project typically proceeds through specific project phases involving different actors in the process. Stakeholders tend to be involved late in the process as a part of formal hearings relating to permit applications or environmental impact assessments (EIA). The possibilities for changes in the plans at this stage of a project tends to be limited, which may cause problem owners to defend the chosen solution instead of ensuring a constructive stakeholder dialogue[84] allowing good cooperation between involved parties. This unfortunate situation may result in significant opposition that will lead to increased costs and delays in the execution phase of contaminated site remediation projects as shown in the thesis[14].

The multicriteria involvement process (MIP) developed in the thesis can be used early in the management process as a way to involve stakeholders and get valuable advice from local citizens and stakeholders, but also as a tool to uncover preferential differences between groups and to pin-point important aspects to consider in the project process.

However, a formal decision setting puts focus more on objectiveness, accuracy of results and handling of uncertainties. The developed SMCA-model for evaluating principal decision problems in a holistic and transparent manner is therefore an attractive alternative to traditional cost-benefit and cost-effectiveness analysis. Use of the model can be valuable in contaminated sediment management as well as in other complex environmental decisions.

7.4 Further Work

This thesis has investigated the Norwegian system of managing contaminated sediments using case based research design. As for all research using cases some of the findings are generally applicable, whereas others are more specifically related. It is therefore important for others to continue this kind of research and evaluate the findings and proposed methodology in a broader setting.

It is important to acknowledge that no single method will be sufficient for all situations or problems. Even though the extensive use of health and ecological risk analysis are challenged in this thesis, HERA is a very important tool for contaminated sediment management. More research on improving management models based on the risk reduction principle is therefore needed.

Research promoting use of a multicriteria framework is also of utter importance to improve decision management within this field. Methodological research and practical case based applications are necessary to facilitate use of multicriteria management within regulatory frameworks.

As quoted from Albert Einstein in the beginning of the thesis - to regard old problems from a new angle, requires creative imagination and marks real advance in science.

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Appendix A: Selected papers

P1 From Ecological Risk Assessments to Risk Governance. Evaluation of the Norwegian Management System for Contaminated Sediments

P2 Evaluation of Factors Affecting Stakeholder Risk Perception of Contaminated Sediment Disposal in Oslo Harbor

P3 Use of Life Cycle Assessments To Evaluate the Environmental Footprint of Contaminated Sediment Remediation

P4 Use of Multicriteria Involvement Processes to Enhance Transparency and Stakeholder Participation at Bergen Harbor, Norway

P5 Towards Sustainable Decisions in Contaminated Sediment Management by use of Stochastic Multicriteria Analysis

P1 - From Ecological Risk Assessments to Risk Governance. Evaluation of the Norwegian Management System for Contaminated Sediments

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Original Research

From Ecological Risk Assessments to Risk Governance: Evaluation of the Norwegian Management System for Contaminated Sediments

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ABSTRACT

Managing of contaminated sediments is a complex process that will naturally have to balance scientific, political, and economic interests. This study evaluates the Norwegian system for managing contaminated sediments toward a generic system for risk governance encompassing both knowledge, legally prescribed procedures, and social values. The review has been performed examining the management plans for 17 prioritized contaminated fjord systems in Norway. The results indicate a strong focus in the Norwegian management system on ecological risk assessment. This facilitates selection of local sustainable remediation measures, but may also complicate the balance toward other relevant interests in a decision-making process. The Norwegian system lacks management tools to identify and handle ambiguity through concern assessments and stakeholder involvement, and the decision-making process seems to a large extent based on ad hoc decisions, making it difficult to incorporate and document multicriteria evaluations into the management process. To develop a sustainable management system, encompassing environmental, economical, and social interests, a stronger focus on concern assessment and multicriteria evaluations is required. Integr Environ Assess Manag 2010;6:240–248. © 2009 SETAC

Keywords: Risk Governance   Sediment   Management   Sediment Quality Guidelines   Multicriteria decisions

INTRODUCTION

Sediment contamination in a river basin or an urban coastal area is often related to a complex situation involving diverse contaminant sources. The sources may have a local origin as effluent releases from present or former industrial sites, whereas other sources are diffuse, such as urban runoff or long-range atmospheric deposition. Sediments often act as a sink, accumulating contamination from all sources, thus making it likely to find elevated levels of contamination in large areas. Due to the nature of the problem, which is characterized by potentially significant volumes with contamination above background levels, the management of contaminated sediments will naturally have to balance cost, environmental aspects, and potentially conflicting social interests. Therefore, no single correct way exists to address sediment contamination; the approach should be driven by the ecological, political, and economic goals of all interested parties (Apitz and Power 2002). The balance between these interests is delicate and involves not only natural science-oriented aspects but is also dependent on social understanding, risk perception, and social acceptance among stakeholder groups and the public.

Norway has taken an approach for sediment management decisions based on site-specific risk assessment. Even though this may be a large step toward a sustainable management of contaminated sediments (Apitz 2008), allowing in situ remediation technologies and natural recovery to be used as remediation options, this study will evaluate the Norwegian management system for contaminated sediments examining the maturity toward a risk governance framework encompassing knowledge, legally prescribed procedures, and social values, and will suggest improvements to guide sediment management in a sustainable direction.

THE NORWEGIAN MANAGEMENT MODEL

Sediment contamination in Norway differs somewhat from the situation in the rest of Europe. The main concern with relation to sediments is not connected to river basins and the need for dredging to maintain navigational depth, but to the presence of contamination in the inner parts of the fjords and in harbor areas. Driving forces for management are increased environmental awareness and urban development in these areas.

The process of sediment management in Norway started in the late 1990s with an extensive investigation of contamination in more than 120 locations along the coast. Based on these investigations, a system for sediment quality guidelines (SQGs) was developed, categorizing the sediment in 5 classes, from background values (Class I) to severe contamination (Class V) (SFT 1997). Initially the different classes were set merely by statistical evaluation and expert knowledge, whereas in the latest revision of the guideline, the values are based on ecotoxicological data and derived probable no-effect values (PNEC) using the EU-TGD system (EU 2003). To complement the basic risk assessment based on SQGs, a system for site-specific risk assessment was developed

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allowing a more detailed assessment based on local site conditions. The local assessment considers risks to human health (exposure for contamination), spreading (to water and uptake in marine organisms), and ecological risk (damage to the local ecosystem) as areas of concern. In parallel with this assessment system, health guidelines for oral intake of fish and shellfish have been prepared. These restrictions on intake of fish and shellfish are based on expert judgment as well as direct analysis of biota and evaluation using a separate set of guideline values. At present, 32 coastal areas have restrictions on seafood consumption.

Based on the investigations and risk assessments, a strategy for mitigating sediment contamination has been developed and formalized through preparation of sediment management plans. These management plans are based on national governmental objectives stating that Norway should be a leading nation with respect to a clean marine environment, striving to reduce exposure to harmful chemicals as far as possible. Today, management plans for 29 areas have been prepared, targeting 17 of them for further remedial actions.

The sediment management strategy is formalized through 2 white papers from the Norwegian Government in 2002 and 2006 (MD 2002; MD 2006).

RISK GOVERNANCE MODELS

One way to expand the process of contaminated sediment management is to incorporate it into a broader risk governance perspective. Risk governance includes the totality of actors, rules, conventions, processes, and mechanisms concerned with how relevant risk information is collected, analyzed, and communicated and how a management decision is taken. One of the main aspects of risk governance is the acceptance and understanding of the duality of risk. In Klinke and Renn (2002), the duality of risk is discussed in terms of realism versus constructivism. The realists consider risk as a representation of observable hazards predicted by calculations unbiased by human views. The constructive camp considers risk as a mental model validated toward the logical consistency, cohesion, and internal conventions of logical deduction. This mental translation of risk is best described as risk perception and is well documented in the social literature (Slovic 2000). A transparent decision model should try to balance socioeconomic and political considerations with scientific evaluations into a governance framework.

Perhaps the most comprehensive conceptual risk governance framework is described by the International Council of Risk Governance (IRGC 2007; Renn 2008). In the literature, there are several descriptions of adaptations of the framework to different applications or development of new models for decision making using the fundamentals of the framework (Pollard et al. 2004a, 2004b; Kristensen et al. 2006; Assmuth and Hilden 2008; Pollard et al. 2008). This study, however, refers to the generic ideas of the 4-stage risk governance framework as presented by IRGC, in Figure 1.

The preassessment serves as the baseline for the risk assessment and management, giving guidance on both the dimension of the risk, the relevance and interests of the stakeholders and the public, as well as the existing foundations such as laws, regulations, and other relevant guidelines. The framework defines stakeholders as socially organized groups who are or will be either affected by the risk or have strong interests in the issue. The public is defined as

Figure 1. Conceptual model of the risk governance framework, IRGC (2007).

individuals, nonorganized groups, or media who are experiencing the outcome of the event or have an opinion on the issue (Renn 2008). The preassessment is important because it allows the duality of risk to be reflected early in the policymaking phase.

The second step is risk appraisal. This step identifies and assesses important information about the risk, to be used in the subsequent characterization and evaluation steps. The risk appraisal contains both the conventional scientific risk assessment based on the identification of hazard, exposure, vulnerability, and probability of occurrence. The framework also contains a concern assessment, which encompasses the associations and the perceived consequence that the stakeholder may associate with the hazard. This assessment will identify the potential bias between the scientific (realistic view) and the perceptive (constructivist view), allowing both sides to be reflected in the evaluation and management of risk.

The third step in the framework consists of risk characterization and evaluation. This step encompasses both the process of characterizing the risk according to the findings in the appraisal phase, as well as an evaluation of the tolerability and acceptability. For the scientific evaluation, this normally means evaluation toward predefined acceptance criteria, whereas this model also includes an evaluation toward social values based on the result of the concern assessment.

The final stage of the framework is the risk management phase. This phase comprises the identification of risk-reducing measures and the decision-making process as well as the design, implementation, and monitoring the effect of these measures. Communication is central in the process, indicating that communication with stakeholders is required to build trust in all phases of the framework.

One of the main elements in this conceptual governance model is to choose management strategy based on the characteristics of the risk. This system divides the risk into 4 different classes: *simple*, *uncertain*, *complex*, and *ambiguous*, depending on the characteristics of the risk. This classification may facilitate both problem framing in the preassessment phase as well as the characterization of risk in the characterization and evaluation phase and the need for stakeholder involvement in the management phase.

MATERIALS AND METHODS

The evaluation of the Norwegian management system for contaminated sediment has been performed as a case study analysis, adapting the methods described by Eisenhardt (1989). This method is based on analysis of cases to build new theory and to compare it with relevant literature in order to validate the results. The theory building proposed in the result chapter is based on analyzing each of the 17 fjord cases prioritized for further actions. The main sources of information for each case are the management plans available at the Web page of the State Control Authority Board (www.sft.no). Each of the management plans have been subjected to textual analysis with respect to consensus to requirements for information and with the 3 elements in the risk governance framework before the management phase itself. The following criteria for consistency have been used in the analysis:

Preassessment: Has one established and agreed on site-specific goals and objectives for the management of sediment contamination?

Risk appraisal: Did the material contain a risk assessment (RA), and was the classification of risk based on general sediment quality guideline values (SQG), or did it contain a site-specific RA? Was the probability of occurrence related to spreading of contaminants evaluated? Was there any evidence indicating that risk perception or public opinion had been evaluated in a concern assessment?

Risk characterization and evaluation: Was the evaluation performed toward predefined ecological risk acceptance criteria? Were qualitative or qualitative methods used to evaluate other socioeconomic factors toward the ecological criteria, or were management decisions taken using ad hoc decision methodology?

Information: Have stakeholders been involved in the analysis work or in the preparation of the management plans?

The performance of the Norwegian management system is analyzed with respect to incorporation of the principles of the risk governance framework, and the gaps toward the risk government framework are identified. The results are compared and discussed toward relevant literature. The relevance of using risk governance as a generic tool toward sustainable management of sediments is also discussed in the *Results* section.

RESULTS

Analysis of the framework

The results from the case study are summarized in Table 1. A positive finding toward the predefined criteria is marked as a positive sign in the table, whereas lack of evidence in agreement with the criteria is marked with a negative sign. The analysis shows that all plans have a strong focus on ecological risk assessment and have used site-specific sediment investigations and sediment quality guidelines to classify the contamination using ecological acceptance criteria. Sixty-four percent of the plans have also performed site specific evaluations of ecological risk, but only 1 of the plans has taken concern assessment into account. Three of the plans have made an evaluation of the probability of occurrence of contamination due to unwanted events. In the Kristiansand and Trondheim management plans, the probability of ship traffic causing sediment mobilization combined with the SQG was used to assess the risk of contaminant mobilization. In the Oslo fjord, the risk of unwanted incidents during the remediation was assessed by using semiquantitative risk matrices. However, in general the risk assessments are executed by assuming the probability of occurrence of the contaminant releasing processes to be equal to 1, i.e., the risk assessment is determining the potential consequence of exposure to contamination using that assumption.

The majority of the plans had incorporated some kind of qualitative decision making in the work, but only a few of the objects had used any kind of comparative or multicriteria-based decision making. The communication in form of stakeholder involvement and participation is found to be present in approximately 50% of the plans.

Case illustrating ambiguity in sediment management

Sediment management decisions are influenced by many factors, and there may be ambiguous characteristics to be considered in the management operations. This is illustrated in the remediation of sediments in the inner Oslo fjord (2006–2009), where perceived risk and social acceptance differ significantly from the risk assessed by the experts.

Due to construction of a new road tunnel in the Oslo harbor area, dredging of contaminated sediments was necessary and a remediation project was initiated. The selected disposal method was to place the sediments in a confined aquatic disposal facility (CAD) approximately 3 km from the dredging site. In addition, other contaminated areas in the harbor would be capped with clean material. The solution was selected as the most feasible option compared with other remedial alternatives and was documented through an environmental impact assessment (EIA). This solution was approved by the environmental agency (The Norwegian Pollution Control Authority [SFT]) and the local city council as an environmentally sound project for remediation of the inner Oslo fjord. National and local nongovernmental organizations (NGOs), however, opposed the disposal method, trying to stop the project during the implementation phase. The main objection to the project was the concept of using the fjord as a disposal site for contaminated sediments. The case has been highlighted in the media, and a search on the Google™ search engine in August 2009 on the combination of "Malmøykalven" (the name of the disposal site) and "gift" (poison) generated 962 hits. A search on the same keywords in Norwegian published material (television, radio, the Web, and newspapers) on the Retriever database (www.retriever-info.com) generated 351 hits. This indicates how the project has been associated with negative perceptive values and socially amplified through media (Kasperson et al. 1988).

Identification of gaps and comparison with literature

Preassessment—All work presented in the management plans is governed through the requirements given in the governmental white paper (MD 2002). The white paper gives the framework for the content in the plans and delegates the responsibility for the preparation process and further management.

The review of the targeted plans shows that approximately 60% of the management plans were containing accepted

Table 1. Review of management plans for the 17 prioritized Norwegian fjords and harbors

	Communi-cation	Pre-assessment	Risk appraisal				Risk characterization and evaluation		
			Risk assessment						
Location	Involvement of stakeholders in the process	Site-specific targets and objectives discussed and established	Classification according to SQG	Site-specific Ecological RA	Evaluating probability of occurrence	Concern assessment	Evaluation against ecological acceptance criteria	Qualitative decision analysis	Comparative or multicriteria decisions
Arendal	–	–	+	+	–	–	+	+	–
Bergen	–	–	+	+	–	–	+	+	–
Drammensfjorden	–	+	+	+	–	–	+	–	+
Farsund	–	+	+	+	–	–	+	+	–
Grenland	+	+	+	+	–	+	+	–	+
Hammerfest	+	+	+	+	–	–	+	+	–
Harstad	+	+	+	+	–	–	+	+	–
Kristiansand	–	–	+	+	+	–	+	–	+
Oslo harbor	+	+	+	+	+	–	+	–	+
Ranafjorden	+	+	+	+	–	–	+	–	–
Sandefjord	–	–	+	+	–	–	+	+	–
Stavanger	–	–	+	–	–	–	+	+	–
Sunndalsfjorden	+	+	+	–	–	–	+	+	–
Sørfjorden	+	–	+	–	–	–	+	+	–
Tromsø	–	+	+	+	–	–	+	+	–
Trondheim	+	–	+	+	+	–	+	+	–
Ålesund	+	+	+	–	–	–	+	–	–

site-specific environmental targets and goals, whereas other plans were merely presenting alternatives allowing the decision makers to decide on targets and objectives. The lack of specific goals and targets guiding the management work may have both advantages and disadvantages. The obvious disadvantage is that, without clear objectives and targets early on, the subsequent decision-making process will be less structured, possibly leading to more ad hoc decisions. On the other hand, the advantage is that this open approach may facilitate adaptive management (Linkov et al. 2006a), allowing several remediation alternatives to be evaluated until a more detailed level of understanding of the problem has been achieved.

Risk appraisal (risk assessment and concern assessment)—The analysis shows that all plans have used ecologically derived generic SQGs to evaluate the sediment conditions at the sites. A total of 80% of the plans have also performed site-specific ecological risk assessments based on the level of contaminants found in the sediments. Transport of contamination from the sediment out of the affected area has been estimated by flux calculations. Risk of human exposure has been estimated by comparing the potential of exposure to the maximum tolerable dose (MTR) for a lifelong exposure. The ecological risk is assessed by performing site-specific calculations of water concentrations derived from the measured sediment concentrations and comparing them with the water quality guidelines (WQGs). The work with ecological risk assessments has been performed according to Norwegian guidelines (SFT 2007).

The framework for performing ecological risk assessments (ERA) and establishing SQGs that are used in the first phase of the ERA is an extensive process. The process is based on available toxicity data for marine organisms, following a statistical interpretation to determine the water concentrations where 95% of the organisms should be protected, giving WQGs. SQGs are then calculated by using the distribution coefficient between sediment and water (K_d-value). There are several uncertainties involved in the process. First the calculation is based on toxicity data derived from aquatic species (PNEC values), which by themselves change over time due to increased knowledge and more scientific data. For components where limited toxicity data are available, safety factors are applied to be conservative in the estimations (EU-TGD 2003). Second, the conversion from WQG to SQG using the generic distribution coefficients between sediment and water may significantly underestimate the tolerable sediment concentrations for some components (Breedveld et al. 2007).

To test the robustness of ERA as a framework for decision making, the SQG values for several organic and inorganic components for 3 revisions of the Norwegian SQGs (SFT 2007) were compared.

Figure 2 shows the standard deviation from a calculated mean value for the 3 revisions (1997, 2005, and 2007). It can be seen that the deviation varies between ±10% and 150%, with large differences both in time and between components. All these variation may be well explained from evolving scientific data and better understanding of the ecological and chemical processes in the modeled system. However, for a risk manager or stakeholder who is only presented with the output from the evaluations, the impact on the interpretation may be significant. This highlights the sensitivity of a management system, only relying on the presentation of

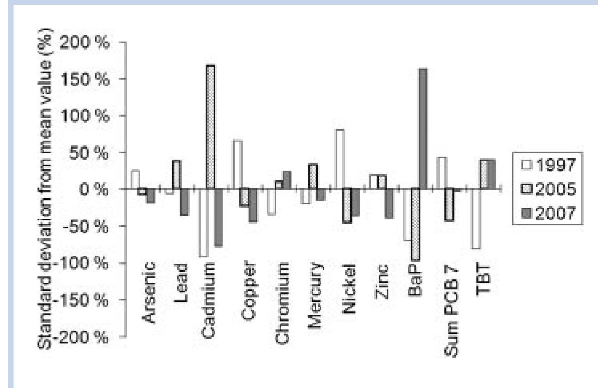

Figure 2. Variation in Norwegian SQG values for selected elements in the period of 1997–2007.

acceptable contaminant values for single components. This conclusion is in line with the findings by Apitz and Power (2002), in which several assessment frameworks were assessed and found useful for flagging potential contaminants in the sediments but less suitable for use as disposal or cleanup criteria.

Efforts to overcome these problems and reduce uncertainties have been undertaken by the standardization of derivation of SQGs from toxicological data through the EU-TGD framework. Another, more innovative approach to reduce the sensitivity of variations in the SQGs and the uncertainty in the evaluations is to use unitless indexes. Apitz et al. (2007) used a weighting method in which each component of interest is divided by the contaminant concentration, forming a contaminant–SQG value ratio (HQ) for each component. A mean ratio was also calculated (mHQ) by dividing the sum of all contaminant–SQG value ratios with the number of components. The system was tested for 19 different SQGs from various countries. The results indicated that, even if there were differences between the assessments of each component, the indexation gave relatively comparable results between the different set of SQGs.

A similar approach to risk assessment of sediment contamination has been taken in the Norwegian offshore oil industry (Singsaas et al. 2008). Here, the focus has been to assess the impact of cuttings from drilling operations on the environment. The main argument for developing a management system for the drill cuttings has been to assess which components in the drilling fluid have the greatest impact on environmental risk, and to compare the environmental effect of cuttings in different regional areas to be able to prioritize risk mitigation. In this system, both toxic stressors, such as the chemical components, and nontoxic stressors, such as oxygen depletion, are assessed. The risk is estimated by calculating the ratio between the concentration of the stressor, often referred as predicted environmental concentration (PEC), and the corresponding PNEC value. This PEC/PNEC ratio for each component is summarized and the integrated value forms an environmental impact factor (EIF).

From a risk manager's perspective, a transparent system using mHQ or EIF values would be valuable. The unitless indexes is less sensitive to changes in SQG values, because elevation of toxicity for some components may be leveled out by reduction of toxicity for other compounds. A unitless index also makes it easier to assess the overall effect of contamination between different locations or even regions, thus allowing the decision to make prioritize the remedial effort. It is, however, important to be aware that all weighing methods have limitations because chemical and toxicological data may get lost in the process. The results should be used with care, allowing full transparency of the background data, and should be subjected to sensitivity analysis.

Concern assessment is scarcely traceable in the analysis of the remediation plans. Only 1 of the plans, specifically that for the Grenland fjord, has addressed risk perception by assessing willingness to pay for remediation of contamination (Navrud and Barton 2006). This result indicates that the part of the risk governance framework relating to the duality of risk and taking perceptive risk into the management process, is only to a minor degree incorporated in the Norwegian framework for managing contaminated sediments. This observation may have 2 logical explanations. The first possibility is that risk of contaminated sediments may be defined as a simple, complex, or uncertain risk requiring low or no stakeholder involvement. The other possibility is that risk of contaminated sediments may have ambiguous characteristics and indicates a gap between the management system and the framework.

To test if it is possible to classify the Norwegian management model, the system for risk classification in the framework was used and adapted to sediment contamination (Table 2).

The review of the Norwegian management model clearly indicates that sediment management in general is not a simple issue, because site-specific ecological risk assessments have been widely used to evaluate the risk and prioritize remediation. To some extent, one may argue that the problem is complex because there may be multiple sources contributing to the effect of contamination in the ecosystem, including contaminant sources and chemical agents not yet identified as being of environmental concern. The management strategy is addressing these issues in such a way that the SQGs reflect the predominant contaminants at the time, and site-specific ecological risk assessments also recognize that there may be other sources contributing to risk.

The main focus in the management strategy is, however, on uncertainty-based management. The use of SQGs is a precautionary approach because the same toxicological assessment system is used to evaluate the environmental risk for new substances introduced to the market through the REACH directive (www.echa.europa.eu). Compared with other risks that people are freely willing to accept, such as smoking or car driving, the risk from contaminated sediments may seem insignificant. However, by using a precaution-based strategy, the handling of contaminated sediments in a way not causing additional risk to humans or the ecosystem is assured.

The use of the risk management system for sediment remediation operations has, as described in the analysis chapter, revealed at least 1 example in which perceived risk and social acceptance may differ significantly from the risk assessed by the experts. This indicates that sediment remediation also may have ambiguous characteristics. This is again evidence of the difference in risk perception between experts and stakeholders. This example emphasizes that sediment remediation may be ambiguous and that concern assessment in these cases should be a part of the management framework.

Risk characterization and evaluation—In 70% of the plans, the management strategy has been formed by using an ad hoc process, meaning that recommendations for remedial actions are not based on a systematic evaluation, weighing, and prioritization of the obtained data. In 4 of the plans (those for Grenland, Oslo, Drammen, and Kristiansand) there has been an effort to use comparative methods to propose management strategies. Three major strategies have been chosen.

The first method, used in the Grenland area, focuses on the environmental benefit a remediation may have on lifting the restrictions for consumption of seafood and shellfish (Saloranta et al. 2008). This method assesses the uptake of contaminants in relevant species from the contaminated sediments with and without remedial actions. The model calculates the time it takes to reach a state of conditions in which the consumption restrictions of fish and shellfish may be lifted. Based on the modeling results, a study on the willingness to pay to remove consumption restrictions, as well as the preference for different remediation alternatives, was initiated. The main conclusion from the study was that the willingness to pay for a remedial action increased in the vicinity of the contaminated area and that capping methods were preferred as the remediation method (Navrud and Barton 2006).

The second strategy, used in Drammen and Oslo, is based on the contaminant transport from an area before, during, and after remediation, comparing it with a reference state. The idea here is to calculate a scoring value (remediation efficiency), comparing the ratio between contaminant flux from a given area during and after remediation with the contaminant flux without remediation. For a given time frame, the remediation efficiency should be positive, i.e., indicating that the release of contaminants during and after the remediation should be less than or equal to an alternative without remediation (Eek et al. 2006).

The third strategy, used in Kristansand, uses an efficiency index in which the cost is divided by the amount of contamination removed. The site index gives the opportunity to assess the cost of remediation for 1 unit of contaminant. Even though these methods use different strategies to support a management decision, they are all founded in a physical–chemical model of the fate of the contaminants.

There are, however, also other methods to facilitate complex decisions associated with sediment management as described in Linkov et al. (2006a) and (2006b). These methods are based on multicriteria decision analysis (MCDA) and may better facilitate decision making, allowing other factors than the fate of chemicals to influence the decision. Unlike the management decision methods used in the Norwegian sediment management, MCDA is derived from the general need to solve complicated management problems and is therefore neutral in the sense of expert knowledge. The core potential of MCDA is the ability to structure, compare, and evaluate complicated management decisions involving both technical data and stakeholder values. The process is interactive, where all relevant factors affecting the decision are identified and weighed against each other. There are several examples in the United States in which MCDA has been used for management of sediment remediation cases (Yatsalo et al. 2007; Kiker et al. 2008; Suedel et al. 2008).

Although there are substantial advantages to use MCDA in complex decisions, there are also difficulties in relation to using the ad hoc method based on common sense (Gamper

Table 2. Risk classification and the consistency with the practice of the Norwegian management model. Adapted from IRGC (2007)

Risk classification	Risk characteristics	Management characteristics	Strategy for sediment management	Stakeholders involved	Consistency with the practice of the Norwegian management model
Simple	Linear, benefits of taking action is uncontroversial	Direct improvement expected by removal of sediments. Uncontroversial disposal of contamination	Routine based strategy, use of regulations based on standard SQG values	Regulatory bodies; experts	Low. Site-specific ERA normally required
Complex	Difficulties in identifying causal links between a multitude of potential causal agents and specific observed effects	Embodying exposure to multiple effects of contamination including new chemical agents	Collection of information to understand complexity. Periodical update of SQG reflecting the predominant contamination at the time.	Regulatory bodies; experts; external scientists; researches	Medium. Ecological RA recognizing multiple contaminant sources
Uncertainty	Refers to lack of clarity or quality of the scientific or technological data to assess the risk	Setting acceptance criteria precautionary to avoid adverse effects of sediment contamination	Precaution based and resilience focused strategies. SQG based on the same toxicological assessment system as for evaluating new substances (REACH)	Regulatory bodies; experts; external scientists; researches Affected stakeholders	High. Adverse effects of sediment contamination to be avoided when setting targets and objectives
Ambiguity	Divergent or contested perspectives on the justification or wider meanings associated with a given threat	Contested views between stakeholders and experts, especially regarding remediation solutions	Discourse-based strategy. Use of stakeholder participation processes. Transparent multicriteria decision making	Regulatory bodies; experts; external scientists; researches Affected stakeholders Public	Low. Stakeholder views are scarcely included. Oslo fjord case indicate the possibility of ambiguity

and Turcanu 2007). One of the disadvantages may be lack of experience with these methods among experts; another may be the need for extensive stakeholder participation and the sharing of knowledge between scientists, managers, and other stakeholders.

Communication—The traceability of communication in the management plan varies among the cases, but in 53% of the plans, participation from property owners and environmental advisors in the local municipality has been identified. By the definition of Rowe and Frewer (2000), the participation in the work may be classified as citizen/public advisory committee. This way of involvement is characterized with a general moderate degree of representativeness and transparency. The influence for decision making varies from case to case, depending on the local decision-making process. The quality of communication and success of involvement by using this method may therefore be questioned compared with other negotiation-based methods.

DISCUSSION AND CONCLUSIONS

The review of the Norwegian management plans indicates that management is heavily influenced by the system for ecological risk assessments and especially the use of SQGs, Figure 3.

The preassessment has been formed over a long time period involving knowledge buildup, investigations, and policy making. The management system is dominated by the view of regulatory authorities and experts on ecological risk assessments, developing the management system, and performing the assessments. This selection of expert competence has framed the system and contributed to encapsulate it from developing into a governance system incorporating the broader picture of risk.

The strong dependence on SQGs as management indicators may also be challenging, because there are uncertainties in establishing these kinds of criteria and they have been changing over time. It is also evident that the focus on consequence-oriented ecological risk assessments may enhance the conservativeness in the management system and limit the practical use as an efficient management tool to reduce risk. To reduce the impact of uncertainties, standardization in the form of weighing and grouping of environmental indicators may be beneficial to increase the transparency and allowing comparison of environmental impact between different locations or even regions. Three major deviations from the risk governance framework are observed in the study.

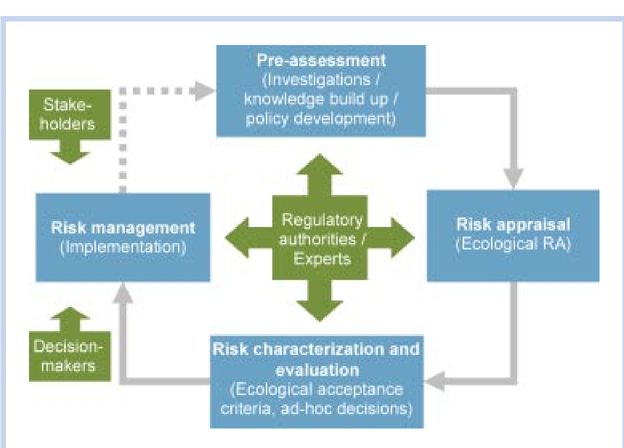

Figure 3. Work process for the existed management system of contaminated sediments in Norway. Adapted from IRGC (2007).

Primarily, the use of concern assessment is low, mainly explained by the strong focus on uncertainty management. Because the study shows indications of an ambiguity gap between expert judgment and stakeholder perception in at least 1 of the performed remediation cases, the management decisions may be biased due to the lack of knowledge about the socioeconomic concerns and public attitudes related to the remediation operations. A stronger focus on concern assessment earlier in the decision-making process will reduce this gap.

Second, the decision-making process in the management system is weak. Even if the different data to support a structured management decision process are in place, most of the recommendations for implementation of sediment management for the different locations are based on ad hoc processes. Implementation of a decision analysis framework supporting multicriteria decisions could be beneficial for a more transparent process. This will, however, require a new type of competence in the management process and should be assessed carefully before introduction.

Third, the involvement of stakeholders and decision makers in the analyzed management plans, and the communication during the process may be categorized as advisory, mainly influencing issues related to the management phase. A stronger involvement earlier in the process will reduce the ambiguity, but will also require a paradigm shift in the regulatory framework.

Finally, it is recommended to direct further research toward methods that may facilitate concern assessment and enhance transparent decision-making processes for sediment management. A well-functioning system for identification, weighing, and prioritization of environmental indicators encompassing environmental, social, and economical factors will contribute to create a sustainable management system for sediment contamination.

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Evaluation of Factors Affecting Stakeholder Risk Perception of Contaminated Sediment Disposal in Oslo Harbor†

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The management of environmental pollution has changed considerably since the growth of environmental awareness in the late 1960s. The general increased environmental concern and involvement of stakeholders in today's environmental issues may enhance the need to consider risk in a much broader social context rather than just as an estimate of ecological hazard. Risk perception and the constructs and images of risks held by stakeholders and society are important items to address in the management of environmental projects, including the management of contaminated sediments. Here we present a retrospective case study that evaluates factors affecting stakeholder risk perception of contaminated sediment disposal that occurred during a remediation project in Oslo harbor, Norway. The choice to dispose dredged contaminated sediments in a confined aquatic disposal (CAD) site rather than at a land disposal site has received a lot of societal attention, attracted large media coverage, and caused many public discussions. A mixed method approach is used to investigate how risk perceptive affective factors (PAF), socio-demographic aspects, and participatory aspects have influenced the various stakeholders' preferences for the two different disposal options. Risk perceptive factors such as *transparency* in the decision making process and *controllability* of the disposal options have been identified as important for risk perception. The results of the study also support the view that there is no sharp distinction in risk perception between experts and other parties and emphasizes the importance of addressing risk perceptive affective factors in similar environmental decision-making processes. Indeed, PAFs such as transparency, openness, and information are fundamental to address in sensitive environmental decisions, such as sediment disposal alternatives, in order to progress to more technical questions such as the controllability and safety.

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Introduction

The rapid rise of environmentalism in response to problems caused by pollution, particularly since the late 1960s, has had a considerable impact on how environmental policy issues and mitigating measures are handled (*1*–*3*). Briefly, roughly from the early 1970s there was increasing recognition among the public that simply diluting and dispersing environmental contamination was not sufficient or acceptable. Thus, solutions to prevent emissions in the atmosphere and in water were introduced and heavily imposed with regulations and legislative actions. From this stage the policies have evolved, and broader interest groups play direct or indirect roles in environmental policy making, as environmental issues have steadily become an increasing public concern.

Policy development for the management of contaminated sediments has lagged behind development in other areas. Part of this is related to the ambiguous nature of regulating polluted sediments. Many sites are contaminated from previous activities ("old sins") and by diverse pollution sources, making it unclear who bears the burden of blame or remediation. Contaminated sediments are therefore still generally managed through a strong postpollution regulative focus similar to the early stages of environmental policy (*4*), rather than through a preventive focus. In Norway and some other countries, however, the awareness of preventive measures has grown, and precautionary ecological risk assessments, which are used to identify, characterize, and quantify environmental hazards, has been advocated (*5*).

As with other environmental issues, involvement of the public in sediment management has become more evident and should be addressed. Owing to such involvement it is necessary to consider risk assessment and management in a much broader context than earlier (*6*). Whereas ecological risk assessments evaluate hazards from contaminated sediments to be related to toxic effects for humans and the ecosystem, certain members of society may use a more intuitive assessment of the risk involved. The distinction between this statistically estimated risk and public acceptability was early identified and addressed as risk perception (*7*). Previous research has documented that risk perception may differ significantly from statistical estimations and is affected by social acceptability (*8*). Later research has nuanced this view, suggesting that risk perception depends on both rational and more intuitive arguments (*9*).

Suggestions on how to address risk in public management ranges from scientific concepts trying to influence and alter risk perceptions via communication and education using scientific risk assessments (*10*), to the more pragmatic approach where the scientific results from risk assessments compete with the outcome from participatory processes (*11*). Other intermediate viewpoints where risk perception is addressed, evaluated, and taken into account in the management process by experts and decision makers are also referred to in literature (*12*).

The gap in risk perception between different parties in the management process may, according to empirical research, only be bridged through communication and involvement, and by placing the same emphasis on lay perception as is placed on technical knowledge (*13*). On the other hand, diversity in risk perception may also be an asset since it avoids concealing important hazards. Examples of such behavior were found in the Former Soviet Union where unwanted hazards were regularly concealed (*14*). Complete

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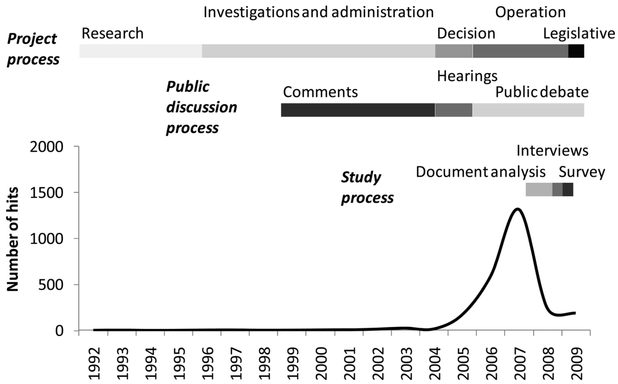


FIGURE 1. Overview of the project and related public discussion process as measured by number of materials published in Norwegian published media (television, radio, the Web, and newspapers) found in the Retriever database (www.retriever-info-com) using the search word "Malmøykalven".

consensus may therefore be both unrealistic and in many cases unwanted.

In this paper we use a contaminated sediment remediation project in Oslo harbor Norway, which has been subjected to substantial social involvement, as a study object to investigate the possible effect of risk perception in the choice of alternative disposal solutions for contaminated sediments. Our study is part of a larger research project aiming to assess methods for improved stakeholder involvement in contaminated sediment management (*15*). The main aim of this retrospective study is to assess whether it is possible to identify risk perceptive factors among the involved participants and to investigate *how* and *why* these factors have affected the view on the disposal alternatives. An additional aim is to identify how risk perception is encompassed in a societal context (*16*). The results herein provide useful recommendations for future stakeholder involvement processes in contaminated sediment management.

Materials and Methods

Study Object. A major sediment remediation project was conducted in Oslo harbor, Norway, during the period 1992−2009. Navigational requirements, urban development, and environmental concern initiated the dredging of approximately 300,000 m^3 of contaminated sediments in the inner harbor area. One of the major issues in the project was related to the disposal of this contaminated sediment after dredging. Two principally different solutions were evaluated during the planning phase. One solution involved transportation of the dredged material on barges to a land disposal site, situated approximately 80 km from the harbor. This site, NOAH Langøya, is a national disposal facility for hazardous waste. The second option was to construct a confined aquatic disposal site (CAD) at Malmøykalven. This site, a 70-m deep sea-basin 3 km from the dredging area, has previously been used for uncontrolled disposal of dredged material.

During the long history of the different project phases public interest and discussion topics changed, as indicated in Figure 1.

The project process started with "research" period that assessed the potential consequences of contaminated sediments to people and environment. This period was followed by a sediment "investigation and administration" period to map the present situation and to come up with potential remedial solutions. Assessing the feasibility of using the CAD at Malmøykalven was an important activity during this phase. Both the use of the CAD and transport to the site with barges were subjected to an environmental impact assessment (EIA). The proposed solution was evaluated against a no-remediation scenario and was found to be feasible. Alternative disposal solutions were only briefly discussed in the EIA. After several political delays, the need to find a solution became urgent in 2004 due to urban development in the harbor area and the construction of a submerged road tunnel. During the brief "decision" phase a development plan was produced and a formal decision process was initiated. This process was finalized in 2005 and resulted in the decision to start the dredging activities immediately and to use the CAD as a disposal solution. The operation started early in 2006 and continued until mid 2009 during the "operation" phase.

Simultaneously to this project process a public discussion process was initiated. This began with a "comment" period, and involved receiving comments to the EIA from the public during the period 1999−2003. In the "hearings" period of 2004−2006 the plans for development and remediation of the area were subjected to formal hearings and public meetings were conducted. As illustrated in Figure 1, media interest in the project started to increase during this period. This suggests how the project started to be associated with perceptive values that were socially amplified through media interest. This pattern of increased media interest during public discussions corresponds to findings from other projects (*17*). During the operation period, the remediation project received substantial societal attention such as civil disobedience actions, protests campaigns and public debate, referred to as the "public debate" phase, most of which were directed toward the chosen remediation operation and the environmental monitoring of the process. As seen in Figure 1, the debate also dramatically influenced media coverage.

Data Collection. Data were collected to reflect the views of the stakeholders involved in the project rather than the general public opinion. Stakeholders are defined here as people, organizations, or groups who are affected by the issue and who have the power to make, support, or oppose the decision or who have the opportunity to provide relevant knowledge to the decision-making process (*18*).

This research is based on the case study method by Yin (*19*) with a mixed method approach to combine the strength of quantitative and qualitative investigation methods (*20*). In this study, interviews and analyses of documents are used as support for a survey, which presented below. This was conducted during the later stages of the operation and public debate period (see Figure 1). Triangulation of results was performed using the validating quantitative data model (*20*). In this model, the quantitative results and conclusions from the survey are validated with qualitative data by using results from the interviews. The idea to base risk perceptive research primarily on quantitative data is advocated by Sjöberg (*21*), who emphasized the need to simplify the interpretation by singling out dominating and important themes by use of statistical methods.

The data collection started with a qualitative review of project-relevant documents and materials as scientific reports and official correspondences. Through use of this material, stakeholders who had been active in the decision-making process were identified and on this basis a list of stakeholders consisting of 160 people and organizations was established.

From this list, a subset of 33 key stakeholders was selected. The key stakeholders were presumed to be the most *influential* and *interested* persons in the process, based on the following definitions. *Influence* was defined as the potential to affect the process either through formal legislative rights or by informal mobilization through media and financial instruments. *Interest* was defined by the potential level of benefits or losses the stakeholder could experience from the process. Like influence, interest was categorized into formal interests such as regulative issues and informal interests such as gain or loss of image and popularity. In-depth interviews were conducted with 23 key stakeholders

TABLE 1. Overview of Generic and Project Specific Affecting Factors (PAF) Influencing Perceptions of Risk[a]

Generic perception affecting factors	Potential project perception affecting factors
Voluntariness	Risk attitude
Knowledge	Degree of involvement General confidence Information about the process Transparency and independence Objectives for choice of disposal solution
Endangerment	Controllability of the solution Environmental effect
Reducibility	Usability of fjord and disposal area after remediation

[a] Adapted from ref *22*.

during the autumn of 2008 (67% participation). No particular pattern of reasons for not participating in the study was evident during the process. Interviews were performed in the stakeholders' environment or in a neutral place and were based on a questionnaire that was distributed before the interviews; see Supporting Information (SI) pages S4−S8. Stakeholders were interviewed anonymously due to the degree of conflict in the project. The questions were mainly open ended to facilitate discussion with the key stakeholders.

To confirm and support the main conclusions from the interviews an anonymous web survey with closed questions relating to the above-mentioned topics was conducted during the winter 2009. Questions are presented in SI pages S9−S14. Recruitment to the survey was based on the original stakeholder list of 160 people, omitting interviewed key stakeholders and people without valid e-mail addresses. This resulted in a list of 92 names. In addition, interviewed key stakeholders were sent an e-mail with the link to the survey with a request to forward the survey to persons they considered suitable. The survey included questions that were tailored to identify and exclude responses not relevant to the proposed stakeholder population definition. The survey received 87 valid responses within a time period of 44 days, whereof 49% were directly recruited parties and 51% were forwarded answers. The response rate among the recruited was 50%. The answers consisted of 29% female and 71% male responses. The majority of the respondents (55%) were between 41 and 65 years old. Sixty-five percent of the respondents lived in Oslo, but people living in the vicinity of the disposal site were also represented (23%). The vast majority of the respondents (94%) had university education (Bachelor, Master, or PhD).

Identification of Risk Perceptive Factors and Their Relationship. One of the ways that risk tolerance can be related to particular situations is through perception affecting factors (PAFs) (*22*). These generic factors were initially developed to estimate perceptive risk for natural hazards, but may after adoption also be used as a basis for defining PAF related to risk perception of the CAD in the Oslo harbor project, Table 1.

The four main PAFs summarized in Table 1 are voluntariness, knowledge, endangerment, and reducibility. *Voluntariness* relates to the risk attitude of people and the willingness to take risks. *Knowledge* incorporates a broad spectrum of items relating to information, general confidence, involvement, and transparency, as well as formulations of objectives. *Endangerment* incorporates the question on how the risk may affect humans and the environment, either negatively or positively. Finally, *reducibility* relates to possible negative considerations associated with use.

Statistical analyses, described below, were conducted to assess whether it was possible, based on the survey data material, to identify and relate any of the PAFs to the perceived risk of the CAD. The study used exploratory factor analysis based on the principal component method (PCA) to identify underlying factors based on the survey model questions. PCA as well as subsequent analyses of variance (ANOVA) and reliability testing were performed using the statistical package SPSS 17.0 (*23*).

Structural equation modeling (SEM), normally used in psychological research, was used to identify structural relationship between the identified factors. SEM combines factor analysis and multiple regression in one operation using model fit indicators to validate the proposed models (*24*). Unlike PCA, which explores the structural relationship between an infinite set of parameters, SEM confirms or rejects a proposed model structure based on a given set of input parameters. The software package AMOS 7.0 (*25*) was used for the SEM modeling.

The statistical modeling consisted of five parts. *The first part* identified PAFs in the data material from the survey by using a two-stage explorative factor analysis procedure (*26*). The procedure started by using all measured linear scaled model questions from the survey to identify underlying patterns in the data material and to select which model questions should be retained in subsequent analysis. To maintain sufficient statistical power in the data material, missing values were replaced using the expectation-maximization (EM) method, SI Table S1. EM uses a recommended iterative algorithm to estimate missing values based on the entered data material (*27*). A theoretical framework for the model question selection is presented in SI page S16.

The factor analysis was then repeated using the retained model questions. The mean factor scores of the latent factors were used for further assessment and statistical testing. The results were triangulated against the results from the interviews.

The second part of the statistical work investigated the correlation between the identified PAF and the perceived risk related to the CAD. The question about perceived risk had been included in the survey as a separate model question. This investigation of correlation was performed using a linear regression model with risk perception as the dependent variable (DV) and the identified PAFs as independent variables (IV). Only IV's with significant correlation to perceived risk of the CAD were retained for subsequent analysis.

The third part of the modeling involved a sensitivity analysis of the results. Since some weaker model questions and factors had been discarded, it was essential to perform a sensitivity analysis on the discarded model questions to assess whether the procedure of model question selection had the potential to bias the results.

The fourth part used SEM to test different structural models assuming that a relation existed between perceived risk of the CAD as a dependent variable and the significantly correlated PAFs identified in the second part. The structural models were validated against a model with no structural relationship.

In *the fifth and last part* of the statistical analysis, the perceived risk related to the identified PAFs was correlated to the preferential disposal solutions of the respondents (the selected aquatic disposal or the alternative land disposal solution) and was analyzed using a one-way ANOVA. The same method was also used to assess whether socio-demographic and participatory aspects were important for the outcome of the process.

TABLE 2. Factor Loadings and Cronbach Alpha Scores, α, for the Model Questions Relating to Project-Specific PAFs (Absolute Values Greater than 0.5 Are Considered to Be Correlated)

		Factor analysis results for the project-specific perceptive affecting factors (PAF)[c]			
		Controllability	Workability objectives	Health−Env. objectives	Transparency
Model question	N^a	$\alpha^b = 0.77$	$\alpha = 0.72$	$\alpha = 0.74$	$\alpha = 0.68$
Added value in addition to env. effect (scale 1−5)	76	−0.16	**0.87**	0.07	0.05
Importance of local solution (scale 1−5)	77	−0.16	**0.87**	0.07	0.05
Reduced human risk (scale 1−5)	77	0.09	0.07	**0.88**	−0.16
Reduced marine risk (scale 1−5)	78	−0.07	−0.05	**0.88**	0.15
Sufficient time for decision making (scale 1−5)	83	0.15	0.04	0.21	**0.72**
All research material accessible (scale 1−5)	85	0.01	0.02	−0.10	**0.90**
Perceived risk of sediments upon project term. (scale 1−3)	81	**0.88**	−0.04	−0.02	−0.10
Spreading of contamination from the CAD (scale 1−3)	80	**−0.79**	−0.07	−0.03	0.04
Future effect of CAD on the fjord (scale 1−5)	82	**−0.72**	0.11	−0.08	−0.19
Effect of CAD on future fish/shellf. cons. (scale 1−5)	55	**0.72**	0.03	−0.01	0.25

[a] Number of respondents before missing value replacement. [b] Cronbach alpha reliability value. A value above 0.70 is normally considered to be acceptable (*29*). [c] Expressed as factor loadings ranging from 0 to ±1. Factor loadings above 0.5 or below −0.5 are shown in bold.

The outcome of the statistical analysis was used to conclude what implications risk perception may have on future disposal projects.

Results and Discussion

Determining Perceptive Affecting Factors. The two-stage exploratory factor analysis procedure described above substantially reduced the number of model questions retained for analysis and gave a proposed structure of four latent factors in the data material (SI Figure S4). Table 2 shows the results of the factor analysis. The factor loadings given in the figure express how well the model questions correlate with each other. The four retained factors shown in the table explained 75% of the variance in the data material (SI Table S6). To evaluate the reliability of each factor, Cronbach alpha, α, which is a reliability indicator for sampling consistency (*28*) was measured. The values ranged from 0.68 to 0.77, where a value above 0.70 is normally considered to be acceptable (*29*).

The first PAF *controllability* incorporates perceived effect, spreading of contaminants, potential change in future consumption patterns, and perception of sediment risk after project execution. This PAF incorporates both endangerment and reducibility, which were not possible to distinguish between in the analysis. The second and third PAF, *workability* and *health−environmental objectives*, respectively, relate to stakeholders' objectives when selecting the preferred disposal solution. The analysis clearly distinguishes between reduction in human and environmental risk by using the preferred solution and objectives related to the workability of the solution, such as the importance of handling contaminated sediments locally and the importance of an added value other than reduction of environmental risk. The fourth PAF *transparency*, also relates to knowledge, and specifically to transparency in the decision-making process with emphasis on accessibility and sufficient time to involve stakeholders in the decision.

The identified PAFs based on the results of the web survey, presented in Table 2, are consistent with results from the in-depth interviews presented in Table 3, as will be elaborated below.

A majority of the interview respondents felt that aquatic disposal had a different risk than other solutions and mentioned different arguments related to controllability, including chemical stability, spreading of contaminants during disposal, weather and stream conditions, as well as long-term effects, as important in risk assessment.

TABLE 3. Arguments, Relating to Determined PAF, Assessed As Important by the Interviewed Key Stakeholders

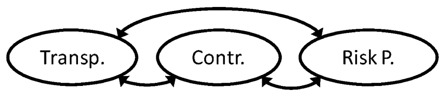
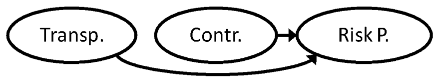
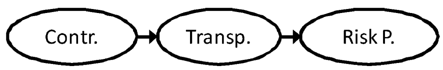
Identified PAF	Arguments in interview responses	Response rate (%)
Controllability	Different risk for aquatic disposal compared to other solutions	77
Workability objectives	Importance of cost, safety and performance for the decision on solution	81
Health and environmental objectives	Importance of human risk reduction, environmental risk, contaminant transportation	77
Transparency	Open discussion	4
	Information/communication	50
	Public decision making	13
	Involvement	32
	Independent control	14

Approximately 80% of the stakeholders interviewed mentioned health and environmentally related objectives (reduced contaminant transportation, reduced bioavailability, etc.) and workability objectives (cost efficiency, safety, performance) as important objectives in the choice of preferred disposal solution.

As to transparency, a number of items relating to participation, such as information/communication, involvement, public decision making and independence, were mentioned as important items in the decision-making process. This observation was more pronounced in the interview results compared to the survey results which merely concluded on transparency as one of several PAFs potentially affecting perceived risk.

PAFs vs Risk Perception. The relationship between the identified PAFs and perceived risk of the CAD, which had been measured directly as an interval scaled variable, was determined through a multiple regression analysis using risk perception as the dependent variable (DV) and the identified factors as independent variables (IV). The results of a *t* test showed significant correlation for *controllability* ($t = 2.13$; $p < 0.05$) and *transparency* ($t = -4.56$; $p < 0.05$) against perceived risk, whereas *health−environmental objectives* and *work-*

TABLE 4. Statistical Analysis (SEM) of the Structural Relationship between the PAFs Transparency and Controllability, with Risk Perception

	Model alternatives	Validation parameters. Recommended values in brackets, (*24*)				
		C_{min} [a]	df [b]	C_{min}/df [c] (< 2)	CFI [d] (> 0.95)	RMSEA [e] (< 0.10)
1		23.751	12	1.979	0.955	0.107
2		61.832	13	4.756	0.815	0.209
3		30.997	13	2.384	0.932	0.127
4		23.909	13	1.839	0.959	0.099

[a] The C_{min} value assesses the discrepancy between the model and a perfect fitting model. [b] Degrees of freedom in the model. [c] The relationship between C_{min} and the degree of freedom. By calculating C_{min} ratio versus the degrees of freedom, the validity of the model fit can be normalized and assessed (*30*). [d] The Comparative fit index (CFI) assesses the closeness to a perfect model (*31*). [e] The Root mean square error (RMSEA) estimates the lack of fit compared to the perfect model (*32*).

ability objectives were found to be uncorrelated ($t = -1.03$; $p = 0.30$ and $t = -1.47$; $p = 0.14$, respectively) with this variable.

Sensitivity Analysis. One important item in the PAF factor analysis is outcome sensitivity with respect to the model questions selected. The study represents a substantial sample of the population, which is satisfactory. On the other hand the sample material is limited and has been subjected to a missing value analysis, which may reduce the statistical reliability. A sensitivity analysis performed using a modified approach that included additional factors that had initially been discarded did not identify additional dependent variables compared to the initial solution (see SI Table S12). The results from this modified approach showed that *controllability* was still correlated to perceived risk when more model questions were included, whereas *transparency* was no longer correlated (SI Table S14). In an ideal situation the web survey should have been altered and repeated for the ambiguous model questions. However, due to the sensitivity of the project, the web survey was performed anonymously and was conducted in an ongoing project process and could therefore not be repeated. Since the results from the interviews confirmed the survey results the initial approach was retained.

Structural Relationship. The possibility of a structural relationship between the PAFs *controllability* and *transparency* with perceived risk of the remediation solution was identified using different structural relationship models.

Structural relationship models (models 2−4) were compared to a "test" model (model 1) in which no structural relationship between parameters was assumed to exist, see Table 4. A presentation of the comprehensive results is found in SI page S27− S31.

The different models are assessed by using a number of evaluation parameters that are recommended in psychological research (*33*). As evident from Table 4, model 4, which shows that risk perception is dependent on controllability which is dependent on transparency, is the only model that fits better than a model with no structural dependence between the parameters (model 1). This relation can only be identified through structural equation modeling and may be important to notice in future stakeholder involvement processes.

Correlations with Preferences in Disposal Alternatives. A variance analysis was performed to investigate whether risk perception and related PAFs had affected the preferences for the disposal solution (CAD/land) and therefore also had affected the potential outcome of the decision-making process. By using the *F*-test, systematic variation in the data material exceeding random variation was investigated. The results show significant differences relating to *risk perception* ($F = 56.3$; df = 1; $\alpha = <0.05$) and the structural related PAFs *controllability* ($F = 27.2$; df = 1; $\alpha = <0.05$) and *transparency* ($F = 26.8$; df = 1; $\alpha = <0.05$) for the alternative solutions. With respect to stakeholders' objectives for the choice of a solution, no differences were found relating to *workability* ($F = 0.18$; df = 1; $\alpha = 0.67$). For the *health and environmental objectives* the *F*-test showed a significant difference between the groups ($F = 5.7$; df = 1; $\alpha = 0.02$). However both groups evaluated this factor as important (value of 2) or very important (value of 1) for their choice of disposal solution. This makes it plausible to assume that differences between the groups in practice are minor. See also SI Table S25 for more information.

These findings support the view that perceived risk and underlying PAFs are indeed vital for choice of preferred remedial solution and therefore may be an important factor to address when selecting disposal solutions in contaminated sediment management. This view is also consistent with the results of the interviews where respondents preferring a land solution often expressed skepticism with regard to the controllability of an aquatic disposal site, especially on a long-term basis. The same respondents also often questioned the openness of the management process.

Socio-Demographic and Participatory Aspects. To assess whether stakeholders' preferences for different disposal options were affected by socio-demographic and participatory aspects, a similar variance analysis was performed for these parameters, see Table 5.

TABLE 5. Variance Analysis of Socio-Demographic and Participatory Aspects for the Alternative Solutions (CAD and Land Solution) showing *F*-Test Values and Corresponding Significance

Subject	Item	Category	*F*	Sig[a]
Socio-demographic aspect	Age	(1) 0−18, (2) 18−40, (3) 40−65, (4) > 65	1.48	0.29
	Gender	(1) female, (2) male	0.32	0.57
	Education	(1) no formal, (2) primary school, (3) secondary school, (4) Bachelor, (5) Master, (6) Master ext., (7) PhD	6.02	<**0.05**
	Work status	(1) unemployed, (2) student, (3) retired, (4) government empl., (5) company empl., (6) NGO, (7) freelance	0.15	0.70
	Residence	(1) at site, (2) vicinity, (3) Oslo, (4) outside Oslo	1.54	0.22
Participatory aspect	Year involved	(1) 1993−2004, (2) 2004, (3) 2005, (4) 2006, (5) 2007−	0.28	0.60
	Reason for involvement	(1) listener, (2) knowledge supplier, (3) critical observer, (4) participant	<0.01	0.98
	Cause	(1) job, (2) interest only, (3) NGO	3.43	0.07
	Function	(1) outside decision process (private, journalist, NGO) (2) within the decision process (governmental, politician, consultant/researcher)	13.95	<**0.05**
	Primary information source	(1) project web NGO webs, (2) scientific reports, (3) meetings, (4) communication with project, (5) personal expertise, (6) project web, (7) NGO webs	0.12	0.73

[a] Bold face values indicate parameters where the *F*-test give a $\beta \neq 0$ (95% confidence).

For the socio-demographic aspects the only systematic variance was found for education, where respondents with extended Master or higher degrees were more in favor of the selected solution, CAD (see also SI table S26). It is interesting to see that geographical location, which tends to disfavor disposal solutions close to residential areas (NIMBY-effects) (*34*) was not a significant distinguishing element in choice of preferred disposal solution in this case.

Limited variance was also seen for the participatory aspects. The only systematic variance identified related to the stakeholders function in the project, was where persons assumed to be closer to the decision-making process such as politicians, governmental organizations, and consultants/researchers were more in favor of the chosen solution (CAD) than persons assumed to be outside the decision-making process such as private persons, journalists, and NGOs. The findings are consistent with the results from the interviews, which indicated that people closely involved in the project were more in favor of the selected solution than respondents with more peripheral connections to the project organization. Interestingly, among the interview respondents that were critical to the chosen solution were some experts. However these experts were generally peripheral to the decision-making process. This critical attitude among the peripheral experts may be a sign of risk aversion (*35*), but is not contradictive to the identified PAF of transparency of decision making and controllability as influencing the preferential choice of disposal solution.

The results of this study are not consistent with the view that there is a sharp distinction in the risk perception of experts (who traditionally make risk estimates) and other stakeholders (who are primarily following individual interests independent from expert opinion). The results also support the view that stakeholders can be very well informed and thus may form alternative expert opinions based on various information sources (*36*). This finding is consistent with other studies which emphasize familiarity, attitude, and trust (and distrust) as important factors affecting risk perception, rather than demographic aspects (*37*).

Implications for Future Remediation Decision Making. A majority of the attention of the Oslo harbor remediation project has been directed toward the selected aquatic disposal solution for contaminated sediments. The management decision or the decision-making process itself with regard to the disposal solution may therefore be considered as the catalyst for the resulting social uneasiness. The stakeholders' preferences for disposal solutions were, with the exception of *education* and *risk aversion*, not impacted by socio-demographical and participatory aspects. This study therefore strongly indicates that management processes in projects concerning contaminated sediments need to address the societal context and the broader interpretation of risk, particularly questions related to the PAFs *controllability* and *transparency*.

In linking stakeholder values and knowledge (*16*), the sediment remediation project in Oslo harbor may be characterized as a moderately structured problem with a high degree of convergence in values, in this case expressed by remediation objectives, but a low convergence in perceived knowledge, in this case represented by the perception of the risk involved. Thus, increasing the transparency of the decision-making process, particularly on items related to controllability, is recommended to account for in policy. To address this kind of situation, Hischemöller (*16*) recommends a stakeholder involvement process using science-based negotiated policy. This management strategy involves the use of knowledge accepted by the actors who have an interest in the issue (*38*). This strategy is also advocated in the framework of the International Risk Governance Council (IRGC) (*12*, *39*) for ambiguous issues with conflicting risk perceptive views. Several strategies have been previously described for stakeholder involvement in contaminated sediment management that, like this one, recommend participatory processes aided by decision analysis techniques such as multicriteria decision analysis (*40*−*42*).

This case study supports the view that there is no sharp distinction in risk perception between experts and other parties involved. Nonexpert stakeholders may be very well informed, adopting their alternative expert opinion based on the various information sources available. As this study confirms, further research on methods that allow for more open and transparent stakeholder involvement processes are warranted to assist in future management decisions.

Acknowledgments

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Supporting Information Available

More information on the statistical analysis and modeling methods as well as background information is available free of charge via the Internet at http://pubs.acs.org.

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P3 - Use of Life Cycle Assessments To Evaluate the Environmental Footprint of Contaminated Sediment Remediation.

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POLICY ANALYSIS

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Use of Life Cycle Assessments To Evaluate the Environmental Footprint of Contaminated Sediment Remediation

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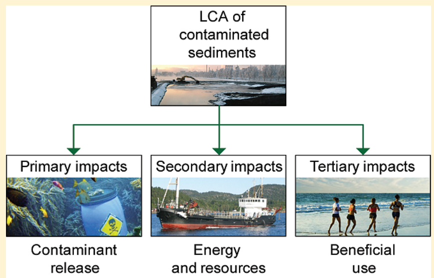
S Supporting Information

ABSTRACT: Ecological and human risks often drive the selection of remedial alternatives for contaminated sediments. Traditional human and ecological risk assessment (HERA) includes assessing risk for benthic organisms and aquatic fauna associated with exposure to contaminated sediments before and after remediation as well as risk for human exposure but does not consider the environmental footprint associated with implementing remedial alternatives. Assessment of environmental effects over the whole life cycle (i.e., Life Cycle Assessment, LCA) could complement HERA and help in selecting the most appropriate sediment management alternative. Even though LCA has been developed and applied in multiple environmental management cases, applications to contaminated sediments and marine ecosystems are in general less frequent. This paper implements LCA methodology for the case of the polychlorinated dibenzo-*p*-dioxins and -furans (PCDD/F)-contaminated Grenland fjord in Norway. LCA was applied to investigate the environmental footprint of different active and passive thin-layer capping alternatives as compared to natural recovery. The results showed that capping was preferable to natural recovery when analysis is limited to effects related to the site contamination. Incorporation of impacts related to the use of resources and energy during the implementation of a thin layer cap increase the environmental footprint by over 1 order of magnitude, making capping inferior to the natural recovery alternative. Use of biomass-derived activated carbon, where carbon dioxide is sequestered during the production process, reduces the overall environmental impact to that of natural recovery. The results from this study show that LCA may be a valuable tool for assessing the environmental footprint of sediment remediation projects and for sustainable sediment management.



INTRODUCTION

Selection of sediment management alternatives for contaminated sediments is often based on human and ecological risk assessment (HERA) frameworks.[1] The Grenland fjord in Norway, which is contaminated by polychlorinated dibenzo-*p*-dioxins and -furans (PCDD/Fs), exemplifies this risk based approach for selection of remedial solutions. In this case, capping of the contaminated sediments has been proposed to mitigate risk above the HERA-derived threshold values in fish and shellfish.[2] The risk-reducing effectiveness of different capping alternatives in current studies is based on the ability to reduce the flux of PCDD/F from the sediments below threshold levels, thus neglecting the environmental footprint of these materials originating from production, use, and disposal. As result, energy and resource intensive advanced capping alternatives may be recommended solely based on HERA.

Whereas HERA is suitable for assessing whether the contaminated sediments constitute an unacceptable human and environmental risk, it does not address environmental consequences aggregated over the whole life cycle of the remediation project and from intended future site use. Even though high-end-capping alternatives may reduce the risk associated with sediment contamination, the material production and placement necessary for implementing these alternatives as well as the energy and equipment use they necessitate, may result in environmental hazards that have not been quantified by traditional HERAs. One common way to determine the relative environmental impact between product systems occurring over the whole life cycle is by use of life cycle assessments (LCA). In this method the inputs, outputs, and the potential environmental impacts of a product system are compiled and evaluated throughout the product's life span.[3] In LCA of contaminated sites, impacts have normally been referred to as primary, secondary, and tertiary effects.[4] Primary effects originate from the contamination source, in this case

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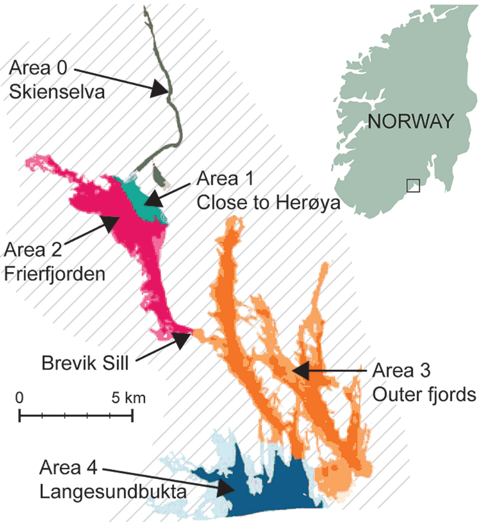

Figure 1. Bathymetric map of the horizontal compartment division in the model application to the Grenland fjords.[12] Different colors indicate the horizontal division of five compartments, while the shading within a color indicates the different bottom depth intervals used in the vertical compartment division.

intended effects of reducing PCDD/F uptake in sea food, local ecotoxicological effects on the benthic fauna, and physical local impacts of the capping operation. Secondary impacts are the effects related to the use of resources and energy during the implementation of a thin layer cap. Tertiary aspects of the remediation may include increased recreational use of the area or increasing commercial fishing after lifting the dietary notice. However, these tertiary effects were considered to be too uncertain and speculative to be included in the study.

Use of LCA in soil remediation projects has shown that the risks originating from the remediation process often exceed the environmental impacts associated with the site contamination.[5,6] Even though life cycle impacts of environmental management in aquatic ecosystems are gaining interest in both academia and industry,[7] LCA has rarely been used in sediment management. One explanation may be that LCA was originally developed primarily for land applications, and the current impact models are therefore only partially applicable to aquatic conditions.

In this paper we use the Grenland fjord remediation case to investigate the feasibility of using LCA to assess the environmental footprint of contaminated sediment remedial alternatives. Based on the results, we generalize and discuss the possibilities for the future use of LCA in contaminated sediment management.

■ MATERIALS AND METHODS

Case Description. The contamination in the Grenland fjord area is primarily due to historical industrial activities occurring from 1951 to 2002. The fjords system consists of an inner system (Figure 1, area 0−2) and an outer fjord (area 3−4), separated by the Brevik sill, which significantly reduces the flux of contaminants from the inner to the outer part of the fjord system. The present paper investigates the effect of capping the sediments in the most contaminated inner area of the fjord (areas 1 and 2).

The fate of contaminants has been modeled by using a multicompartment fate model, linking the abiotic processes describing the fate of chemicals from the sediments into the ecosystem, with the biotic process describing the fate of chemicals in selected marine species.[2] The performed HERA uses toxic-equivalent-based (TEQ) factors to calculate the risk originating from exposure to PCDD/Fs by expressing concentrations in 2,3,7,8-tetrachloro dibenzo-p-dioxin (TCDD) units.[2]

Due to elevated levels of PCDD/Fs (app. 200−300 ngTE/kg ww)[8] in fish and crayfish above the threshold established by the Norwegian Climate and Pollution Agency, the Norwegian Food Safety Authority has issued a dietary advisory for consumption of fish and shellfish from the area. In the management plan,[9] sediment capping has been proposed to further reduce the risks associated with sediment contamination. The long-term objective is to remediate the sediment and transition the site to unrestricted use for public recreation and commercial fishing. The model results indicate that capping has to cover a substantial part of the fjord in order to be effective.[2]

Remediation Alternatives. Due to the size of the remediation area, only thin layer capping of the contaminated sediments has been considered as a feasible remediation method.[9] The use of either passive material to reduce the PCDD/F flux or active carbon containing materials adsorbing PCDD/F[10] have been suggested as viable options. An ongoing large-scale pilot project in the Grenland fjord is currently evaluating the feasibility of using this method as a remediation method for the site. In this pilot project three materials are used: locally dredged clay, crushed limestone from a regional source, and activated carbon (AC).

The capping materials used in the pilot study are also used in this LCA study with one exception; in the field trials, AC is mixed with clay; however, here AC alone is assumed as a plausible future scenario. Two different sources for the production of AC are also included in this LCA study: a fossil anthracite coal-based product from China and a biomass-derived AC from India utilizing coconut waste as starting material. In the field trial only anthracite AC is used. From a holistic environmental perspective, the biomass derived AC differs from anthracite-produced AC, since it is based on a renewable material. In addition, a net carbon sequestration effect may result from the amendment of the biomass-derived AC to the seafloor instead of its combustion as a fuel.[11,12]

LCA Approach. The LCA investigates the environmental footprint of the active and passive capping materials considered as plausible remediation alternatives and compares them with the footprint of a natural recovery scenario from natural resedimentation. The assessed system can be divided into production, use, and disposal phases (Figure 2). The production phase is relevant for passive and active capping materials and relates to impacts from material production, transportation, and the capping operation. The use phase includes contaminant release during the phase when the cap will be active in reducing the contaminated flux from the sediments. Impacts in this phase are relevant also for the natural recovery scenario. Public recreational activities and fishing are assumed for all alternatives in the use phase. Impacts related to monitoring the performance of the cap are considered to be outside the scope of this analysis, since it is governed through national monitoring programs independent of remedial strategies. Since the capping materials will eventually be a part of the natural seabed, no environmental impact connected with disposal is foreseen.

The inflow consists of the use of raw materials and energy consumption to produce, transport, and apply materials. The outflow consists of emissions to the various relevant compartments: air, water, soil, and sediment. Resource use and effects due

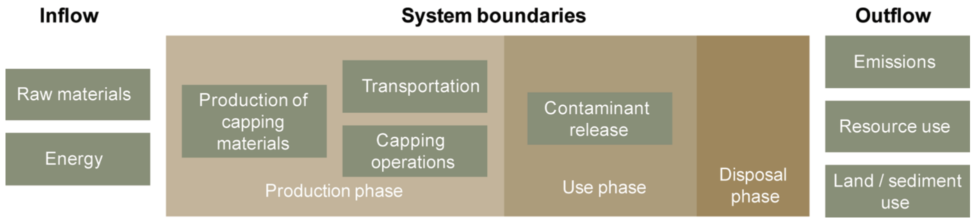

Figure 2. System boundaries for the different capping scenarios assessed in the study. The natural recovery scenario will only have impacts related to contaminant release in the use phase.

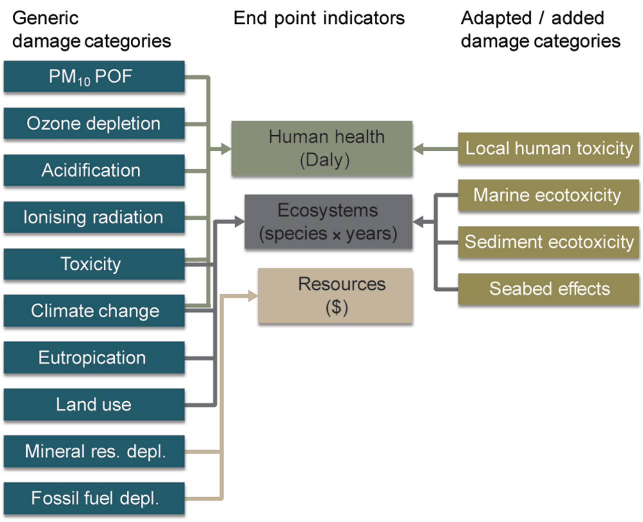

Figure 3. Combination of the generic and adapted/added damage categories into end point indicators for the ReCipe impact model used in the study.

to the physical impacts of land and sediment use are also addressed in the analysis.

Functional Unit. Based on recommendations for a life-cycle framework for the assessment of site remediation,[13] the functional unit is set equal to the remediation of an area of sediments the same size as to the whole inner fjord (23.4 km^2), conservatively assessed for a 90 year time period. This is assumed to be longer than necessary for a successful natural recovery scenario estimated to be approximately 35 years.[2]

Inventory Analysis. The life cycle inventories, i.e. the aggregated environmental data collected for the modeled system, are derived from three sources. The main source used for the majority of processes is the Ecoinvent 2.2 database. This includes production data for limestone, transport data, and energy data. Contaminant fluxes have been calculated with the local fate model using the same settings as in earlier studies.[2] All production and emission data for AC production as well as estimates for diesel consumption during dredging and capping have been obtained from the vendor (Jacobi Carbon. Ragan S and Agder Marine Høyvold P; personal communication 2010). An overview of the inventory data used in the analysis, with reference to their source, is given in the Supporting Information (SI) (Figures S1 and S2 and Tables S1−S8).

Impact Assessment Methods. The marine application of LCA has implications on the choice of methodology used to convert the inventory data into information about environmental effects. Marine aquatic toxicity, which is important for this study, is scarcely addressed in available impact models for toxicity.[14] Sediments, if included in the models, are normally seen as a sink and not as a source for marine contamination. The ReCipe impact model[15] which utilizes USES-LCA[16] is at present the only readily available impact assessment method that includes a marine release compartment and was therefore selected for this study. The UNEP-SETAC UseTox initiative[17] targeted to develop a multimedia chemical fate, exposure, and effect model does not address marine ecotoxicity presently and has therefore not been used here.

An end point method was used for the impact assessment in order to achieve maximal agreement with the comparative and management-oriented objectives of the study (Figure 3). End point indicators describe the integrated damage of the components from the inventory, in contrast to midpoint indicators which address effects only. For global warming, a typical midpoint indicator would be the effect of radiative forcing (global warming potential), whereas the end point approach would assess the human and environmental damage based on radiative effects. Use of end point indicators facilitates the interpretation of results for management purposes and allows integration of results to a single score indicator. However, end point indicators are expected to have a higher degree of uncertainty compared to midpoint indicators.[18]

Local Model Adaptations with Regard to Marine and Human Toxicity Effects. The USES-LCA is a multimedia effect model combining a contaminant fate model and an effect model for the estimation of toxicological effects by use of characterization factors (CFs) for human toxicity and ecotoxicity. The CF is an integrated value based on factors describing the contaminant fate (FF) and toxicological effect (EF) and is calculated for each substance (j) and emission compartment (i); soil, water, and air

$$CF_{i,j} = FF_{i,j} \times EF_{i,j}$$

The strategy in the present study was to use the best available information to adapt CFs to assess toxicity to the local fjord system and to add these locally derived CFs to the generic CFs from the USES-LCA model, which assesses consequences on a continental scale as the minimal resolution.[19] The contaminant flux between the inner and outer fjord was assumed to be the interface between the local adapted model and the default USES-LCA model. Fluxes in the inner fjord were assessed as a part of the local system, whereas the fluxes to the outer fjord were assessed to be a part of the continental scale and incorporated in the default model (Figure 4).

FFs for the local-scale-impact-model adaptations have been based on TCDD flux, water, and sediment concentrations using the local abiotic transport model,[2] see the SI (Figure S5). For sediments, ecotoxicological effects are assumed to be related to the pore water only,[14] converting sediment concentrations into pore water concentrations using the sediment pore water partition coefficient (K_d), see the SI (Table S9). For all effect calculations, the standard EFs from USES-LCA 2.0 were utilized.

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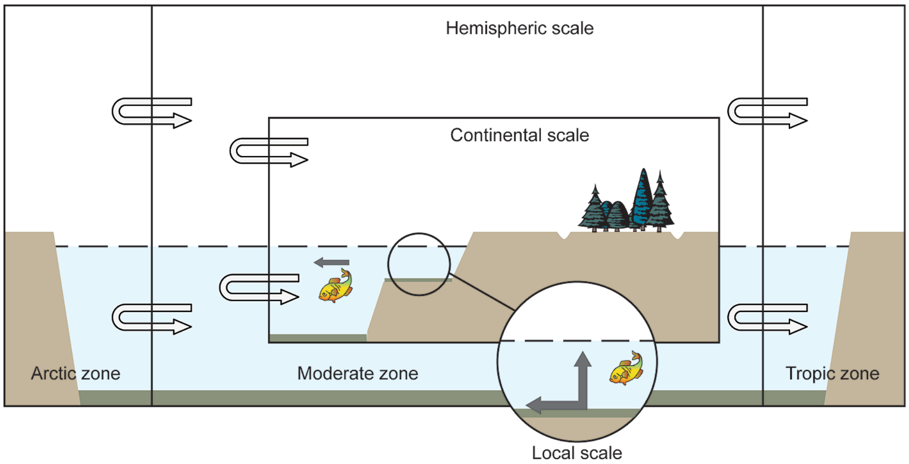

Figure 4. Incorporation of environmental effects into the USES-LCA model by introducing a local scale. The dark arrows show direction of contaminant fluxes to water and sediment-pore water. Fluxes through the Brevik sill are considered to be the connection between the local scale and continental scale models. Adapted from ref 19.

For the characterization of human toxicity, the USES-LCA model assumes the consumption of fish as the single exposure pathway. In this case, an intake fraction of fish (IF) was calculated using locally derived values for contaminant fate and exposure. Of note is the fact that the intake rate (IR) of fish, which depends on the ratio between areal population and the volume of the water compartment, is significantly higher for the local fjord compared to generic values (SI Table S11). As for ecotoxicity, the fate calculations are combined with the generic USES-LCA 2.0 effects factor (EF) values describing the toxicological effects via oral ingestion of PCDD/F exposed fish. The locally calculated CFs are given in SI Table S10.

Local Model Adaptations with Regard to Sediment Use. One topic not previously introduced in LCA is changes in the benthic fauna caused by the physical impact of a capping operation. Effects may be caused by e.g. depletion of oxygen due to degradation of capping material, sediment burial, or variations in grain size between the cap and the natural seabed.[20] For capping with clean materials, oxygen depletion due to degradation is not relevant. However, sediment burial, referred to as *sediment occupational effects*, and variations in grain size, referred to as *sediment transformational effects*, are necessary to consider. In both cases a five-year time horizon may be anticipated for these postcapping effects.[21] By using the relationship between the cause of hazard and the ecological effect, expressed as potential affected fraction of species (PAF), the CF for seabed effects was calculated as follows (see ref 22)

$$\mathrm{CF}_{\mathrm{seabed}_{\mathrm{e}}\mathrm{ff}} = 5 \times \frac{0.5\mathrm{PAF}}{\mathrm{HS50}}$$

The cause of hazard for *occupation* (HS_o) is given by thickness of the cap and for *transformation* (HS_t) is given by the difference in grain size between the capping material and the natural seabed. HS_o and HS_t were determined based on work performed by Smit et al.[23] (SI Table S12).

Normalization and Weighting. Using a normalization process allows damage effects to be transformed into unitless indexes (ecopoints) and thus allows a comparison between impact categories. Both *external normalization* relating effects against an external reference situation and *internal normalization* where results are related internally are relevant methods to apply in LCA. In this case external normalization was selected to facilitate the relative significance of results across categories, even though this also assumes a delineation of effects within a spatial and temporal resolution.[24] The estimated effects from the study were normalized against the effects from the annual contaminant releases of 28 European countries during the year 2000 scenario,[25] using end point characterization factors from ReCipe (www.lcia-recipe.net) for effect calculations (SI Table S15).

Weighting may be applied in order to summarize damage effects into single score indicators. This study has weighted the different effect categories using the following weights: ecosystem 40%, human health 40%, and resource use 20%, thus reflecting the time horizon and the objectives of common policy principles emphasizing ecosystem damage and human health to resource use.[15]

The use of indicators, normalization, and weighting has been heavily debated,[26−28] since all approaches have advantages and disadvantages. For this exploratory and comparative study, a pragmatic view utilizing recommended values has been used. The results are however discussed with respect to model sensitivity and it is applicability to contaminated sediment remediation.

■ RESULTS AND DISCUSSION

Primary Effects Affecting the Fjord System. The normalized impacts values of the different remediation alternatives affecting the fjord system are given in Table 1. Based on primary effects, all active remediation scenarios were favorable compared to a natural recovery scenario. Impacts of human toxicity dominated over impacts of marine and sediment ecotoxicity. Local toxicity impacts were also higher than regional impacts. These findings are as expected due to the chronic nature of PCDD/Fs toxicological effects and the higher exposure in the local fjord system model as compared to the background level. The physical impact of the capping operation on the benthic community is also relatively high and outweighs the ecotoxicological effects. These findings are supported by experimental data indicating that the physical effects of a capping operation may have a significant short-term impact on the benthic fauna compared to the chronic toxicological effects.[29,30]

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Table 1. Normalized Impact Values (Ecopoints) for Primary Effects of Contaminated Sediments[c]

impact effect	compartment[b]	nr	clay	limestone	anthracite AC	biomass AC
human toxicity[a]	local	122	24	24	6	61
	regional	4	7×10^{-2}	7×10^{-2}	2×10^{-2}	0.2
marine ecotoxicity[a]	local	3×10^{-4}	5×10^{-5}	5×10^{-5}	1×10^{-5}	1×10^{-4}
	regional	1×10^{-5}	2×10^{-6}	2×10^{-6}	6×10^{-7}	6×10^{-6}
sediment ecotoxicity	local	2×10^{-5}	5×10^{-6}	5×10^{-6}	1×10^{-6}	1×10^{-5}
sediment transformation	local	-	-	86	-	-
sediment occupation	local	-	12	12	0.9	0.9

[a] The reduction of accumulated contaminant flux due to active capping in comparison to the natural recovery scenario is as follows: clay and lime 80%, anthracite AC 95%, biomass AC 50% [b] Local compartment refers to the fjord specific characterization factor, whereas regional refers to use of generic impact factors from USES-LCA 2.0. [c] This includes local and regional effects for human toxicity and marine ecotoxicity as well as local sediment ecotoxicity of PCDD/F. It also includes local sediment transformational (difference in grain size) and occupational (cap thickness) effects of the capping operation.

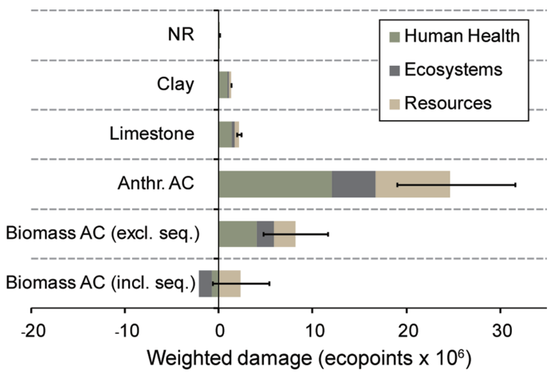

Figure 5. Normalized and weighted results (ecopoints $\times 10^6$) obtained using the ReCipe hierarchist end point with the European normalization values and the average weighting set.[25] The standard deviation (SD) for the alternatives was calculated based on Monte Carlo simulations using the predefined SD for the single unit processes and the SD for the flux calculations (SI Figure S4). A distribution of SD between the end point indicators is given in SI Table S17.

Overall Impacts Including Secondary Effects. Figure 4 presents the overall normalized and weighted results; detailed results, including unweighted data, are presented in the SI (Tables S13 and S14). Each stack in the figure contains the integrated weighted value of the potential effects on human health, ecosystem damage, and use of nonrenewable resources. In contrast to the primary impact results, the overall impact was higher for the active capping alternatives than for natural recovery, thus the resources used for active remediation (see SI Table S18) were not compensated for by the gains from toxicity source reduction. This is consistent with LCA studies for contaminated soil[31] and indicate that the amount of energy and resources necessary to remediate contaminated sediments result in a large environmental footprint, especially for use of anthracite based activated carbon. Evidently the carbon sequestration effects of using biomass-based AC[11,12] is important with respect to overall life cycle impact, and if this effect is incorporated in the LCA this alternative exhibits a reduced environmental footprint that allows it to be compared with a natural recovery scenario. The degree of allocation of carbon sequestration for use of biomass-derived AC is a subject of discussion,[12,32] and Figure 4 therefore shows a case with and without this allocation.

Uncertainty and Sensitivity Analyses. Uncertainties in LCA may originate from sources related to data, methodological choices, and impact assessment model.[26] In this study, uncertainties connected to inventory data are addressed by the use of standardized inventories and locally derived values. The error bars given in Figure 5 represent the combined uncertainties in qualitatively estimated uncertainty values[33] from the unit processes in SI Tables S5−S8. The error bars for natural recovery are based on standard deviation in the abiotic fjord model, see SI Figure S4. Methodological and impact related uncertainties have been addressed through careful choice of the base impact model and through model adaptation to fit the local setting, with the inclusion of site specific effects like sediment use, as described in the methodological section. Different weighing sets will also effect the absolute values of the weighted damage potentials. However, the relative order between the alternatives is only to a minor degree effected (SI Figure S6).

The results of the LCA are sensitive to variations in the input data, and changes in the inventories may have substantial impacts on the results. In Figures S7 and S8 in the SI the sensitivity to changes in the operational dredging efficiency (diesel use) and material efficiency (cap material use) is presented. Even though higher efficiency is beneficial in both cases, operational efficiency is more important for locally derived capping materials, whereas engineered materials with higher life cycle impact in the production phase benefit more from higher material efficiency. In contrast, biomass-derived AC including sequestration is nonsensitive to operational and material efficiency, since the positive carbon sequestering effect outweighs the negative impacts in the production phase.

In addition, variations in contaminant concentrations may affect the results, especially for the natural recovery scenario. This study averages PCDD/F fluxes over the whole inner fjord system according to the selection of the functional unit. By narrowing the scale further, the effect of natural recovery will vary depending on the local sediment contaminant concentration within the fjord. However, in order for an active remediation scenario to be beneficial from a life cycle perspective, PCDD/F fluxes have to be 2 orders of magnitude higher than the scenario used (SI Figure S9) which is unrealistic.[34]

Future Use of LCA in Contaminated Sediment Management. Sustainable sediment management can only be achieved by a holistic approach toward assessing remedial alternatives. This study shows that LCA may be a valuable tool for assessing the environmental footprint of sediment remediation projects and can be used for prioritization and optimization of remedial alternatives from a life cycle perspective. Even technologies with a relatively low resource-intensity, such as thin layer capping, can have a significant environmental footprint which approaches that of site-specific implementations for some of the more resource intensive solutions (e.g., dredging and disposal).[35] The use of LCA in contaminated sediment management would enhance the relative attractiveness of remedial solutions with limited raw material and energy use. LCA may be especially relevant for addressing beneficial sediment and alternative energy uses, such as the use of biomass-derived AC instead of coal based AC as discussed in this paper.

There are many issues that need to be carefully considered in implementing LCA for sediment management. In this paper, the environmental risk factors associated with sediment contamination have been extended to incorporate effects associated with the implementation of sediment management alternatives. The difference between traditional HERA results and results from the LCA are however substantial,[36] and the LCA can therefore only be attempted for comparative assessment of remedial alternatives found to be acceptable through HERA. The comparative nature of such LCA implementation allows for dealing with the uncertainty that is attracting increasing attention within LCA and HERA communities.[26] Even though many parameters may be uncertain, they are likely to result in similar over- or under-estimation of risks for all considered alternatives and are thus unlikely to affect the final ranking.

The question of relevant scale and focus is important for both LCA and HERA. In general, HERA considers the local scale and focuses on risk of specific stressors, while LCA operates on a global scale, normalizing and weighting impacts for relative comparison. As for other specific LCA applications,[37] the results from this study emphasize the necessity of including a local compartment to the impact assessment model for future LCA applications in coastal areas to reach an acceptable resolution in the impact assessment. Even so, based on the standardized normalization and weighting procedures applied in this study, the damage from primary aspects are assessed as relatively minor compared to the secondary aspects. From a life cycle perspective, contaminant levels have to be substantially higher to justify commonly accepted remediation practices, which may contradict public values. Therefore, instead of basing the weighting on standardized damage categories more focus may be given to the perspective of the decision maker, thus giving higher focus to local (primary) effects than global (secondary) effects in the LCA.

In addition, both LCA and HERA do not explicitly consider many factors important in the selection of sediment management alternatives. One way to address this may be to assess the tertiary effects related to the remediation.[38] Examples of such effects would be increased recreational use of the area or increased commercial fishing after lifting the dietary advisory. This approach would, however, require a more developed system for monetization of social and economical impacts.[39] Establishing a more complex cause and effect related weighting systems may, on the other hand, reduce the transparency of the study and increase the use of controversial criteria which is undesirable.[40]

An alternative to avoid controversial weighting procedures is to combine LCA and multicriteria decision analysis (MCDA). MCDA integration would allow tertiary effects to be added separately to the standardized LCA results, and the weighting between impact categories could be assessed using values elicited from stakeholders also incorporating uncertainties in the evaluation.[41] Further research may be directed toward developing such an integrated framework for sustainable sediment management.

ASSOCIATED CONTENT

Supporting Information. More detailed information about the LCA assumptions and detailed inventory results as well as detailed results from the impact analysis. This material is available free of charge via the Internet at http://pubs.acs.org.

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Use of Multicriteria Involvement Processes to Enhance Transparency and Stakeholder Participation at Bergen Harbor, Norway

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ABSTRACT

Use of participatory stakeholder engagement processes could be important to reduce the risk of potential conflicts in managing contaminated sites. Most stakeholder engagement techniques are qualitative in nature and require experienced facilitators. This study proposes a multicriteria involvement process to enhance transparency and stakeholder participation and applies it to a contaminated sediment management case study for Bergen Harbor, Norway. The suggested multicriteria involvement process builds on the quantitative principles of multicriteria decision analysis and also incorporates group interaction and learning through qualitative participatory methods. Three different advisory groups consisting of local residents, local stakeholders, and nonresident sediment experts were invited to participate in a stakeholder engagement process to provide consensual comparative advice on sediment remediation alternatives. In order for stakeholders or residents to be able to embrace a complex decision such as selection of remediation alternatives, the involvement process with lateral learning, combined with multicriteria decision analysis providing structure, robustness and transparent documentation was preferable. In addition, a multicriteria involvement process resulted in consistent ranking of remediation alternatives across residents, stakeholder, and experts, relative to individual intuitive ranking without the multicriteria involvement process. Integr Environ Assess Manag 2011;7:414–425. © 2011 SETAC

Keywords: Stakeholder involvement   Contaminated sediment management   Multicriteria decision analysis   Citizens jury

INTRODUCTION

Emerging environmental challenges coupled with increased stakeholder awareness and concerns call for more effective stakeholder involvement processes for environmental management. A structured stakeholder involvement process could help in overcoming disagreements and result in better management alternatives (Slob et al. 2008). Examples of qualitative involvement processes include focus groups with facilitated communication between parties to reach consensus (Kitzinger 1995). Cooperative discourse methods are also described by Renn (1999) involving establishment of development criteria and alternatives using value trees elicited by stakeholders and experts in round table meetings. Group Delphi is another systematic, interactive forecasting method that relies on a panel reaching consensus through sequential use of questionnaires and intermittent discussions.

Multicriteria decision analysis (MCDA) has been proposed as a method to enhance stakeholder involvement in sediment management and to facilitate decision making of complex problems (Linkov et al. 2005; Yatsalo et al. 2007; Kim et al. 2010). The purpose of MCDA in these studies has been to support evaluation and selection among management alternatives in an interactive process involving decision makers, stakeholders, and scientists. Methodologically, MCDA requires developing a hierarchy of criteria and metrics to compare management alternatives and subsequent elicitation of weights to quantify relative importance of criteria, as well as scoring of alternative performance based on these criteria. The MCDA approach overcomes the limitations of unstructured individual and group decision making by providing decision transparency and focusing discussion on assessing the weights and scores. Thus, MCDA may be valuable in quantitative decision making; however, focus on participatory aspects in the involvement processes for sediment management is also warranted (Sparrevik et al. 2011).

In order to enhance the value of participatory stakeholder involvement in environmental management, we propose a multicriteria involvement process (MIP) that builds on the quantitative principles of MCDA and also incorporates group interaction and learning through qualitative participatory methods. The process bears resemblance to earlier proposed MCDA processes for sediment management (Kiker et al. 2005; Alvarez-Guerra et al. 2010; Hong et al. 2010). However, this process also addresses recruitment and includes an involvement and learning step inspired by deliberative decision making using citizens' juries (Smith and Wales 2000; Soma 2010). The application of the MIP is illustrated for a sediment remediation case at Bergen Harbor, Norway, by conducting the process for 3 different advisory groups: local residents, local stakeholders, and nonresident sediment experts. A comparison of individual versus group consensus-based ranking of alternatives is also presented.

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THE MULTICRITERIA INVOLVEMENT PROCESS

Stakeholder involvement in contaminated site management

A project execution process for managing contaminated sediments typically proceeds through specific project phases involving different actors in the process. Problem owners are usually active in the problem formulation and the approval phase where the selected concept is being approved by regulatory authorities. Consultants are normally active in the concept evaluation phase, collecting lines of evidence and evaluating different concepts of remedial solutions based on these data (Sparrevik and Breedveld 2010). This often also includes preparing permit applications or environmental impact assessments.

Stakeholders, defined here as people, organizations, or groups who are affected by the issue and who have the power to make, support, or oppose the decision (Susskind et al. 1999), tend to be involved late in the process as a part of formal hearings and therefore act right before the decision is made. Advice from individuals, in their capacity as concerned residents, is often not considered in the formal decision-making process, because many individuals are not directly affected by project impacts and therefore are not included in the formal hearings.

The proposed remedial solution circulated prior to a public hearing is often designed based on technical feasibility, budget, time, and political perspective. The maneuvering space for changes at this stage of a project tends to be limited, which may cause problem owners to defend the solution instead of ensuring a constructive stakeholder dialogue (Kasperson and Kasperson 2005). This unfortunate situation may result in significant opposition that could lead to increased costs and delays in the execution phase of contaminated site remediation projects (Sparrevik et al. 2011).

Description of the MIP methodology

The MIP methodology as shown in Figure 1, uses multi-criteria decision analysis and consensus-based deliberation to structure the involvement process:

Objectives. The 1st step of the process includes formulation of the project objectives, selection of alternatives, and recruitment methods. The problem owner responsible for

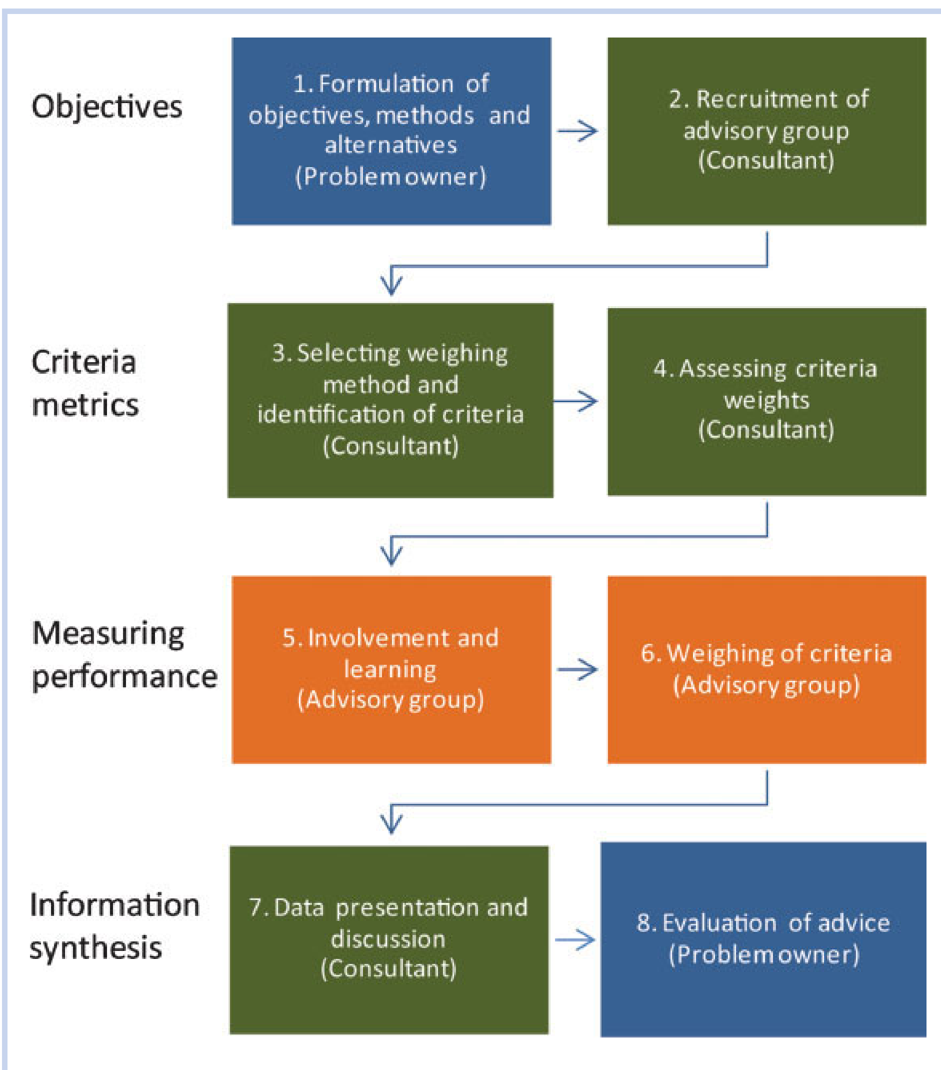

Figure 1. Schematic overview of the proposed MIP.

the decision is presumed to be highly active in this phase. The 2nd step consists of recruitment of the advisory group. Depending on the project objectives defined in the previous step, inhabitants representing public interests or stakeholders representing specific business interests may be recruited. The method used to recruit participants is important with regard to representation of different points of view and values in relation to the project impacts. The consultant is recommended to be responsible for this step.

Criteria metrics. The 3rd step consists of identification of criteria. The consultant is assumed to be heavily involved in this step, possibly in cooperation with the problem owner. In the 4th step, impacts for each criterion are assessed, based on available technical information and expert judgment. We recommend that the consultant carry out this task, because it requires detailed technical knowledge about the alternative performances for selected criteria.

Measuring performance. The intention of the 5th step is to allow the advisory group to discuss the alternatives, allowing the consultant to invite experts with specialized expertise to clarify questions. The advisory group should also evaluate and alter the earlier proposed criteria if necessary. The 6th step includes criteria weighting.

Information synthesis. In the 7th step, data are processed by the consultant, and results are presented to the advisory group. In the 8th and last step, the advice from the group is presented to the problem owner.

MIP APPLICATION TO THE BERGEN HARBOR SEDIMENT REMEDIATION STUDY

Objectives, alternatives, and methods

Bergen Harbor study objectives. The harbor area of Bergen is contaminated from previous industrial activities such as naval shipyards and manufacturing industries, earlier releases of municipal sewage, and urban runoff from diffuse sources. One of the major contributors to harbor contamination is polychlorinated biphenyls (PCBs), which originate from paint on house facades (Jartun et al. 2008). The area is 1 of 17 fjords in Norway prioritized for remedial actions by the Norwegian Government (Norwegian Ministry of Environment 2002, 2006). The area also has a dietary advisory for fish consumption based on PCB and mercury concentrations. The work with contaminated sediment management has been progressing for several years, focusing on site investigations, risk assessments, and preparations of management plans. At present, complementary archaeological investigations, as well as plans for field trial studies to assess remediation methods, are being executed.

The objectives with the MIP in this case was to provide valuable advice to the problem owner on how advisory groups perceive hypothetical remediation alternatives distilled from the recommendations laid out in management plans. However, the MIP has been developed as part of a larger research project (Oen et al. 2010). So, although the advisory groups were aware that they participated in a research setting, the MIP was conducted in close cooperation with the municipality of Bergen (problem owner) and the results have been subsequently used as input to the ongoing contaminated sediment remediation project at Bergen Harbor.

Remediation alternatives. Five sediment remediation alternatives were suggested based on discussions with the problem owner. Alternative 1 constitutes natural recovery. The sources of contamination in Bergen Harbor have significantly decreased due to reduced industrial activity, better emission control, and wastewater treatment. It is estimated that background values for contaminant fluxes to the water from the sediments will be reached within a time span of 50 years due to natural deposition of clean sediments on top of the contaminated sediments (Oen et al. 2005). Alternative 2 consists of an active reduction of the contaminant flux by capping a 1.5-km^2 area in the inner fjord basin with a 30-cm layer of clean material, as indicated in Figure 2. Capping has proven to be efficient to reduce contaminant transport from contaminated sediments (Eek et al. 2008), but because it reduces sailing depth, it may have practical limitations (Palermo 1998). Three combined alternatives were therefore suggested, consisting of capping the majority of the area and dredging areas with the highest concentrations of PCB (hotspot areas) and where sailing depth could be an issue (Figure 2). The 3 alternatives are differentiated by the method of disposal. In alternative 3, near-shore disposal facilities were assumed to be constructed with the possibility to reclaim land for property development, whereas alternative 4 and 5 consisted of land disposal at local and national waste disposal facilities, respectively. The transport distances to each disposal facility were 1 km (near-shore disposal), 12 km (local disposal), and 800 km (national disposal site).

Recruitment of advisory group

The MIP process requires involvement of advisory groups in the management process. Three advisory groups were created in this case: local residents, local stakeholders, and nonresident sediment experts as summarized in Table 1.

The local residents were randomly recruited based on census lists in order to represent the general community interest in Bergen. To ensure an adequate recruitment process, a commercial market research institute was engaged to recruit participants based on specifications from the research project, which included a fair gender and age representation and residential distribution for both the immediate vicinity of Bergen and more than 3 km from the Bergen Harbor area. None of the persons recruited were allowed to have previous experience with contaminated sediment management. In total, 20 participants agreed to participate, with 17 (85%) participating in all 3 sessions. The residents were compensated for their participation and received a gift voucher, following normal market research practices.

At the last meeting, 4 residents panel groups (5 participants in each group) were established for consensus-based deliberative evaluation. In comparison to earlier meetings, when the local residents were asked to act in their capacity as individuals, thus promoting their private household values, the residents' panels were asked to act as community representatives in a citizens' jury setting (Soma and Vatn 2010), thus promoting values representing the community needs.

The local stakeholder group was recruited to reflect specific interests in the Bergen Harbor remediation case based on a review of available documents, commentaries to the prepared management plan, media interest, and dialogue with the problem owner. In total, 103 potential parties were identified.

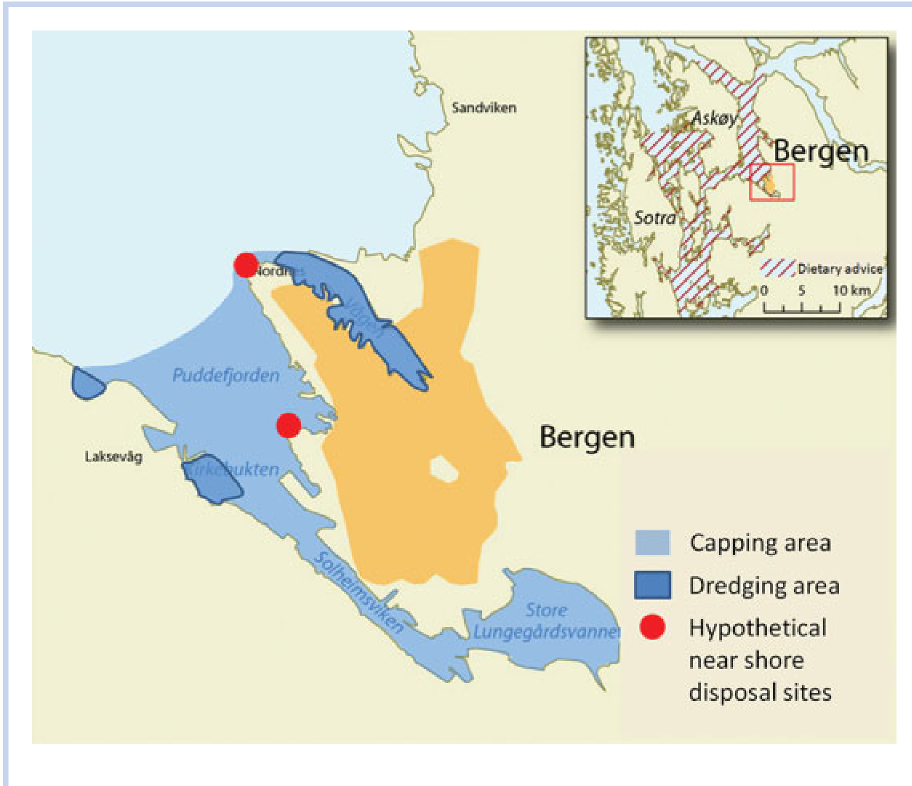

Figure 2. Overview of Bergen Harbor. Areas for capping, dredging, and near-shore disposal are indicated in the map. Modified from www.klif.no.

From this list, a set of 14 stakeholders were selected by the research team on the basis of mapping in an influence and interest grid (Chinyio and Olomolaiye 2010). The subset was selected to reflect the most influential and interested stakeholders. ''Influence'' was defined as the potential to affect the process either through formal legislative rights or by informal mobilization through media and financial instruments. ''Interest'' was defined as the potential level of benefits or losses the stakeholder could experience from the process. Similar to influence, interest was categorized into formal interests such as regulatory issues and informal interests such as gain or loss of image and popularity. Discussions with 2 of the selected stakeholders and the problem owner were conducted, and as a result from these discussions, the list of stakeholders was subsequently expanded from 14 to 23. These individuals were not only invited to participate in advisory meetings but also to potentially function as an advisory group in the forthcoming project stages at Bergen Harbor. In total, 16 individuals of the 23 agreed to participate in a stakeholder group and 11 (48%) participated in the advisory meetings. The participation was voluntary without financial compensation.

The nonresident sediment experts were scientists or consultants and regulators working with contaminated sediments on a daily basis. The experts were recruited from the Oslo, Norway, area without particular connections to Bergen.

Table 1. Descriptive statistics for participants and analysis of variance (*F*-test) between the groups[a]

	Sociodemographic aspects					Initial perception of risk	
	N	Age	Gender ratio (male/female)	University education	Residence in Bergen	Risk attitude[c]	Risk perception of sediments[d]
Group	N	(year)	(%)	(years)	(years)	(rank)	(rank)
Local residents	17	37.9 (10.9)	53	4.6 (2.8)	26.7 (16.5)	1.9 (0.7)	4.5 (2.7)
Local stakeholders	11	52.8 (7.8)	91	6.0 (2.3)	34.7 (15.9)	2.1 (0.8)	3.1 (2.6)
Nonresident experts	12	42.4 (11.3)	58	6.5 (1.7)	2.4 (5.7)	2.1 (0.5)	2.8 (1.5)
Variance between groups[b]		5.44 (0.09)		2.37 (0.11)	14.7 **(<0.01)**	0.47 (0.62)	2.08 (0.14)

[a]Mean values. Standard deviation in parentheses.
[b]*F*-test values. Significance in brackets. Bold face values indicate parameters where the *F*-test give a $\beta \neq 0$ (95% confidence).
[c]Risk attitude according to ordinal scale: 1 = Risk avoidance, 2 = Acceptance within limits, 3 = Natural and beneficial if respondent in control.
[d]Ordinal scale (1–10) where 1 indicates low risk and 10 indicates high risk.

One session included 4 researchers from 1 of the institutes participating in this study; a separate session with 8 participants (19 invited) was also conducted using web-based recruitment to specifically target consultants, researchers, and regulators working with contaminated sediments in the Oslo region. As for the stakeholders, the participation was voluntary without financial compensation.

Selecting weighting method and identification of criteria

Evaluation of different MCDA weighting methods for use in sediment management has been previously investigated, and all methods have their strengths and weaknesses (Linkov et al. 2007). The selection of appropriate weighting methods is basically a choice between accuracy of the utility or value-based methods (multi-attribute value theory, multi-attribute utility theory), user friendliness of the analytical hierarchy processes (AHP), or the simplicity of outranking methods. The use of MCDA with an advisory group early in the project process places the emphasis on finding methods that are simple and user friendly. Later stages of using MCDA for decision making should emphasize consistency and robustness as well. The AHP method (Saaty 1987) was thus selected for this study on the basis of its advantages in scoring and user friendliness. AHP completely aggregates the decision problem into a single objective function and uses a compensatory optimization approach. AHP uses a quantitative comparison method that is based on pairwise comparisons of decision criteria, rather than utility and weighting functions. However, combinations of AHP and utility-based methods are also practiced in the literature (Zahedi 1987).

The pairwise comparison may be performed on different levels in a decision tree to allow people to compare criteria in pairs in order to avoid cognitively more challenging multiple simultaneous comparisons. In this study, a hierarchical decision tree was used by organizing criteria in 3 levels to reflect the different pillars of sustainable development: environmental, societal, and economic aspects (UN 1987) (Figure 3). Under each criterion, subcriteria were added. The advisory groups were able to discuss and comment on the criteria, but only a limited number of alterations were performed in order to assure consistency between the groups.

Assessing criteria weights

Weights for each criterion were set on the basis of the environmental impact assessment as presented in the management plan for Bergen Harbor (Oen et al. 2005) as well as consultations with sediment experts. The criteria and the criteria weights are provided in Table 2 and are briefly described below.

Environmental criteria. The environmental risk was expressed as a reduction in flux of PCBs from the contaminated sediments, compared to today's baseline scenario. Calculations showed that both capping and dredging would be very effective in reducing the flux of contaminants from the sediments. The effect of dredging is slightly lower due to resedimentation of dredged material on top of the newly dredged seabed (Oen et al. 2005).

The reduction in human health was assessed on the basis of 10% exceedance of the maximum tolerable risk (Baars et al. 2001), compared to the percent exceedance calculated for the current situation. The calculation of maximum tolerable risk is mainly based on consumption of fish (15 meals per month from the contaminated area) and to some degree from direct exposure to water and sediment during bathing. Greenhouse gas emissions were calculated on the basis of vessel transport distances from shore to capping area for alternatives 2 through 5, also including the distances from the dredging area to shore and truck transport to disposal sites for alternatives 3 through 5. Emission data from Statistics Norway (www.ssb.no) was used for the calculations. In order to illustrate the magnitude of emission values to the advisory group participants, the figures were normalized against emissions from the estimated yearly emission from a private car (1530 kg CO_2-equivalents; www.naturvern.no).

Societal criteria. The impact of construction was assessed as an ordinal number proportional to the surface area impacted

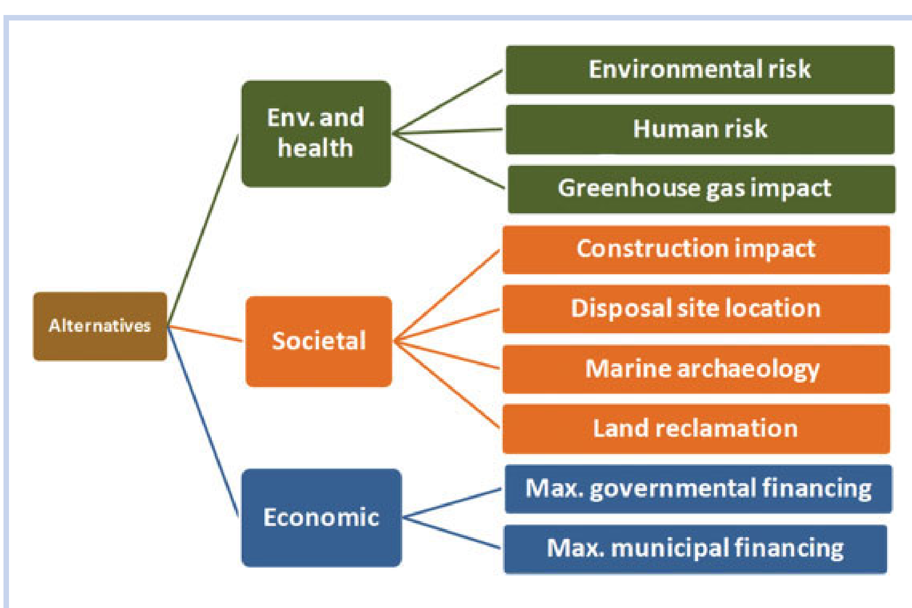

Figure 3. Decision tree showing the criteria used in the MIP for Bergen Harbor.

Table 2. Formulation of consequences and assessment of criteria weights for the alternatives in the MCDA

			1	2	3	4	5
Criteria	**Subcriteria**	**Unit**	**NR[a]**	**Cap[b]**	**Cap+NS[c]**	**Cap+LD[d]**	**Cap+ND[e]**
Environmental	Reduction of environmental risk (PCB flux from sediments)	%	0	99	98	98	98
	Reduction of human health risk	Times MTR exceedance	50	2	4	4	4
	Greenhouse gas impact	Personal car eq/y [f]	0	461	354	648	2961
Societal	Construction impact (spatial influence)	Score	0	1	2	2	2
	Disposal site location	Score	0	0	1	2	3
	Marine archaeological preservation	%	100	90	70	70	70
	Area for property development (land reclaimed)	m^2	0	0	10 980	0	0
Economical[i]	Maximize governmental/minimize municipal financing	NOK $\times$ 10^6 [g]	0	102	91	108	115
	Maximize municipal/minimize governmental financing	NOK $\times$ 10^6 [h]	0	52	55	62	64

[a]Natural recovery of area (NR)
[b]Capping with 30 cm layer (Cap)
[c]Capping and dredging with near-shore disposal (NS)
[d]Capping and dredging with local land disposal (LD)
[e]Capping and dredging with national land disposal (ND)
[f]Normalized against the amount CO_2 released from a car during 1 year
[g]Values represent total municipal cost. Values in Norwegian crowns (NOK)
[h]Values represent total governmental cost. Values in Norwegian crowns (NOK)
[i]The phrasing of questions were changed during the MCDA. In the original setting ''total cost'' was weighed against ''municipal cost.'' A conversion was made as follows: strong weighting of ''total cost'' = strong weighting on ''minimized municipal financing.'' Neutral or strong weighting of ''municipal cost'' = neutral weighting of ''minimize municipal financing.''

during remediation. It was also assumed that capping would be both faster and less disruptive to maritime activities than dredging.

The disposal site location was also addressed as an ordinal number, where a local solution was considered more favorable than using a national disposal site. This is based on the assumption that contaminated sediment remediation is best handled by local solutions (Breedveld 2007). The criterion was also used to investigate whether disposal solutions close to residential areas were disfavored (NIMBY [''not in my backyard''] effects) (Dyer and Sarin 1982).

The wharf area, Bryggen, is defined as a United Nations Educational, Scientific and Cultural Organization world heritage site (whc.unesco.org) and thus was an important aspect to be addressed in the MIP. It was assumed that all marine operations will negatively affect the preservation of marine cultural heritage, with dredging resulting in more negative impact than capping.

The possibility of land reclamation is only relevant for the dredging and near-shore alternative, where it was assumed that construction of a confined disposal site will establish land for property development. The area of reclaimed land was used as the criteria weight.

Economic criteria. Economic criteria were developed to observe how the distribution of local (municipal financing) versus national (government financing) costs were evaluated by the advisory groups. It was assumed that government financing would cover 25% of the costs for all alternatives, and the remaining costs would be shared by local enterprises and the municipality. It was further assumed that enterprises would partially finance the dredging operation, because they would benefit from port development. Initially, private household financing through municipal taxation was addressed in the MCDA. However, this criterion turned out to be problematic for the nonresident expert group and was therefore not used in the data evaluation.

Involvement and learning

Three meetings were conducted with the residents and the stakeholders, whereas nonresident experts were invited to 1 session only. The involvement and learning step began in the 1st meeting and included familiarization, general discussions about the study and contaminated sediments, and dissemination of written material including a description of the MCDA method, remediation alternatives, and how the consequence criteria weights were estimated. The nonresident expert group received this information by e-mail. The content of the documents was explained in the 2nd meeting (1st meeting for experts) and sufficient time was allowed for questions and comments. As evident from the material presented and their existing knowledge of the subject, the residents and stakeholder groups invited expert witnesses to clarify and address specific topics related to the issue. The intention was to introduce a deliberative discussion valuable for both the advisory groups and the expert witnesses (Renn 2006) using an approach based on citizens' jury methodology (Soma and Vatn 2010).

Weighting of criteria

Participants were asked to score the consequence criteria weights using questionnaires and were asked to weight the criteria and respective subcriteria in pairs. The scoring was performed in the meeting for the resident and stakeholder groups, whereas the experts were asked to perform a preliminary scoring via e-mail prior to the meeting. Based on earlier experience (Soma 2010), the original 9-value scale (Saaty 1987) was replaced with a less comprehensive scale. We used in total 3 values guided by the text ''strong weight'' to emphasize high relative importance of the criteria and ''neutral weight'' to emphasize equal weighting. For participants not answering the question or marking all alternatives, the neutral score was used in the data presentation step. In this study, participants were also directly asked to perform an intuitive ranking of the alternatives.

Data presentation and discussion

The software DEFINITE (Janssen and Herwijnen 2007) was used to process the data. This software uses AHP for elicitation of weights, and multiattribute utility theory for normalization of criteria and calculation of impact scores. In this case, a linear normalization was used, assuming a linear relationship between criteria and utility values.

For the residents, the results of the MCDA based on individual weighting of criteria were presented and discussed in the group as a whole, before they were divided into residents' panels. Results of weighting based on individual versus residents' panels were then compared and discussed. The stakeholders performed their scoring at the last meeting, and therefore the results were not presented to the group, due to lack of time. The experts were presented preliminary results from the scoring performed prior to the meeting, but they were allowed to change the scoring based on information given at the meeting. In all cases, the final weighting results were used for ranking the results presented in this study.

Evaluation of advice

Representatives from the residents' panels were invited to present their recommendations to the problem owner in a separate meeting. During the stakeholder meetings, the problem owner was actually present. This local stakeholder group is continuing to follow the process in Bergen with regular meetings, allowing them to contribute to ongoing discussions about sediment remediation in the harbor.

RESULTS AND DISCUSSION

Results from the MCDA

Table 1 shows the descriptive statistics for the different groups, where there is a significant variance between the groups only with respect to years of residence in Bergen. However, this was intentional, because this group was recruited from persons not living in the Bergen area. The stakeholder group had mostly male participants, whereas the households and residents' panel was recruited to achieve a balanced male and female representation. It is especially interesting to observe that the differences in initial risk-perceptive values relating to contaminated sediments are not significantly different between the groups. Sharing the same initial beliefs about the subject may facilitate unbiased advice, because experiential beliefs may also influence the analytical outcome of a decision (Slovic et al. 2004).

Table 3 shows the results from the scoring of sediment remediation alternatives. The results have first been calculated individually for each participant and subsequently integrated into mean values for each advisory group. A grand mean has also been calculated that summarizes results from all groups. In addition, the results from a hypothetical scenario with an equal score on all weights are presented to illustrate the influence of weighting on the results. Table 3 suggests that the MCDA results favor alternatives 2 and 3. A t test shows that the difference among alternative 1 ($t=-17.3$; $df=3$; $p<0.05$), 4 ($t=-5.5$; $df=3$; $p<0.05$), and 5 ($t=-5.5$; $df=3$; $p<0.05$) against the grand mean of alternative 3 is significant. The difference between alternative 2 and 3 is nonsignificant ($t=-0.5$; $df=3$; $p=0.63$). It is also clear that the advisory group weighting significantly affects the results compared to a hypothetical scenario with equal scores, which results in a preference of natural recovery.

Additional information may be extracted from the MCDA by analyzing how participants weight the criteria, using centered weight analysis (Tervonen et al. 2009). This method normalizes each criterion against a scenario where all criteria are given equal weight. A positive value indicates that

Table 3. Scoring of alternatives based on mean values of quantitative scores for the different groups and a grand mean for all groups [a]

	1	2	3	4	5
Group	NR	Cap	Cap+NS	Cap+LD	Cap+ND
Individual residents	0.44 (0.19)	0.69 (0.19)	**0.71** (0.15)	0.60 (0.18)	0.50 (0.17)
Residents panel	0.38 (0.11)	0.79 (0.04)	**0.80** (0.03)	0.70 (0.05)	0.65 (0.06)
Local stakeholders	0.44 (0.17)	**0.78** (0.07)	0.77 (0.06)	0.65 (0.11)	0.57 (0.12)
Nonresident experts	0.47 (0.14)	0.73 (0.11)	**0.76** (0.09)	0.63 (0.13)	0.48 (0.14)
Grand mean	0.43	0.75	**0.76**	0.64	0.55
Equal scoring	**0.67**	0.61	0.61	0.42	0.29

[a]In addition, a simulated case with ''equal score'' for all the subcriteria weights is provided. Bold values indicate most preferred alternative for each advisory group. Standard deviation is presented in parentheses.

participants weight this criterion higher than average, and a negative value indicates that the criterion is weighted less than average. The averaged criteria weights are higher than normal for the reduction in human and environmental risk (Figure 4). This observation could explain the low score of a natural recovery scenario, because this alternative has lower weights for the reduction in human and environmental risk than the other remediation alternatives.

In addition to mean values, standard deviation is also presented in Figure 4. An analysis of variance concludes that there exists a significant difference between advisory groups for 2 of the scored criteria: construction impacts and marine archaeological preservation. This indicates that for these criteria, the differences between groups are significantly larger than the variance within the group. Stakeholders are significantly more concerned by construction impacts ($F=6.0$; $df=3$; $p<0.05$) than are the other groups. This is natural because this group includes representatives from organizations with close ties to Bergen Harbor such as harbor authorities, boat owner associations, and the like. For marine archaeological preservation ($F=3.9$; $df=3$; $p \leq 0.05$), stakeholders are again significantly more occupied with the subject than the nonresident sediment experts. It is also interesting to observe that when the residents respond as individuals, they are less occupied with marine archaeological preservation than when they act on behalf of the community in a residents' panel. This finding is consistent with social science theory and the beliefs in differences between individual normative and social normative values. Soma and Vatn (2010) also observed this phenomenon where a deliberative citizens' jury panel setting favored mobilization of social values rather than individual values in decision-making.

A histogram of the inconsistency scores for the weighting of criteria is presented in Figure 5. Because the AHP weighting is based on pairwise comparisons between relevant criteria, the process may give illogical results, which is expressed by the inconsistency score. A completely consistent weighting of a criterion will therefore result in an inconsistency score of 0. Normally, a value below 0.1 is considered to be a sufficiently consistent scoring (Saaty and Vargas 1984). Figure 5 shows that 43 of 120 frequencies of inconsistency scores are below 0.1, with a mean value of 0.22. However, higher values and outliers are observed, indicating inconsistent scoring for some participants.

In general, statistical analysis of MCDA weighting is expected to show high variances, because the method is designed for decision makers representing broader interests. Large standard deviation in some cases may also be explained by participants giving inconsistent weights, as documented by some of the high inconsistency scores.

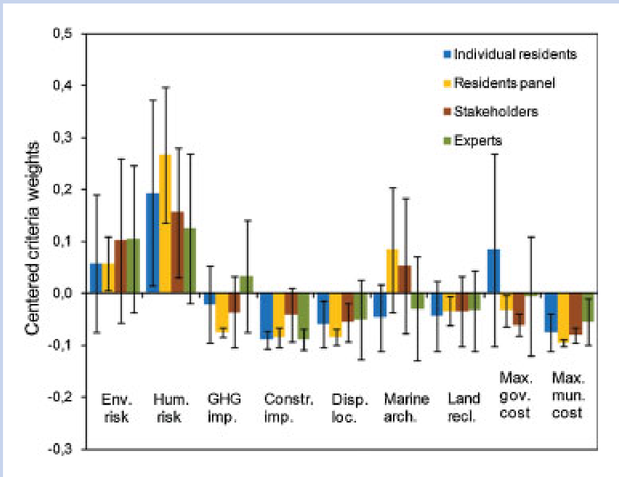

Figure 4. Centered weight analysis for the different advisory groups. A positive value indicates that the criterion is scored higher than average; a negative value indicates that the criterion is scored lower than average. The error bars represent 1 standard deviation. Criteria at bottom are (from left to right) environmental risk, human risk, greenhouse gas impacts, construction impacts, disposal location, marine archaeological preservation, land reclamation, maximum government financing, and maximum municipal financing.

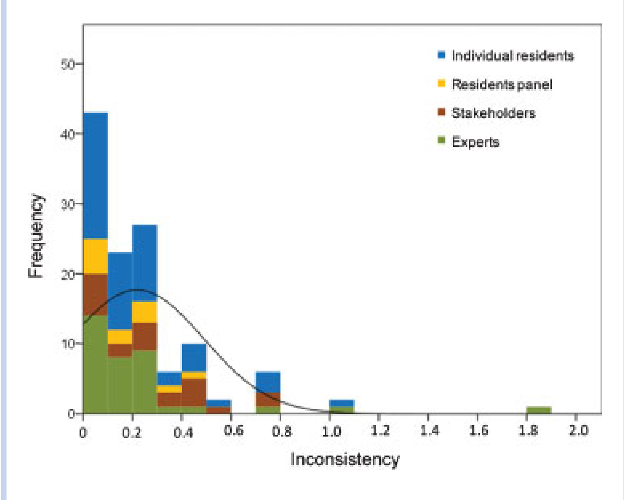

Figure 5. Histogram of inconsistency scores for the different advisory groups. Scores for choice between environmental, societal, and economic criteria as well as scores within the environmental and societal criteria are given in the figure. The economic criterion has only 1 pair and therefore has no inconsistency.

Comparing MCDA to intuitive ranking

Table 4 summarizes the number of participants who indicate a specific remediation alternative as their preferred alternative either through MCDA or via intuitive ranking. The results given in Table 4 indicate that both methods suggest that alternative 2 and 3 are the most preferred. It is also clear that although the intuitive ranking shows that some of the participants also select other alternatives as their preferential choice, the MCDA clearly deselects these other alternatives.

Figure 6 illustrates the standard deviation between intuitive ranking and MCDA. This comparison provides valuable information regarding the robustness of the process. The results show that, in most cases, the standard deviation is lower using MCDA compared to using intuitive ranking.

Use of results for management advice

Although the MCDA and the intuitive ranking result in the same most-preferred remediation alternatives for all groups, the MCDA is better equipped than intuitive ranking to sort out the ''worst alternatives.'' This potential use of MCDA has also been documented in earlier studies (Linkov et al. 2007). In addition, the use of MCDA in this study also results in

Table 4. Most preferred alternative based on direct ranking and results derived from the MCDA[a]

		1	2	3	4	5	
Group	**Method**	**NR**	**Cap**	**Cap+NS**	**Cap+LD**	**Cap+ND**	**Total number of participants**
Individual residents	Direct	0	**10,5**[c]	3,5[c]	2	1	17
	MCDA	2	**7,5**[c]	**7,5**[c]	0	0	
Residents panel	Direct	0	1	**3**	0	0	4
	MCDA	0	**2**	**2**	0	0	
Stakeholders	Direct	2	**2,5**[c]	1,5[c]	1	0	7[b]
	MCDA	0	**4**	3	0	0	
Experts	Direct	0	4	**7**	1	0	12
	MCDA	1	2	**9**	0	0	

[a]Bold values indicate alternative preferred by most participants in the group.
[b]Only 7 stakeholders participated in this exercise whereas 11 stakeholders participated in the MCDA exercise.
[c]In cases where 2 alternatives were ranked first, the responses were divided between the alternatives. If the participant did not use the full scale (1–5) to rank the alternatives, the scale has been normalized to an ordinal 5 value scale.

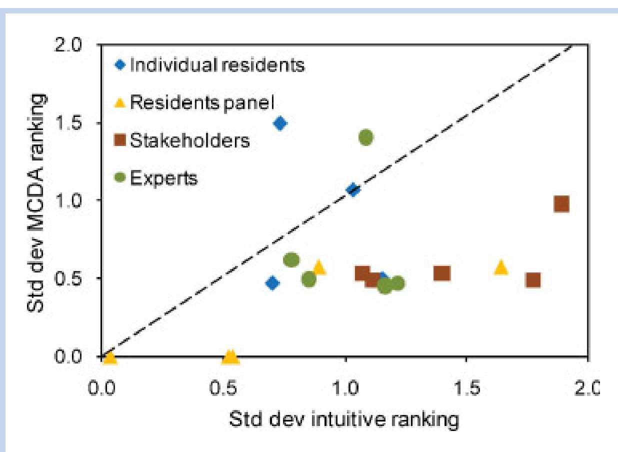

Figure 6. Analysis of the standard deviation (std dev) within the groups for the different alternatives for both intuitive ranking and MCDA.

lower standard deviation compared to intuitive ranking, which also is in line with findings in earlier studies (Linkov et al. 2009). For the residents' panel, the lower variance may also be a result of the deliberative consensus-based weighting in the residents' panels, relative to individual-based non-deliberative weighting.

The centered weight analysis of the scoring responses indicates a strong focus on human and environmental risk reduction. The standard deviation presented in Figure 4 is in some cases relatively high, which usually is problematic for interpretation of results. In this case, it may however be a valuable piece of information, because it may indicate potential disagreement and therefore should be specifically addressed in the management process.

MIP methodology evaluations

The role of MCDA in the process. One may argue that some of the data obtained through the structured involvement process with MCDA may be obtained by other less resource-intensive survey-based methods. Willingness-to-pay studies have previously been used to map preferences and to map household and recreational users willingness to pay for sediment remediation in Norway (Barton 2009). The advantage of such a large sample survey-based approach is its ability to quantitatively assess public opinion about a project. Stated preference survey-based valuation methods are also the only valuation methods that address so-called nonuse and existence values. However, the use of contingent valuation methods has been questioned due to possible differences in the willingness-to-pay contingent in hypothetical project alternatives, versus the actual willingness to pay once actual alternatives are on the table. In Norway, low sample response rates have also been shown to affect representativeness of the affected population (Barton et al. 2009). Choice experiment surveys, another stated preference survey-based method, have been proposed as an alternative with cost-saving, small-sample, and preference elicitation advantages over contingent valuation (Bateman et al. 2002). Choice experiment surveys can be used as a formal approach to "map" stakeholders' individual preferences for remediation alternatives. However, a condition for using survey-based valuation methods is that the choice of sediment remediation alternatives must be described in terms of their component attributes in a survey setting. The experiences from our study question this possibility, because a highly interactive process seemed to be necessary for participants to be able to understand the relationship between alternatives and their impact criteria, and then to subsequently weight alternatives. However, survey-based studies may have a role in confirming preferences for a small set of specific project design criteria; for example, once MCDA has narrowed alternatives and identified contentious criteria, willingness to accept reduced accessibility to certain beach locations during a remediation period could be assessed. In order for stakeholders or residents to be able to embrace a complex decision such as the selection of remediation alternatives, an involvement process with lateral learning, combined with MCDA to provide structure, robustness, and transparent documentation, is preferable and certainly an advantageous step before conducting stated preference surveys to evaluate the representativeness of stakeholder preferences in the population.

Use of AHP as weighting method. One of the main advantages of using a compensatory weighting approach such as AHP is the pairwise weighting. This builds on the assumption that decision makers are more relaxed with making relative comparisons between the objectives than to score in absolute values (Baron 1997). This assumption is to a large degree supported by this study. Even though, especially, local stakeholders and nonresident experts criticized the choice of criteria and related weights, the majority of the participants were able to perform the weighting with logical results. This indicates AHP is a suitable method for performing MCDA in advisory settings.

It is, however, important to note that use of compensatory methods also has disadvantages compared to outranking methods. The standardization of the utility function is an obvious challenge. In this study, as in many other studies, the standardization of the criteria weights was assumed to be linear for all criteria. However, according to prospect theory, this is incorrect (Kahneman and Tversky 1979). This theory argues that the value curve is asymmetrical to the reference point, i.e., people generally put more emphasis on ''losses'' than on ''gains.'' Thus, the asymmetrical value curve is steepest at the reference point, subsequently overemphasizing small losses compared to larger losses. In our case, this means that criteria involving negative aspects, such as human and environmental risk, should be standardized differently to gains, such as land reclamation (Figure 4). This standardization requires a reweighting of impact scores. In our methodology, the impact assessment scoring document was explained to the groups, and despite substantial simplification, 2 rounds of expert witnesses were required to clarify its complexities for the residents' panel. In our opinion, further reweighting of impacts based on assumptions about an individual's aversion to risk would probably have confused rather than clarified the panel's understanding of the environmental impact assessment information used in the MCDA. We therefore elected to disregard more advanced impact scoring methods.

Another challenge with the AHP weighting approach is its hierarchical structure, because uneven distribution of subcriteria between the main criteria will ''dilute'' the importance of the subcriteria in groups having a greater number of criteria. In outranking methods, these issues are not present. Therefore, outranking methods may be the preferable choice for decision making where simplicity and robustness is favored over user friendliness in criteria weighting. Use of outranking also allows application of stochastic multicriteria acceptability analysis to the results. In stochastic multicriteria acceptability analysis, criteria weights may be entered as distributions, and a probabilistic approach is used to determine the most preferred alternative (Alvarez-Guerra et al. 2010). The focus of this study was to initiate a structured involvement process facilitated with MCDA. The aspect of uncertainty evaluation was therefore not highly prioritized. Criteria were selected to reflect the interest of residents, stakeholders, and experts, rather than to be a comprehensive baseline for a decision because the project is in an early exploratory phase. It is evident that as the project process advances, the selection of MCDA methodology should evolve, possibly focusing more on uncertainties in the criteria, thus requiring the use of other weighting methods.

Evaluating advisory group perception of the MIP. Within each session of the advisory group meetings, the participants were asked how they perceived the session and the involvement process. The general impression from both the discussions and the results of the questionnaires was that people agreed that the involvement and learning process was positive in terms of information exchange between both expert witnesses and the advisory groups. These results are encouraging for the future application of MIP because it indicates a successful exchange of information, and that both residents and stakeholders can produce valuable advice for the management process, with results well in line with what experts suggest.

There are, however, differences between the dynamics in the groups. The residents participated, using the available information and methods within the timeframe of the 3 meetings. The nonresidential experts performed the MIP within the first meeting, reflecting their familiarization with the subject; however, they were occupied with the assumptions made when assessing criteria and criteria weights. The work flow with the stakeholder group was different. Although the meetings were constructive, more time was spent on familiarization, clarification of the mandate for the group, and questions relating to the MCDA process. The stakeholders also to some degree questioned the objectives of the study and focused on their role in the forthcoming project process. These discussions about roles and expectations are common in stakeholder involvement processes (Gerrits and Edelenbos 2004) and emphasize the need to invest time in familiarization and formulation of objectives when stakeholders are included in involvement processes (Sjöberg and Drottz-Sjöberg 2008). It is also important to consider these differences when deciding what kind of advisory groups to engage in the MIP.

CONCLUSION

In this article, we propose use of MIP to enhance participatory involvement processes and to provide valuable advice for a problem owner in a contaminated sediment remediation process. The results from the MIP have also subsequently been used at Bergen Harbor for practical remedial considerations and for addressing risk communication in the project.

The evaluation of the Bergen Harbor case study supports the feasibility of this method for these processes. This statement is supported mainly by 2 findings in the study. First, the results show that using MCDA as an integral step in the MIP adds structure and robustness to the involvement process and provides good documentation of criteria to be further addressed by the problem owner. Second, we perceive involvement and learning as important for the participants in order to be able to perform the MCDA in the selection of remediation alternatives.

However, challenges arise when MCDA is used in an advisory process. First, considerations must be made regarding the use of the MCDA method. In this case, the AHP was selected due to user friendliness, but other settings may require the use of other methods that have been shown in the literature to produce more robust results. Second, this study showed that the quantitative scoring was perceived as problematic and was questioned especially by stakeholders and experts. The interactions and the qualitative information gained from the different advisory group discussions, as suggested in the MIP, are therefore important in order to reduce misunderstandings and misinterpretations.

Finally, it is important to remember that the emphasis on method and process should be balanced with both quantitative and qualitative methods, as proposed in the MIP. By including MCDA in the MIP, the structure and documentation of the process is ensured, thus providing quantifiable results that can be replicated by third parties. By engaging group interaction and learning through participatory methods, the quality of the involvement process from recruitment to final discussions is preserved, thus setting the stage for successful sediment remediation projects for both stakeholders and problem owners.

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Towards Sustainable Decisions in Contaminated Sediment Management by use of Stochastic Multi-criteria Analysis

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ABSTRACT: Due to the complexity inherent in contaminated sediment management problems, decisions methods supporting the sustainable management of contaminated sediments will require use of multiple methodologies to be conclusive. This may be achieved by combining different analytical approaches like risk analysis (RA), life cycle analysis (LCA), multi-criteria analysis (MCA) and economic valuation methods. We propose use of stochastic multi-criteria analysis (SMCA) based on outranking algorithms to implement this integrative strategy. This method allows explicit handling of uncertainties and is less sensitive to correlations than MCA using internal normalization. In the paper we use SMCA to select the preferable magnitude of clean-material sediment capping for the dibenzo-*p*-dioxins and -furans (PCDD/Fs) contaminated Grenland fjord in Norway. In the analysis, the positive utility of health risk reductions and socio-economic benefits from removing seafood health advisories is evaluated against the negative utility of remedial costs and life cycle environmental impacts. A value-plural based weighing of criteria is compared to criteria weights mimicking traditional cost-effectiveness (CE) and cost-benefit (CB) analyses. Capping of highly contaminated areas in the inner or outer fjord is identified as the most preferable alternative under all criteria schemes and the results are confirmed by an information-based sensitivity analysis. In general, this methodology may serve as a flexible framework for future decision support and can be a step towards more sustainable decision making for contaminated sediment management and other related fields.

INTRODUCTION

Management of contamination in urban coastal zones is often characterized by large geographical affected areas and complexity in the causes and relationships in contaminant pathways between sources and receptors like biota, marine life and humans (*1*). One way to address this complexity in decision making is to use human and ecological risk assessments (HERA) elucidating causal mechanisms and predicting adverse health and ecological end points. By use of multiple lines of evidence related to chemical properties, toxicity and alterations at the receptor level, the total risk is assessed and threshold values are established for acceptable concentrations in sediments and water (*2*). However, basing management solely on the use of precautionary health and ecological risk assessments may be insufficient, since the costs and wider societal benefits of outcomes are overlooked. This single-criterion focus may promote extensive remediation strategies potentially having significant costs and negative environmental impacts from the remediation measures themselves (*3*). Several additional methods can complement HERA in data acquisition to promote a more holistic perspective (Table 1).

Table 1 Comparison of different data-aggregation and evaluation methods relevant for complex environmental decisions

Decision phase	Method	Strength	Weaknesses
Aggregation	Human and ecological risk assessments (HERA)	Quantitative estimates of absolute risk	Resources used to reduce risk are not addressed
	Life cycle assessments (LCA)	Holistic perspective for impact assessment	Global focus with insufficient resolution for site assessment
	Contingent valuation (CV)	Monetization of non-market benefits of environmental quality	Stated hypothetical responses regarding stakeholder preferences
Evaluation	Cost effectiveness analysis (CEA)	Avoids monetization of controversial topics	Only single criteria may be evaluated
	Cost benefit analysis (CBA)	Quantitative well known decision method	Requires monetization
	Multi-criteria analysis (MCA)	Aggregation of multiple data without monetization	Subjective weighing of criteria is necessary

Use of life cycle assessment (LCA) can supplement HERA to create an enhanced, systems approach to sediment management. In LCA, the inputs, outputs and the potential environmental impacts of a system are compiled and evaluated throughout the whole life cycle of the remediation project. In contrast to HERA, which addresses risks associated with a specific chemical or stressor, LCA aggregates impacts in space and time to find the total environmental impact—including aspects originating from the contaminated sediments and from the remedial process.

A truly sustainable perspective will need to address environmental, societal and economic issues within a single framework. Monetization of social and environmental impacts is required if they are to be compared directly with direct cost for different remedial strategies. Monetization is already present in existing life cycle impact methods (LCIA), but only as a common denominator for comparing resource depletion (*4*). The perceived benefits of remediation not readily reflected in market prices are scarcely addressed in current approaches to LCA (*5*). In the case of sediment remediation, such non-market benefits may include the enjoyment of recreational fishing, avoiding adverse health effects, and preserving the fjord ecosystem for future generations. For the purpose of this paper 'socio-economic benefits' refer to this mix of perceived direct use, existence and bequest values. Survey-based contingent valuation (CV) of such non-market benefits has been used in environmental economics for many years (*6*), but only recently applied to contaminated sediments (*7*).

While the above methods are valuable for data aggregation in contaminated sediment management, prioritization is often conducted based on comparing a monetary valuation of benefits against remediation cost. One traditional way of performing this evaluation is through cost-benefit analysis (CBA), where alternatives are ranking according to the net additive value of their benefits and costs. In cases where costs may need be compared against a single not-easily monetized impact, cost effectiveness analysis (CEA) can be used as an alternative to CBA (*8*). Traditionally, CEA has also been used in Norwegian contaminated sediment management by evaluating remediation costs against the avoided health risk from reduced exposure to sediment contamination (*9*).

Regardless of the metrics selected, both CEA and CBA have been criticized for their use in environmental projects because their monistic approach simplifies utility to a single value dimension (*10,11*). Multi-criteria analysis (MCA), in which both quantitative and qualitative criteria may be combined without the need to reduce parameters to a single unit, has been proposed as a method to overcome these traditional problems of CBA and CEA in public decision making (*12*). MCA, however, has its own criticisms. MCA recognizes that choices and decisions involve multiple subjective values and therefore require extensive use of deliberative methods and expertise in cognitive science to determine the values of decision-makers (*13*). CBA and CEA often rely on a combination of large sample sizes and simulation-based probabilistic scientific models which it is claimed provide more representative, replicable and less subjective values (*14*). While experts will continue to debate the pros and cons of different decision-support tools, decision-makers may be more willing to accept policy advice that is consistent using different approaches and available information.

In this paper we advocate the use of an MCA framework to prioritize the selection of contaminated sediment remedial strategies. As indicated in Table 1, each of the identified methods for data aggregation and evaluation has its own strengths and weaknesses, implying that a mixed-method approach is required for a conclusive interpretation of results. Though MCA is intrinsically value plural, it can also mimic results from CEA and CBA by manipulation of weights, without any additional monetization, thus allowing the decision maker to produce and evaluate results in a flexible manner to meet the requirements of the any relevant regulatory or political needs. By implementing MCA through use of stochastic outranking algorithms (SMCA), the effect of data uncertainties on the decision can also be evaluated (*15,16*).

SMCA allows the probabilistic simulation of uncertainty across a large number of criteria and has been previously demonstrated for contaminated sediment management (*17*). We develop the method further by expanding the sensitivity analysis to evaluate both the effect of existing uncertainties on the decision and the potential value of reducing uncertainty through the collection of additional information (the value of information). This 'information-based' approach to stochastic analysis in MCA addresses the challenges of quantifying uncertainty across multiple criteria based on varying mixes of observation, simulation and expert judgment. This type of sensitivity analysis has also been recommended for traditional CBA (*18*).

To show the applicability of this method, we illustrated its use for a complex sediment management situation in the dibenzo-*p*-dioxins and -furans (PCDD/Fs) contaminated Grenland fjord in Norway. This use of SMCA illustrates what can be expected in realistic sediment remediation cases, given current knowledge of impacts. Based on the results, the feasibility of bringing together different decision methods through a multi-criteria framework for sustainable decision making in similar cases is discussed.

▪ METHODS AND CASE STUDY

The case study discussed in this paper exemplifies the value of using SMCA as a meta-model for evaluating decision problems in a holistic and transparent manner without extensive monetization of impacts or definition of causal relationships (*19*). This method is generally applicable to other cases where decisions need to balance societal, environmental and economical aspects of a problem, for instance, based on uncertain information.

Stochastic multi-criteria analysis using outranking algorithms. Multi-criteria analysis is a well known decision method for the management of contaminated sites (*20*-*22*). Methodologically, MCA can be separated in two principally different sets of methods: multi attribute utility methods (i.e., MAUT), where criteria values are transformed into normalized utility functions, and outranking methods, which comparatively assess and order alternatives without normalization (*23*). This study uses the PROMETHEE II outranking algorithm to order the alternatives (*24*). This method determines a directional preference vector for one alternative over all others without resorting to preference functions or partial preordering. Preference in PROMETHEE II is assessed simply by assuming that complete preference (i.e., $c=1$, where c represents preference) is reached if one alternative (e.g., a) performs better than the other alternative (e.g., b) in a pair wise comparison on a single criterion (i.e., $a>b$), and not otherwise (*else* $c=0$).

The aggregated positive directional vector (c) is the linear-weighted sum of all pair wise comparisons over k number of criteria, where w_k represents the relative weight assigned to each criterion.

$$c = \sum_k w_k \times c_k$$

The negative directional vector is also calculated based on the reverse assumption (*if* $b>a$ *then* $c=1$, *else* $c=0$). The net weighted preference vector, or net flow, is then computed as $\sum_{b\neq a} [c(a\geq b) - c(b\geq a)]$. This procedure is repeated for all alternatives, producing a preferential rank order from highest to lowest net flow. In SMCA the PROMETHEE II algorithm is used to stochastically simulate distributions of rank orderings through Monte Carlo simulations using probability distributions instead of discrete values as input data. A total of 10,000 simulations were used in this study, which yielded sufficient stability in results for further analysis.

Information based sensitivity analysis. Since the scores of each alternative on the various criteria vary with the given input distributions, the results represent the probability of each alternative having a given rank order (rather than the absolute preference indicated without uncertainty). This uncertainty in the input scores makes it especially important to assess the robustness of results to potential changes in the state of knowledge; we do this through a two-step sensitivity analysis assessing the value of the information.

The first step consists of probabilistic dominance analysis; if the net flow of one alternative is higher than that of all other alternatives across the entire distribution of results, this alternative exhibits first order dominance. This means that the probability of ranking the alternative first is always highest and that it should be preferred by all decision makers, whether risk adverse or risk seeking. If first order dominance is not present for any alternative, the alternative having highest area under the curve of the cumulative distribution of the net flows exhibits a second order dominance over other alternatives. Second order dominance is, however, much weaker, and may call for more thorough evaluations to be conclusive for a risk-adverse decision-maker (*25*). This can be accomplished through the introduction of new criteria or through further data collection to reduce uncertainty in the input parameters.

Barring alterations to the decision criteria, the second step in the sensitivity analysis is to estimate the benefit of obtaining improved information on the input values. In this paper we use the algorithms described by Brennan et al. (*26*) to measure the partial expected value of perfect information (partial EVPI). Brennan et al. define this value of additional information as the difference between expected payoff, in this case increased net flow, that would be achieved under posterior knowledge and the expected payoff under current knowledge. 'Partial-perfect' information assumes absolute certainty on one or more input values without addressing the uncertainty of the others, while complete perfect information assumes simultaneous absolute certainty of all inputs.

Exploration of the case study. The entire Grenland fjord spans over an area of 58 km^2 and is divided into inner and outer fjord systems. Due to previous industrial activity, a substantial portion of this area has elevated levels of PCDD/Fs. There are at present state established public health advisories against consumption of fish and shellfish (dietary advisories) from the area. The shallow Brevik sill separates the systems, limiting the contamination transport between the inner and outer fjord, and allows remediation to be considered separately for the two parts. A more extensive description of the case history is given in earlier publications (*27*).

There are several criteria through which proposed remediation operations can be compared. A public health objective suggests active remediation by use of sediment capping to reduce contaminant flux to levels no longer necessitating the fish and shellfish bans (*28*). Without active remediation, the flux of contaminants will still eventually be reduced due to natural re-sedimentation of clean material on top of the contaminated sediments, but it will take significant longer time. Removing dietary restrictions also has socio-economic benefits which argue for active remediation, rather than slower natural recovery (*7*).

Health and ecological risk modeling suggest that the capping has to cover large portions of the fjord to be effective (*27*). Extensive capping, however, may be inferior to natural recovery when environmental impacts are assessed from a life cycle perspective (*3*). The magnitude of the remediation and large capping operations needed to reduce contaminant fluxes and remove dietary restrictions also imply high remedial cost.

These inherent conflicts and complexities call for a structured decision analysis to explore the space of potential optimal tradeoffs and decisions. Acknowledging that the potential range of capping alternatives is large, we here investigate six distinct scenarios based on differing magnitudes of the capping operation (Figure 1).

The first scenario is one of natural recovery (NR), in which natural re-sedimentation with clean material from the watershed will over time reduce the PCDD/F fluxes to background concentrations. The next two scenarios describe situations where only the highest contaminated areas in the inner fjord (HIFC) or outer fjord (HOFC) are capped with a layer of 5 cm of locally dredged clay. As a comparison, the effect of individually capping the entire contaminated area of the inner fjord (IFC), outer fjord (OFC), or whole fjord simultaneously (WFC) are also analyzed.

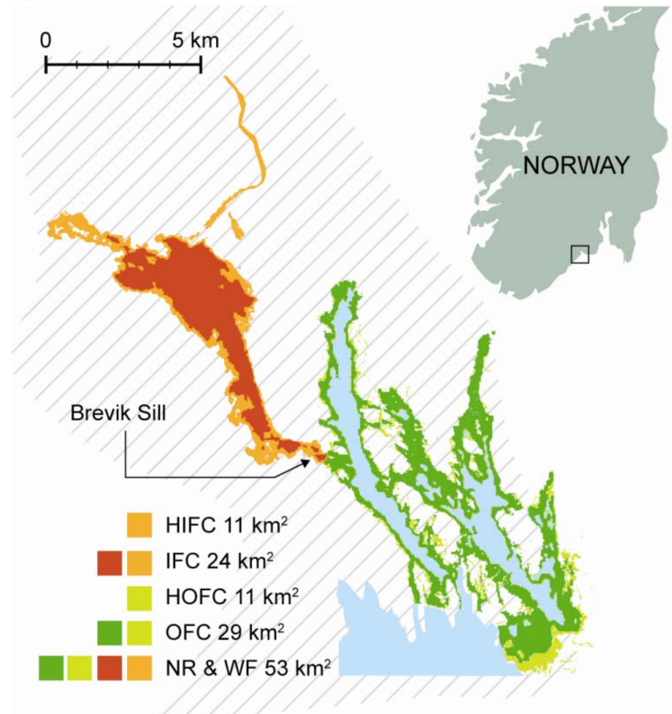
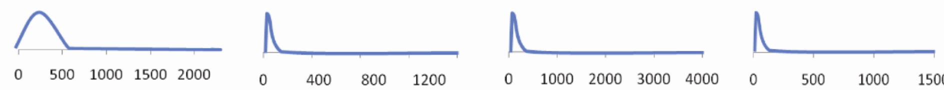
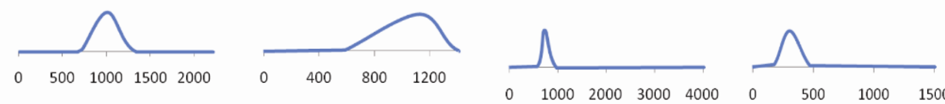
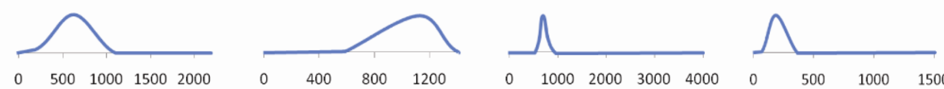
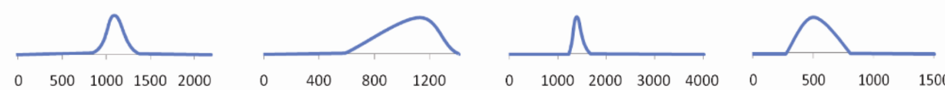
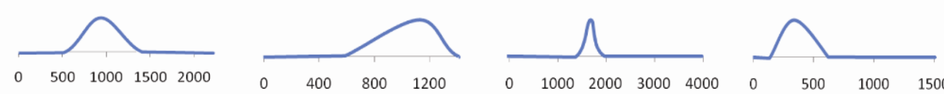
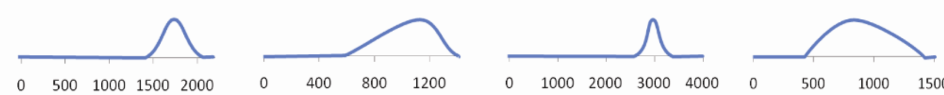


Figure 1. Overview of the six selected remedial alternatives for the Grenland fjord and subsequent estimated capping areas. Inner and outer fjord systems are separated by the Brevik sill. Adapted from (*27*).

Utility curves are defined for four specific criteria relevant sustainable decisions incorporating social, environmental and economic objectives (*29*) (Table 2). *Positive utility* is defined for both health risk reductions and socio-economic benefits of the remediation operations, based on the results of a previous contingent valuation study. *Negative utility* is defined for monetary remediation cost and life cycle environmental impacts. Table 2 shows estimated values for these utilities based on median and 5% and 95% percentile values. To account for non-normality in the input distributions, numeric approximations of skewed normal distributions has been used for the simulations (*30*).

Table 2 Social, environmental and economical criteria used to preferentially asses capping alternatives for the PCDD/F contaminated Grenland fjord [1].

	Positive utility		Negative utility	
Alternative	Health risk reduction ($y \times km^2$)	Socio econ. benefit ($NOK \cdot 10^6$)	Env. impact (ecopoints)	Cost ($NOK \cdot 10^6$)
NR				
HIFC				
HOFC				
IFC				
OFC				
WFC				

[1] In order to illustrate possible uncertainty in fixing a baseline against which alternatives are compared we also use a skewed normal distribution in the NR alternative giving a stochastic low, but non-zero mean for socio economic benefit, environmental impact and cost.

Health risk reduction is here defined as the expected number of years (y) before dietary restrictions can be removed. This is based on predictions of when values of PCDD/Fs in cod and crab decrease below levels considered “insignificantly polluted” (i.e., below 15 ng kg^{-1} wet weight for cod liver and crab hepatopancreas) (*27*). It is assumed that the entire affected area (a) will be released for unrestricted use as soon as threshold values for PCDD/F in fish are reached. For this analysis, an area-dependent health-risk-reduction index (R) is constructed to address the positive risk reducing effect for each alternative (i) in relation to the alternative with maximum risk exposure.

$$R_{(i)} = (y \times a)_{max} - (y \times a)_i$$

The socio-economic benefit of alternatives is estimated from the contingent valuation method estimating local people’s willingness to invest in remedial measures to remove the dietary advisory. The willingness to pay (*WTP*) was estimated based on a survey with 267 completed answers from households (*7*). These were segmented based on the vicinity of the fjord system, as the study found a decrease in *WTP* with distance from the contaminated area. The total *WTP* for all households was calculated based on the annual *WTP* multiplied by the number of households in the municipality (j), and summed across all municipalities. Discounted present values are estimated for a 10-year period.

$$WTP_{acc} = \sum_j WTP_j \times n_j$$

The study was not able to differentiate *WTP* based on remedial magnitude and the same *WTP* is used in this study for all active capping-remediation scenarios. Since the *WTP* decreases with distance, the lower end of the *WTP* distribution is based on values aggregated only from neighboring municipalities, while the upper band includes all municipalities, explaining the skewed distributions.

Environmental impact has been assessed based on the environmental footprint of different capping alternatives, from a life cycle perspective (*3*). This involved a life cycle analysis (LCA) of all energy and resources necessary for each remedial alternative, including the beneficial effect of reducing the flux of PCDD/Fs. In contract to health risk reductions, which focus on the benefits of reducing PCDFFs from a local perspective, LCA addresses impacts to the entire value chain from a holistic perspective, including effects relating to the remediation operation itself.

Cost is estimated from previous studies and includes dredging and disposal of hot-spot areas combined with capping for the inner fjord and capping only for the outer fjord (*31*).

The cost is estimated based unit-area costs of remediation. Uncertainties have been evaluated based on qualitative estimates from an expert panel and historical cost data from other remediation sites (*32).*

Table 3 Use of predefined weights to mimic cost-effectiveness, cost-benefit and value-plural decision approaches. The weights represent the preference by the decision maker for the different criteria and sum to 1.

Weighing of criteria		Cost-effectiveness	Cost-benefit	Value-plural (balanced)
Positive utility	Health risk reduction	0.5	0	0.25
	Socio-economic benefit	0	0.5	0.25
Negative utility	Environmental impact	0	0	0.25
	Cost	0.5	0.5	0.25

Exploration of weights_The SMCA described in this paper emphasis the importance of exploring weights since the criteria only assigns the direction of the net flow vector, whereas the weights directly determine the strength of the net flow. Previous application of SMCA for contaminated sediment remediation used weights representing views of different hypothetical stakeholder interests (*17*). In practice, for stakeholders to be able to embrace a complex decision such as presented in this case, substantial involvement is necessary This may be achieved by the use of deliberative decision processes, but requires a structured involvement strategy reflecting the societal values of all involved stakeholders in order to be successful (*33*). In a formal setting, direct stakeholder participation may in fact also be a source of conflict due to the exposure of direct weighting positions in a decision making process (*12*). As proxies for their policy positions, conflicting interests may mobilize different assessment approaches which in turn emphasize different criteria. To illustrate such a situation, we implement weights to mimic pre-defined decision-making approaches, rather than assumed stakeholder positions (see Table 3).

RESULTS AND DISCUSSION

Preference of alternatives. Table 4 gives mean values for the net flows of each capping alternative. The results indicate a preference for the capping of the highest contaminated areas in either the inner or outer fjord, depending on the decision methodology. Strongest preference is found for capping hot spots in the outer fjord (HOFC) based on cost-benefit principles. From a cost-effectiveness standpoint considering only health risk reduction and cost, capping contaminated hot spots in the inner fjord (HIFC) proves most beneficial. A value-plural weighting balancing weights across all four criteria shows a marginal preference between the two dominant alternatives (HOFC and HIFC) for capping highly contaminated areas in the fjord.

Information Based Sensitivity Analysis. Table 4 does not provide information on the first or second order dominance of remediation alternatives. To do this we must look at the result of the stochastic simulations. The robustness of the conclusions on preferential choices is confirmed by looking at the stochastic dominance between cumulative distributions of net flow (CDF), Figure 2 (the rightmost net flow curves are preferred).

Table 4 Mean net flows from an MCA analysis with 10,000 simulations using the alternative criteria distributions given in Table 2. Bold values show the alternatives with highest preference under each weighting scenario.

Decision methodology	NR	HIFC	HOFC	IFC	OFC	WFC
Cost-effectiveness	0.03	**0.08**	-0.05	-0.07	-0.04	0.05
Cost-benefit	0	0.12	**0.33**	-0.16	0.06	-0.35
Value plural	0.02	**0.20**	0.12	-0.04	-0.12	-0.18

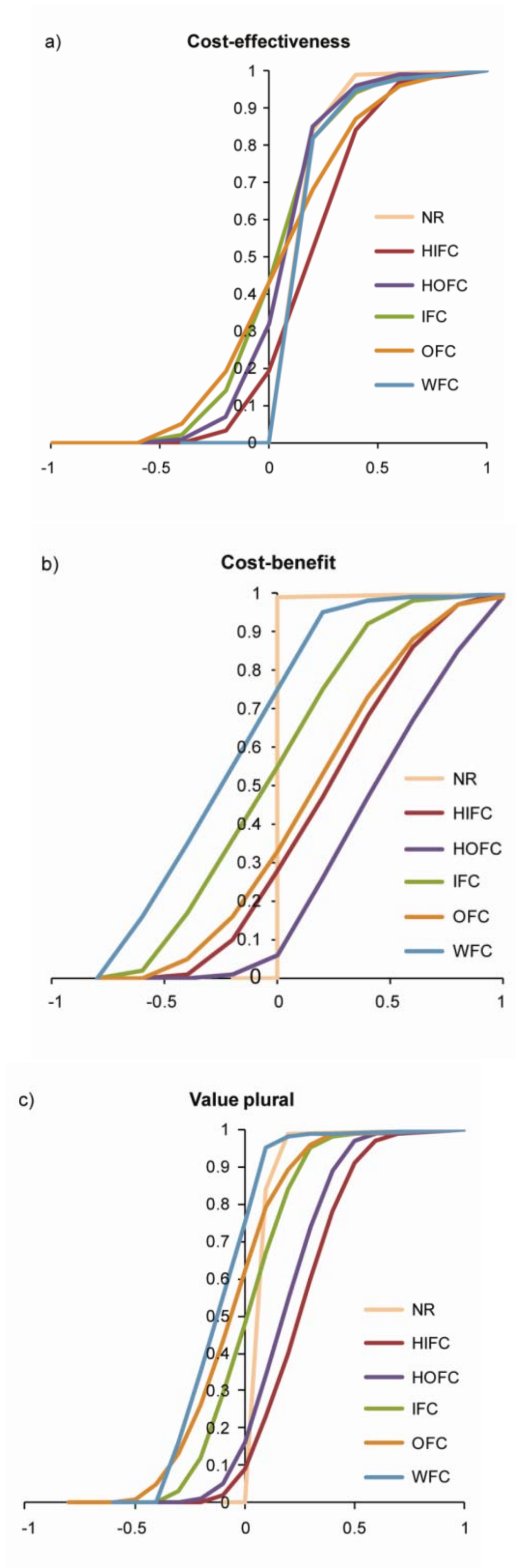


Figure 2 Cumulative distribution functions illustrating preference for each alternative with; a) cost- effectiveness, b) cost-benefit and c) value plural weighing

If a single remedial alternative were to garner the strongest preference (highest net flow) across the entire spectrum and exhibits first order dominance across the alternatives (*25*), further stochastic analysis would not be necessary since first order dominance also implies second order dominance. In this case first dominance is observed for most alternatives. However, especially in the cost-effectiveness scenario, complete rank ordering according to second order dominance is necessary in order to identify when the overall-dominant alternative may be suboptimal in some few specific possible states.

The robustness of the Grenland fjord results are further confirmed by looking at the results of simulations of the expected value of perfect information of each criterion (Partial EVPI) (Figure 3). Evaluation of partial EVPI allows decision-makers and researchers to identify which criteria are significant for decision-making in terms of further data collection.

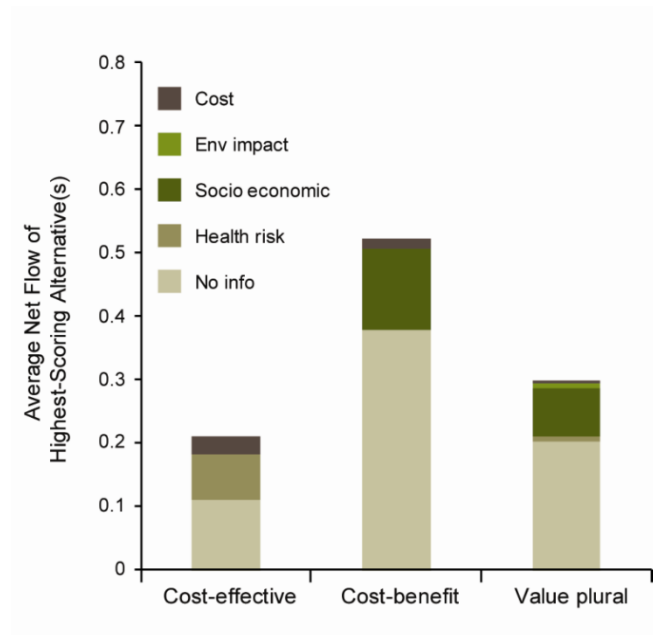


Figure 3 Measuring the partial expected value of achieving perfect information (Partial EVPI) on criteria expressed by average net flow of highest scoring alternative(s)

Perfect information about health risk will give the highest increase in expected value in the cost-effectiveness scenario. In this case, weak second order dominance is observed and generation of better information (reduced uncertainty) may yield more robust results and changes in the rank order (as illustrated by the larger proportion of average net flow ascribed to "health risk" in the cost-effectiveness approach). Collection of information about the socio-economic criterion is preferential in the other scenarios as well, but because there is little ambiguity in rankings in the benefit-cost and value-plural approaches, additional information on socio-economic benefits is not expected to impact the overall decision in most cases (illustrated by the smaller proportion of average net flow ascribed to "socio economic" benefits in these two approaches).

Method Sensitivity. The proposed methodological approach using outranking as a partly compensatory method has advantages in terms of reducing method uncertainties. First, outranking is insensitive to systemic errors as long as the relative rank order is unchanged. This may be beneficial for example using LCA-derived criteria where the absolute results are known to be sensitive to the selection of impact models (*34*). Second, outranking is less sensitive to correlated criteria than internal normalization methods. In our study, health risk reduction, socio-economic benefit and environmental impact are correlated since they are partly or completely derived from effects originating from sediment PCDD/F fluxes entering the ecological and human food chain. In cases using internal normalization methods such as MAUT, this may be a source of error (*35*), but it is unproblematic in outranking methods since the relative preference is determined along each criterion individually. Third, by avoiding normalization of incommensurate criteria, aggregation may be performed without introducing further uncertainties about the relationship between criteria and their respectively normalized values.

However, the partly compensatory nature of outranking means that the magnitude of relative underperformance in a criterion versus the magnitude of over-performance is not considered. This will favor solutions performing well on average over a large number of criteria, masking the effect of superior performance on a single criterion. For example the natural recovery scenario is superior to other alternatives on cost. Even though this criterion may be very critical to a decision maker, this superior performance can only partly compensate for inferior behavior on health risk reduction and socio-economic benefit. Therefore, it is recommended to review criteria performance when using outranking to see whether use of other methods may complement the decision making process.

Applicability in contaminated sediment remediation management. Sustainable management decisions in contaminated sediment management requires use of multiple methods for data aggregation and evaluation (*13*). We find that stochastic multi-criteria analysis using outranking algorithms (SMCA), as a composite method, is a flexible and robust tool in this respect. As with other MCA techniques, it can be used to mimic traditional cost-effectiveness (CE) and cost benefit (CB) analysis decision methods or in weighting schemes that represent pluralistic evaluation.

MCA can also incorporate non-tangible metrics, avoiding problems with valuing all impacts using a single-dimensional monetary indicator as in CBA (*36*). Where SMCA distinguishes itself from other MCA techniques is in combining outranking methods with probabilistic simulation. This opens for addressing uncertainties and incorporating uncertainty analysis, which is beneficial for policy applications (*14*). The problem of many simulation techniques ignoring correlation between evaluation criteria is also mitigated by using an outranking approach (*15*). Including estimates of the value of reducing uncertainty by collecting additional information further enhances the uncertainty analysis.

The results from the case study point to several obstacles that have to be addressed in future research. We assume that a decision with respect to contaminated sediment management will have to incorporate multiple criteria reflecting sustainable values. We have used HERA to reflect the local risk perspective, contingent valuation to reflect socio-economic benefits and we have complemented the financial evaluation of cost by also adding environmental impact derived from LCA analysis. This implies a holistic view of the decision maker requiring use of new and perhaps more controversial methodologies than normally practiced in a risk based management of contaminated sediments (*37*). All models, being representations of the real world, are also subject to biases. For example, contingent valuation of non-market environmental benefit is subjected to discussion for providing hypothetical rather than actual revealed willingness to pay for remediation (*10*). Use of risk based health indicators are precautionary site specific estimates and not an accurate measure of actual exposure to contaminants (*37*), while LCA uses significantly less conservative human and ecotoxicological models (*38*). While the method shown here opens up for jointly assessing different impact models, future research could also explore alternative strategies to evaluating social preferences using deliberative methods. In particular how uncertainty analysis options demonstrated here may aid or hamper deliberation given differences in interests and risk aversion of stakeholders.

Another important aspect is the information lost when synthesizing data within the outranking algorithm. As opposed to a CEA or CBA which is fully quantitative, this methodology gives only a relative rank order between alternatives. There is effectively a trade-off between evaluating the strength of preferences for alternatives versus controlling for biases in the uncertainty analysis due to correlation in the measures of impacts presented to stakeholders. This calls, as demonstrated in this case, for a sufficient analysis of the robustness of the results as presented in this paper.

Finally, this study has focused on contaminated sediment remediation, but the method should be applicable to other multi-dimensional environmental decision problems requiring a pluralistic value approach. We therefore advocate further research acknowledging the necessity of plural methodologies to enhance sustainability in environmental decision analysis.

ACKNOWLEDGMENTS

The authors would like to thank previous studies for the work performed on the Grenland fjord and especially the Opticap project (www.opticap.no), the SEDFLEX project and the Norwegian Research Council for financing the studies. A special gratitude to Bjørn Vidar Vangelsten NGI for adapting the SMCA to use of skewed normal distributions. The third and last author would like to acknowledge the funding from the Dredging Operation Environmental Research (DOER) program by the US Army Corps of Engineers. Permission was granted by the USACE Chief of Engineers to publish this material.

BRIEF

Use of stochastic multi-criteria analysis addressing environmental, social and economical values is proposed as a step towards sustainable decision making in contaminated sediment management

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Appendix B: Secondary papers

SP1: Sediment and society: an approach for assessing management of contaminated sediments and stakeholder involvement in Norway

Oen M.P.; Sparrevik M.; Barton D.N.; Sekhar U.D.; Ellen G.J.; Breedveld G.D.; Skei J.; Slob A.

Journal of Soils and Sediment. **2010,** 10 (2), 202–208.

Management options for large-scale contaminated sediment remediation projects can be challenging with regard to competing stakeholder interests. This has become apparent during the Oslofjord sediment remediation project (2005–2009) which caused considerable public discussion. To learn from this project, the ‘Sediment and society’ project was initiated to develop a collaborative approach that will incorporate local and scientific knowledge in order to achieve mutual gains, win-win outcomes for the stakeholders, in the management of contaminated marine sediments. The project focuses on two Norwegian harbours: Oslo Harbour and Bergen Harbour. The Oslo Harbour case has been analysed ex-post, using elements of risk governance: participation, communication, information/ knowledge and risk perception. The Bergen Harbour case is focused on the establishment of a citizens' jury as well as a stakeholder panel in Bergen Harbour. Thus far, the results suggest three important commonalities or challenges for stakeholder involvement: (1) how to include people who have important management information and local knowledge, but not much influence in the decision-making process; (2) how to secure resources to ensure participation and (3) how to engage and motivate stakeholders to participate early in the sediment remediation planning process.

SP2: Use of Life Cycle Assessment for Improved decision making in contaminated sediment remediation

Sparrevik M.; Linkov I.

Integrated Environmental Assessment and Management. **2011**, 7 (2), 304-305.

The selection of remedial alternatives for contaminated sediments is a complex process that balances environmental, social and economic aspects. The decision to remediate and the identification of relevant remedial options are often based on quantitative ecological risk assessment (ERA) (Bridges et al. 2006) with qualitative consideration of other factors within frameworks of feasibility studies and environmental impact assessments. While ERA is suitable for assessing whether contaminated sediments constitute an unacceptable environmental risk or whether remediation may reduce this risk below acceptable threshold levels, the life cycle impact of a remedial action is often overlooked. Specifically, environmental consequences associated with the use of energy and other resources and environmental impacts incurred during remediation may differ between different remedial strategies. Furthermore, any beneficial uses of removed sediments are not integrated in the ERA. We argue that quantitative Life Cycle Assessment (LCA) can supplement ERA in this respect, to create an enhanced systems approach to sediment management.

SP3: Coupling Multi-Criteria Decision Analysis, Life-Cycle Assessment, and Risk Assessment for Emerging Threats

Linkov I.; Seager T.P.
(Contribution by Magnus Sparrevik, se acknowledgement below)

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The recent emergence of new materials, technologies, and other environmental stressors in both the marketplace and the public consciousness coincides with increased recognition of the importance of an integrated systems approach to environmental health and safety that includes life-cycle thinking, public participation, and adaptive management of risks associated with emerging threats. While the fields of risk assessment and risk management have always operated in situations of uncertainty, emerging threats greatly increase the challenge to risk analysts already hard-pressed to elucidate the potential hazards of traditional chemicals. A recent report from the National Research Council (NRC) recognizes that ten years or more is typically required to complete risk assessments for environmentally important chemicals. Given the extraordinarily high levels of variability and uncertainty intrinsic to emerging materials (such as nanomaterials), the data and experimental resources required to complete technical analyses can be expected to be even greater than for traditional materials. Consequently, it seems highly unlikely that the classic risk analytic framework of hazard identification, source term characterization, environmental fate and transportmodeling, exposure assessment, and dose-response assessment could be sufficient in practice to keep pace with the rate of technical innovation in emerging technologies. A series of NRC committees dating back to at least 1989 have reiterated the importance of a "decision-directed" approach to risk management that allocates analytic resources to discovering new information that is most informative in a specific decision context. Given the consistency of these recommendations over the last two decades, it may seem remarkable that a structured decision analytic framework has yet to be adopted by risk management agencies, such as the Environmental Protection Agency (EPA) and Department of Homeland Security (DHS). Although the recent NRC report includes "a framework for riskbased decision-making," the framework structures only the riskanalytic aspects of risk management, not the decision-analytic which may illustrate the persistence of the ideal vision of detached scientific objectivity in risk analysis. By contrast, a recent NRC report with regard to public participation emphasizes the importance of deliberative processes for bringing together disparate public and stakeholder views to help generate decision criteria, rank-order alternatives, deal with uncertainty in regard to competing objectives, and formulate management trade-offs between objectives in the context of risk. Such deliberative processes can be incorporated in a multicriteria decision-analytic approach (MCDA), as has been proposed for remediation of contaminated sites, life-cycle impact assessment, sustainability, environmental policy, or integrated risk assessment and life-cycle assessment. MCDA refers to a collection of methods used to impart structure to decision processes that invoke incommensurate or irreducible objectives, multiple and divergent stakeholders, and (in many cases) incomplete information. This paper presents an approach for using MCDA to integrate uncertain information collected from risk analysis and life-cycle assessment in the context of emerging environmental threats. The objective of this approach is to establish an analytic basis for prioritizing research needs that are most informative to decision-makers such as product developers, regulators, or end-users.