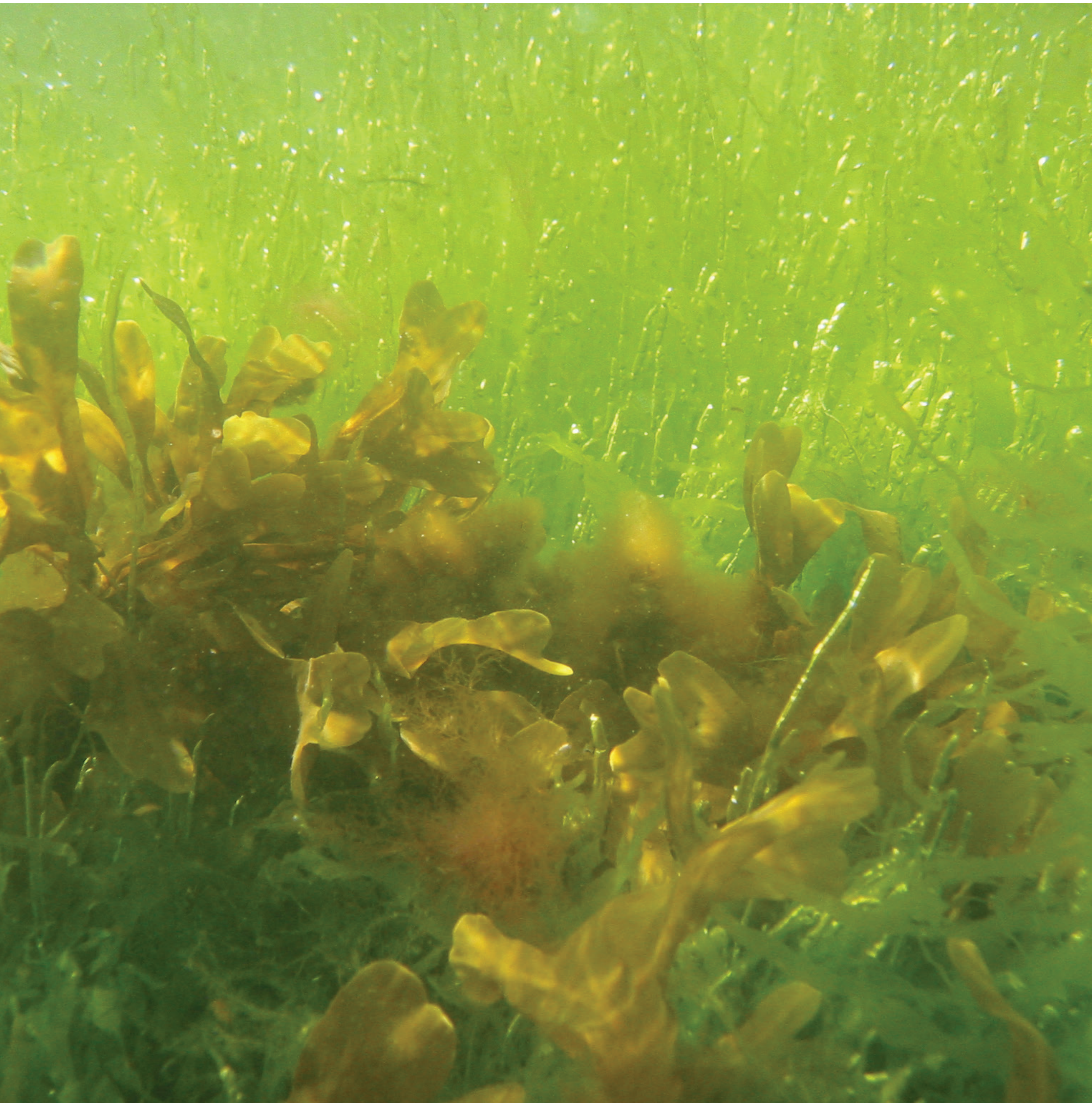
MOPODECO

Modeling of the Potential coverage of habitat-forming species and Development of tools to evaluate the Conservation status of the marine Annex I habitats



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Modeling of the Potential coverage of habitat-forming species and Development of tools to evaluate the Conservation status of the marine Annex I habitats

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TemaNord 2012:532

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TemaNord 2012:532
ISBN 978-92-893-2368-0
http://dx.doi.org/10.6027/TN2012-532

Cover photo: Martin Isaeus/AquaBiota

This publication has been published with financial support by the Nordic Council of Ministers. However, the contents of this publication do not necessarily reflect the views, policies or recommendations of the Nordic Council of Ministers.

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Content

Executive summary

The main aim of MOPODECO is to fill the gaps in harmonisation of the definition of the EU Habitat Directive Annex I habitats and the view on the main pressures and threats to these habitats between the Nordic countries and countries surrounding the Baltic Sea. The project also aims to develop standards for describing the functional characteristics of the Annex I habitats, including standards for application of modelling tools for quantifying the coverage of habitat-forming species. In MOPODECO, the harmonised definitions, functional descriptors and habitat model application standards are integrated into proposals for unified indicators of favourable conservation status for Annex I habitats in Nordic waters. Common standards for assessing the favourable conservation status of these habitats are suggested by applying these indicators in case studies from the northern and southern parts of the Baltic Sea.

The project has been structured into four main activities:

- Identification of pressures and threats to the Annex I habitats
- Harmonisation of the definitions of Annex I habitats
- Spatial models of habitat-forming species
- Development of indicators and tools for assessment of favourable conservation status of Annex I habitats

Identification of pressures and threats as well as the sensitivity of the Annex I habitats to these pressures is a prerequisite in order to sustain or improve the conservation status of marine Annex I habitats. Thus, a pressure evaluation matrix and a table of conflicts and compatibilities among human activities were outlined for the habitats in the Baltic Sea and the Kattegat. The impacts evaluated in the pressure evaluation matrix were split into a) impacts that haven't necessarily been ascertained yet, but are by literature or expert appraisal considered possible, and b) impacts that have been observed, modeled and/or reported, and therefore, considered likely. The pairs of human activities compared were determined as either a) compatible (not in conflict with each other), b) probably compatible (no indication of the contrary could be thought of or found in the references) and c) incompatible (they conflict with each other thus excluding each other on the same site at the same time).

The marine underwater habitats listed in Annex I of the Directive are mainly large habitat complexes, and the views and definitions of these habitats vary between the EU member states. The work on harmonisation of the definitions of Annex I habitats has focused on clarifying the differences in the definitions between the countries of the Baltic Sea area and between the Nordic countries for reefs, sand banks and lagoons. The work has been undertaken with an eye on the geographical continuum of the Annex I habitats, and the need to develop the practical interpretation in order to reach similar and harmonized practical implementations of the EU Habitats Directive. The national detailed definitions for habitats are discussed and compared with the official EU definitions as well as with each other.

Three sub-types of biogenic reefs and twelve sub-types of geogenic reefs have been defined from various sources as containing features characteristic for reefs of the Baltic Sea. Further development of measures and thresholds for qualifying features of reefs is needed in order to support a common understanding of this habitat type. However, development of measures and thresholds for individual sub-types rather than reef habitat type in general is recommended. Due to the high variability of the coastal benthic environment, contribution of regional experts comprehensively covering different Baltic Sea regions will be of utmost importance in generating common measures.

Sandbanks must be topographically distinct from the surrounding seabed, and the sandbank height criteria used in the selection of banks for the European Submerged Sandbank Database has been set to a minimum of 5 m with respect to the surrounding seabed. The slope, however, has not been determined as quantitative criteria and may vary considerably depending on type of a sandbank in the Baltic. More gentle slopes can be characteristic for sandbanks formed in the process of land upheaval (e.g. in Finnish waters) in comparison to open Baltic sandbanks. Several distrinct types of sandbanks can be identified each characterized by the dominant biota. In this work we have focused on three types: (1) sandy bottoms almost without vegetation but with a large versatility in the sediment, (2) eelgrass (*Zostera marina*) meadows and other rooted macrophytes with less versatility in the sediment, and (3) mussel banks with more than 25% coverage.

The proposed definition for lagoons has been based on features common to all Baltic sub-types of lagoons in order to fully cover the diversity of this habitat type in the Baltic Sea. Baltic lagoons are expanses of shallow coastal waters, wholly or partially separated from the sea by a barrier: sand spit, shingle, gravel or by rocks and bedrocks or by land upheaval. The salinity of the Baltic lagoons varies depending on the freshwater input and water exchange with the sea and may range from brackish water to freshwater. Topographically flat lagoons with gently sloping shores and surface area of up to several hundreds of km^2 are typically situated in the south-western and south-eastern parts of the Baltic Sea, whereas small lagoons surrounded by steeper rocks and/or formed by land upheaval are covering less than 30 ha are characteristic for the northern Baltic. The depth of the Baltic lagoons rarely exceeds 5 m. Baltic lagoons are mainly differentiated according to the type of connection with the sea and the stage of geomorphological succession, and 9 sub-types have been classified as lagoons in different Baltic Sea countries. Development of measures and threshold values for individual sub-types of lagoons is needed in order to support a common understanding of this habitat type. Due to the high variability of the coastal benthic environment, the contribution of regional experts comprehensively covering different Baltic Sea regions will be of utmost importance in generating common measures for lagoons.

Central to the problem of characterising the ecosystem functions of Annex I habitats is the lack of quantitative data and data collected in a uniform way on the coverage and state of habitat-forming species, e.g. stands of submerged vegetation and mussel beds, which mark essential habitats to benthic as well as pelagic animals. Thus, descriptions are needed to link Annex 1 habitats to the structure and functioning characteristics of the habitats and especially to the coverage and state of habitat-forming species or other biological key elements suitable as indicators. These descriptions may then be used as common indicators across the Baltic Sea and Nordic countries. Mapping of habitat-forming species has so far largely relied on surveys and remotely sensed technologies without attempting to capitalize on the development of empirical and statistical spatial modelling techniques to extrapolate survey results to wider areas. By developing these tools in co-operation across the Nordic and Baltic Sea countries, this project has helped to establish common data sets and harmonize the practices used in different countries. The project has provided the first attempt to estimate the coverage of habitat-forming species and biological key-elements for a whole region on the basis of co-ordinated spatial models using state-of-the-art techniques and a combination of structural and dynamic parameters. The developed models should provide a useful showcase for the follow-up to the recommendations of the Nordic Forum on MPAs in Marine Spatial Planning and the work carried out on mapping and modelling of marine habitats in the Baltic Sea region under the Interreg III B project BALANCE.

One of the key modeling activities has been a large scale test of total erect macroalgal cover as a common metric of reef habitats from Skagerrak to the eastern Baltic Sea. Total cover of algal vegetation is integrative over time and available over a large depth range when suitable hard substrate is available. At the same time total vegetation cover is highly relevant in an ecological perspective as the benthic macroalgal vegetation in general plays an important role in structuring hard bottom habitats and plays an important role as primary producer.

Total macroalgal cover can be used as an indicator of both the reef and water quality over a wide geographic range, as it reflects the most important abiotic and biotic factors that influence hardbottom habitats in the photic zone. Spatial variations in algal variables sampled at biological monitoring stations were related to the physio-chemical variables salinity, nutrient concentration, chlorophyll concentration and Secchi depth sampled at nearby oceanographic stations. In order to facilitate a direct comparison between stations annual and area-specific total algal cover was estimated for two chosen standard depths, 7 and 15 m and linked with mean values of water quality variables in the 6 months period (January–June) prior to macroalgae monitoring in the summer season. The results of the statistical models supported the hypothesis that eutrophication has significant negative effects on total macroalgal cover across the large region from the open Norwegian North Sea to the inner Baltic Sea. On the other hand our hypotheses regarding a positive effect of salinity on total cover and Secchi depth were only partly confirmed. In the more saline open waters, salinity did show a positive correlation with Secchi depth and total cover.

Despite the variability in the relationships between total macroalgal cover and environmental variables induced by differences in methodology and incomplete physic-chemical description of the sampling sites our results demonstrate that it is possible to describe a key element for hard bottom habitats, in this case "total algal cover" over wide geographical ranges. The study also highlights the advantage of applying models which allow harmonization and comparison of data sampled at different depths, years, seasons and by different divers and subsequently relating the data set to physico-chemical regulating factors. A future adjustment of national monitoring programs to include a direct measurement of total algal cover on hard stable substrate could be implemented to a very low extra cost and would reduce the random variability in data and thereby improve the predictive power of future models. Coupled information on macroalgal cover and environmental variables can thus further be useful in quantifying the importance of reefs by allowing the extrapolation of existing site-specific total and cumulative cover models to other local reef areas with available high resolution data on bathymetry and substrate. The potential for developing regional models was demonstrated using Kim's Top and Lilla Middelgrund in the Danish and Swedish part of Kattegat as a case study.

The brown seaweed bladderwrack *Fucus vesiculosus* constitutes a true key species in the Baltic marine ecosystem, and apart from being characteristic of many reef areas, the bushes of bladderwrack also serve as shelter and feeding habitat to many crustaceans, a wide array of invertebrates, such as sea snails, and many fish species with their offspring. As most monitoring data on the coverage of bladderwracks are biased towards areas of known presence of the species the potential for developing regional models of the coverage of bladderwrack was tested using a presence-only statistical model technique MaxEnt using data from 7 different countries around the Baltic Sea.

The bladderwrack presence models were developed at a resolution of 200 m using fetch, depth, salinity, surface sediments and salinity as predictor variables, and they were validated against independent presence-absence data. The predictive ability of the model was very good (AUC=0.946). However, when validating the model with presence/absence external data the model performed poorly (AUC=0.62). This is partly explained by the fact that the model is somewhat overfitted, i.e. fitted too close to the data and not able to generalize to the whole predicted area. The problem of overfitting in this case is mainly due to the model being fitted to data that is unevenly geographically distributed and does not represent the whole area in a correct way. However, the major problem is the poor quality of the predictor layers available for the bladderwrack prediction. Especially, the lack of information on substrate in a resolution applicable for species modelling is evident. Therefore, the result must be interpreted as probability of bladderwrack presence provided there is appropriate hard substrate. For local scale planning detailed modeling should be used based on more accurate input layers and evenly distributed field data both environmentally and geographically.

A deterministic model was established with the aim to estimate the coverage of blue mussel *Mytilus edulis*. The modeling strategy differed from that used for modeling coverage of macro-algae and *bladderwrack* by estimating growth of blue mussels directly on the basis of the available food supply (phytoplankton) modeled by DHI's ecosystem model "BANSAI" for the Baltic Sea. The *Mytilus* model design was based on three model elements: a regional and local hydrodynamic model, a biogeochemical model and a deterministic filter-feeder model. The ecological model consists of an eutrophication model describing the pelagic system with 13 state variables, and seven state variables describing the exchangeable nitrogen and phosphorous pools in the sediment. The pelagic system includes phytoplankton, described in terms of their concentration of carbon, nitrogen and phosphorus, chlorophyll-a, zooplankton, detritus, inorganic nutrients, total N and P and dissolved Oxygen.

A carrying capacity (CC) model at 617 m resolution was established using the output from the hydrodynamic and ecological models. The CC model built on the a concept of combining a physiology-based growth model for a standard individual with an advection term that replenish the food ingested by filter-feeders. On large scale CC depends on the local primary production and on smaller scale current speed plays an increasing role for CC. We used a combination of a quadratic function of modelled current speed and a functional response between phytoplankton concentration and individual mussel growth to express an index of carrying capacity for benthic suspension feeders. In nature, filter-feeding bivalves aggregate in dense assemblages if current speeds are high, e.g. in tidal areas as the Wadden Sea. In low-current environments plankton algae removed by filtration are only slowly replenished and such environments cannot sustain dense populations. Therefore, the growth functions were supplemented by an equation that describes the replenishment of food.

The blue mussel model was validated using independent coverage data collected by Hans Kautsky, Stockholm University from a range of locations along the Swedish east coast between 2005 and 2007 and by Aarhus University for the Danish part of the Kattegat between 1992 and 2009. The ROC plot for the model using a threshold for presence at a CC value of 30 indicated that the deterministic mussel model performed very well in the Baltic (AUC = 0.91), and accurately predicted the areas with a coverage of blue mussels exceeding 10%. The ROC plot for the model using external validation data from the Danish part of the Kattegat indicated that the deterministic mussel model performed badly in the Kattegat, where the AUC value for predicted coverage of blue mussels exceeding 10% is as low as 0.49. This result was expected however, as predation by sea stars is known to eliminate extensive coverage of blue mussels from areas north of the Belt Sea. As the deterministic mussel model applied here does not include predation processes the predicted potential growth of blue mussels in areas of high abundance of sea stars is over-estimated.

The results of the modeling activities have highlighted both the potential and limitations for applying the available suite of habitat models at the regional scale as a means to provide statistics on the status of the structure and functions of reefs and other benthic habitats. The model work has documented that with calibration data the coverage of blue mussel and other suspension-feeding invertebrates can now be predicted confidently using deterministic ecological models over large regions of the Baltic Sea and other Nordic waters where intensive predation by sea stars is not present. Within the short term it is expected that higher resolution depth and sediment data will be available for the entire distribution range of the bladderwrack. This will allow fine-scale habitat models to be developed and applied to predict the coverage of this and other key species of vegetation in the Baltic Sea and other Nordic waters.

The aim of Work package 3 was to develop simple conceptual models which describe the linkages between pressures and structure and functioning of habitat-forming species in different habitats, and to test various tools for assessment of favourable conservation status of Annex I habitats. The conceptual models have been made for three Habitats Directive's Annex I habitats: reefs (1170), sandbanks (1110) and lagoons (1150) focusing on characteristics that could be used as common indicators across the Baltic Sea and Nordic countries. The three most relevant pressures were selected in line with the Marine Strategy Framework Directive Annex III table 2: eutrophication, physical disturbance and climate change.

The suggested indicators are species, species groups which occur commonly in the habitat, and are easy to identify and monitor. The ecology of the indicators is quite well known and they are known to respond to enhanced nutrient levels in water. The conceptual models are simple descriptions of the most important effects of the pressures on habitats. In the conceptual models pressures and associated habitat effects are first decribed. In the conceptual models of eutrophication habitat change levels are also described: habitat alteration, habitat fragmentation and habitat loss. In the conceptual models of physical disturbance and climate change the habitat change levels are absent because they are more difficult to identify. In the conceptual models of eutrophication different kinds of indicators can indicate the effects on these three habitat change levels.

For reefs the following key indicators are recommended:

- Total/cumulative algal cover
- Coverage of perennial macroalgae
- Fraction of opportunistic algae
- Number of late successional species
- Depth limit of perennial macroalgae
- Species composition of habitat-forming species
- Density/biomass and size of mussels

For sandbanks the following key indicators are recommended:

- Coverage of angiosperms (+charophytes)
- Depth limit of angiosperms (+charophytes)
- Species diversity of angiosperms
- Fraction of opportunistic algae
- Species composition of habitat-forming species
- Species composition of benthic fauna
- For lagoons the following key indicators are recommended
- Coverage of angiosperms and charophytes (+perennial macrolagae)
- Depth limit of angiosperms and charophytes (+perennial macroalgae)
- Loss of charophyte-communities
- Fraction of opportunistic algae/opportunists in plant-communities
- Species composition of vegetation
- Species diversity of habitat-forming species

Several case studies were undertaken to apply and modify the recommended indicators using the habitat models developed and available assessment tools. The assessments of reef quality in the Kattegat could be made using the same set of total coverage models that were used for mapping vegetation cover of reef habitats. As no "historic" data existed that could be used as a reference and target level for development of algal vegetation we had to use the same models to predict a target given an input of nutrient load assumed as reasonable values for the boundary between favourable and unfavourable conservation status.

Three case studies for assessment of specific habitat quality of NATURA 2000 areas were undertaken in Finland; two in the coastal lagoons of Maa-Sarvi flad in the Bothnian Bay and Danskogfladan flad in Hanko peninsula on the south coast of Finland, and one in the Granbusken Reef in the Hanko Peninsula. Species composition, numbers and coverages of charophyte, angiosperm, and endangered species, and the coverage of filamentous macroalgae and the lower depth limit of the *Fucus vesiculosus* belt were used as indicators.

Further, it was demonstrated using the HELCOM Biodiversity Assessment Tool (BEAT) how the indicators including site-specific target values for total cover of submerged aquatic vegetation are useful in regard to assessment of conservation status within four Natura 2000 areas, all being stone reefs, in the Kattegat. Acknowledging that long-lived macroalgae are characteristic for light exposed stone reefs, the case study highlighted the need to include macroalgae in future assessments of the conservation status of communities living on stone reefs.

Concluding, the MOPODECO results support the Environmental Action Programme 2009–2012 of the Nordic Council of Ministers in several ways:

- The harmonisation of the definitions of Annex I habitats has assisted the harmonization of environmental protection work in the Nordic countries
- The application and tests of regional models of habitat forming species and the development of indicators and tools for assessment of favourable conservation status of Annex I habitats have assisted the implementation of the EC Marine Strategy in the Nordic environment
- The development of indicators and tools for assessment of favourable conservation status of Annex I habitats has assisted the development of the HELCOM EcoQOs
- The pressure evaluation matrix has strengthened the sustainable use of the resources in the sea and the basis for ecosystem approach to management
- The habitat modelling activities has assisted in filling the gaps on habitat mapping pointed out as a priority action for 2007–2008 in the mid term evaluation of the Environmental Action Programme for 2005–2008
- The pressure evaluation matrix and the development of indicators and tools for assessment of favourable conservation status of Annex I habitats have assisted in meeting the goals set in the Strategy for Sustainable Development
- The development and tests of regional habitat models and the development of indicators and tools for assessment of favourable conservation status of Annex I habitats have supported HELCOM in developing assessment tools for biodiversity and conservation status of the species and habitats in the Baltic Sea

Acknowledgements

The following persons are acknowledged for help on and/or contributions to the report:

- Catherine Munsterhjelm, University of Helsinki: macrophytes and general characteristics of the Danskogfladan flad case study
- Essi Keskinen, Metsähallitus: characteristics of the Maa-Sarvi flad case study
- Hanna Piepponen, Marine Research Centre, SYKE: production of maps
- Visa Mäkelä and Samuli Neuvonen, SYKE: the SYKE aquatic macrophyte database
- Antti Lappalainen, Finnish game and fisheries research institute: fish of the Danskogfladan flad
- Kirsi Kostamo and Heikki Peltonen, SYKE: supplementary information on macroalgae and fish, respectively
- Mats Westerbom, Metsähallitus: general information on Finnish case studies
- Ilpo Huolman, Centre for Economic Development, Transport and the Environment of the Uusimaa region: information on nature protection areas outside Tvärminne
- Aino Juslén, Ilpo Mannerkorpi and Terhi Ryttäri, Biodiversity unit at the Natural Environment Centre, SYKE: information about threatened species in Finland
- Janne Gitmark (NIVA, macroalgae field sampling, Norway)
- Birger Bjerkeng and Anna Birgitta Ledang (NIVA, physico-chemcial data from Norway)
- Rolf Karez (LLUR, nutrient data from Schleswig-Holstein, Germany).
- Mario von Weber (LUNG, nutrient data from Mecklenburg-Vorpommern, Germany)
- Marine Research Department (physico-chemcial data from Lithuania)

Staff from the Danish Environmental Centers for data collections in coastal waters.

The EU project WISER co-financed some of the additional data collection and statistical analysis carried out in chapter 4.1 (contract # 226273). This additional financiation allowed us to include a large extra dataset from Germany.

SMHI delivered hydrographical datasets from the Shark data base. The University of Upsala provided data on nutrient run off.

1. Introduction

The EU Habitat Directive aims to achieve the favourable conservation status of the Annex I habitats. The conservation status of habitats and species listed in the Annexes of the Habitats Directive are reported to the EU by the Member States every six years. This takes place by using the EU reporting formats. In EU there have been two reporting rounds which have been conducted without common understanding of the favourable conservation status of the marine habitats. It is essential to reach similar patters of interpretation by harmonizing the concepts of definitions such as the favourable conservation status and identifying the pressures and threats of the habitats along the coastal areas of the Baltic Sea and the Nordic countries in order to plan the sustainable use of marine resources and develop tools for spatial planning dealing with nature conservation.

The marine underwater habitats listed in Annex I of the Directive are mainly large habitat complexes, and the views and definitions of these habitats vary between countries. Also many countries are still lacking practical tools to assess the conservation status of these habitats/habitat complexes and the assessment is done based only on expert consultation. This results in varying practices and methods between countries, and more importantly, also in varying results when reporting the status of these habitats. In the past years there have not been active efforts to produce harmonized concepts how these habitat descriptions have been interpreted, which characters are used when assessing their conservation status or when the favourable conservation status is reached.

Central to the problem is the lack of quantitative data on the coverage and state of habitat-forming species, e.g. stands of submerged vegetation and mussel beds which mark essential habitats to benthic as well as pelagic animals formed within the large-scale habitats. Thus, descriptions are needed to link Annex 1 habitats to the structure and functioning characteristics of the habitats and especially to the coverage and state of habitat-forming species or other biological key elements suitable as indicators. These descriptions may then be used as common indicators across the Baltic Sea and Nordic countries.

In order to harmonize the views on these habitats and the way the favourable conservation status is assessed and defined, co-operation across the countries is needed. Co-operation is also needed to fully capitalise on the recent developments in spatial modelling techniques. Across the Baltic Sea these habitats share common characteristics, and some of them are even unique to the Baltic Sea. Therefore co-operation across the Baltic Sea, between Nordic countries as well as the adjacent area is essential to harmonize the methods, distribution models and tools used when assessing the conservation status of these habitats. This project has brought together marine experts and modellers that have worked on the EU Habitat Directive implementation process recently. We are confident that the results of this cooperative project will ease and harmonize the reporting to EU in the following years.

By following a common strategy for assessing the status of Annex I habitats, the Baltic Sea and Nordic countries can more easily join forces when tackling problems concerning the marine habitats and the biodiversity. It is essential to apply a similar strategy in order to make use of the best expertise, to benefit from the existing national experiences and to include best practices for repeated reporting on biodiversity to the European Union and reporting to HELCOM on the state of the coastal zone.

The project has been structured into three work packages:

- Identification of pressures and threats to Annex I habitats, and harmonisation of the definitions of Annex I habitats
- Spatial models of habitat-forming species
- Development of indicators and tools for assessment of favourable conservation status of Annex I habitats

The work package on identification of threats and harmonisation of the definitions of Annex I habitats have focused its work on developing a common pressure evaluation matrix for the Baltic Sea, and on clarifying the differences in the definitions between the countries of the Baltic Sea area and between the Nordic countries for reefs, sand banks and lagoons. The results of this work are presented in Chapters 2 and 3, with an eye on the geographical continuum of the Annex I habitats, and the need to develop the practical interpretation in order to reach similar and harmonized practical implementation of Habitats Directive. The national detailed definitions for habitats are discussed and compared with the official EU definitions as well as with each other.

The work on spatial models of habitat-forming species, which is presented in chapter 4, builds on experiences from e.g. BALANCE and ALGAMONY. The work package has developed landscape-explicit marine spatial models of the potential coverage of *Fucus vesiculosus* and *Mytilus* *edulis* in the Baltic Sea. This work has demonstrated how both the description and assessment of favourable conservation status of Annex I habitats can be improved by adding modelled data on key biological elements on these habitats. While showing how the development of wave exposure models and time series of hydrodynamic and ecological model data for the entire North Sea, Skagerrak, Kattegat and Baltic Sea have improved the potential for developing regional habitat models the chapter also provides guidance on the present limitations of the application of the various modelling techniques.

The development of indicators and tools to assess the favourable conservation status of the Annex I habitats has built on experiences and tools developed in HELCOM and ALGAMONY. This work package has addressed the urgent need to identify key factors for determining the conservation status of these habitats as well as to develop biodiversity indicators for the entire Baltic Sea and the Nordic waters, and to reach coherence in setting the boundaries between favourable vs. non-favourable conservation status. The results integrate the outputs from the other two work packages, and are presented both in terms of conceptual response models and indicators in relation to each of the selected Annex I habitats as well as in terms of case studies in which indicators and tools are tested.

2. Pressure Evaluation Matrix

2.1 Introduction

Identification of pressures and threats as well as the sensitivity of the habitats to these pressures is a prerequisite in order to sustain or improve the conservation status of marine Annex I habitats. Below, a pressure evaluation matrix and a table of conflicts and compatibilities among human activities are outlined. The geographical area is defined as the Baltic Sea, including Kattegat.

The selection of variables in the matrices is as follows. The human activities considered relevant in this context were chosen based on:

- The UNESCO/IOC marine spatial planning manual (Ehler and Douvere 2009)
- The HELCOM assessment of biodiversity in the Baltic Sea (HELCOM 2009a)
- The MarLIN "marine and coastal activities to environmental factors" matrix. Available at http://www.marlin.ac.uk/

The selection of relevant activities was a compromise and a combination of the ones in the references above.

The impacts caused by the selected human activities were chosen based on:

- The EU marine strategy framework directive, Annex III. Available at http://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=OJ:L:2008:164:0019:0040:EN:PDF
- The MarLIN "marine and coastal activities to environmental factors" matrix. Available at http://www.marlin.ac.uk/

The selection of impacts was a compromise and a combination of the ones in these references.

The habitats are the ones covered in the MOPODECO project, which were selected from Annex I of the EU Habitats directive, available at http:/eur- lex.europa.eu/LexUriServ/ LexUriServ.do?uri=CONSLEG:1992L0043:20070101:EN:PDF.

The selected species are the ones that are either important habitat-forming species, especially vulnerable, typical or good indicator species in the Baltic Sea. They were chosen based on:

- The EC interpretation manual of European Union habitats (European Commission 2007b)
- The HELCOM fact sheets (HELCOM 2009b)

2.2 Matrices and keys to their symbols

There are two matrices:
Left part: *Pressure evaluation matrix*: Pressures and impacts caused by human activities in the marine environment.

Key to symbols:

- Possible effect: an impact that hasn't necessarily been ascertained yet, but is by literature or expert appraisal considered possible
- Probable effect: an impact that has been observed, modelled and/or reported. Therefore, it is considered likely

Right part: Influence of human activities on marine habitats and species.

Both the intensity of impact and the distance of influence are evaluated. The distance of influence is the distance from which a particular human activity operation will have an influence on the target habitat/species.

Key to symbols: Distances of influence

- *No effect/not relevant*: The considered human activity has no effect or is not relevant (e.g wouldn't take place) on the particular habitat/species in question
- *On-site/local*: The considered human activity has an effect on the habitat/species in question only when occurring on-site or locally, that is within the particular habitat itself or immediately on/next to an individual of the particular species
- *Areal*: The considered human activity has an effect on the particular habitat/species in question if it takes place in the same area (but not necessarily at the habitat itself) where the habitat or individual of the species is situated. The size of this area is place specific, but in most circumstances with a maximum radius of about 10 km

- *Sub-basin*: The considered human activity has an effect on the particular habitat/species in question if it takes place in the same sub-basin (*sensu* HELCOM)
- *Baltic wide*: The considered human activity has an effect on the particular habitat/species in question if it takes place anywhere in the Baltic Sea

Conflicts and compatibilities among human activities

Key to symbols:

- Compatible: The compared two human activities are compatible, if they are not in conflict with each other and can thus be carried out on the same site at the same time
- Probably compatible: The compared two human activities are probably compatible with each other, if no indication of the contrary could be thought of or found in the references
- Incompatible: The compared two human activities are not compatible, if they conflict with each other thus excluding each other on the same site at the same time

2.3 Annotations

The assessments of impacts suggested in the matrices are founded both on references (see reference list below) and expert appraisals. For Figure 2, the effects on habitats are assessed both on the selected habitat-forming species (MarLIN approach: the sensitivity of a community or biotope is dependent on the species within it) and on the physical/chemical characteristics of the habitat. In Figure 3, the compatibilities and conflicts are assessed mainly spatially. Therefore, a potential spatial conflict may disappear in a different time period.

The effects of wastes originating from the different human activities are not treated separately, but are included in the impacts of the respective activity. For different coastal and offshore constructions, the effects of construction, usage and removal phases are all considered in the impacts of the construction in question.

The definition of habitat 1180 (Submarine structures made by leaking gases) comprises two different types, “bubbling reefs” and pockmarks. In the Baltic Sea, bubbling reefs are found only in Kattegat, whereas pockmarks, although most likely present in the Baltic Sea area, have not yet been assessed. Here it is assumed that they occur in deep soft sediment seabed areas of the Baltic Sea.

Figure 1. Table of pressures classified into types of human activities

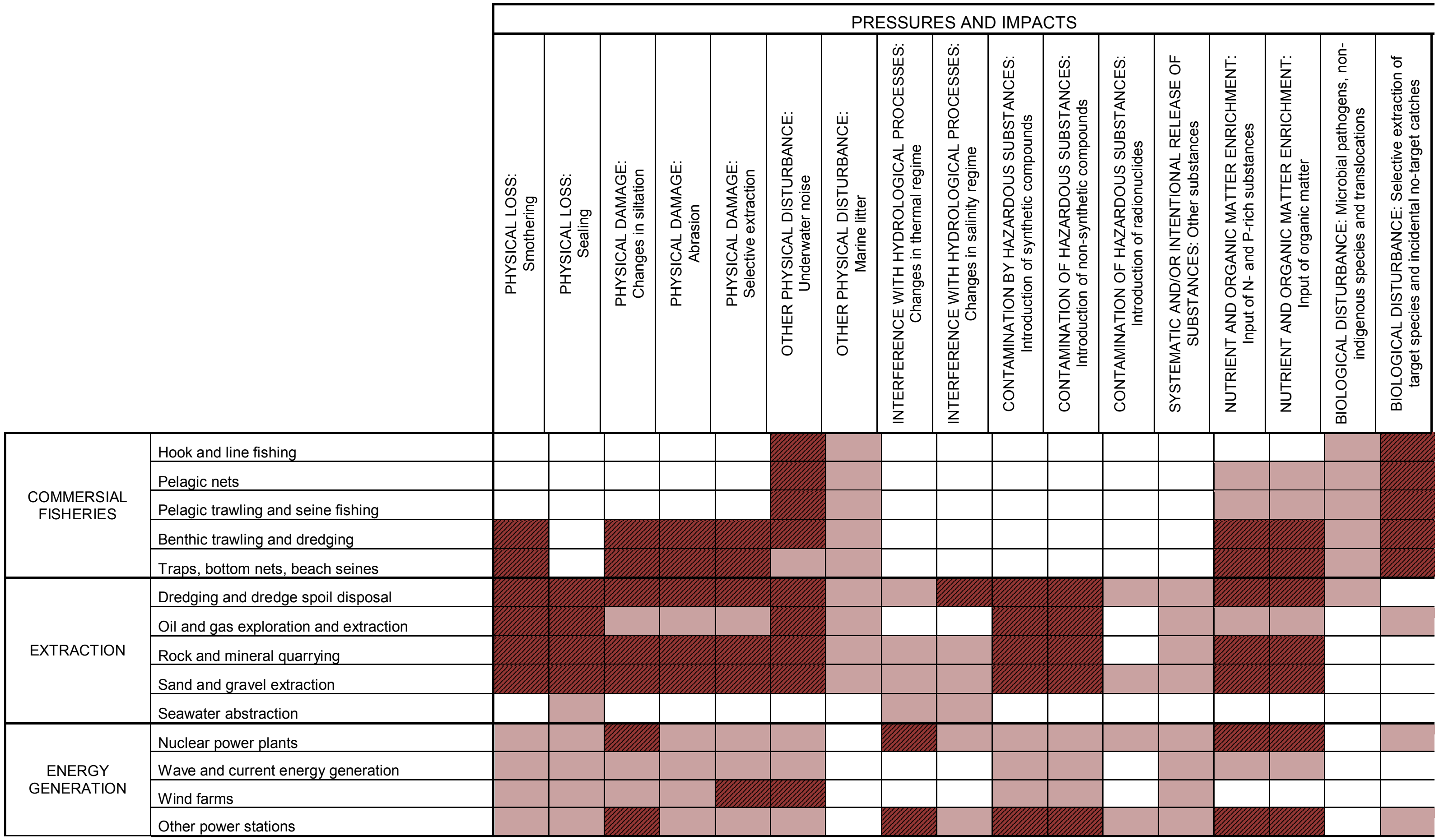


Figure 1 (cont.)

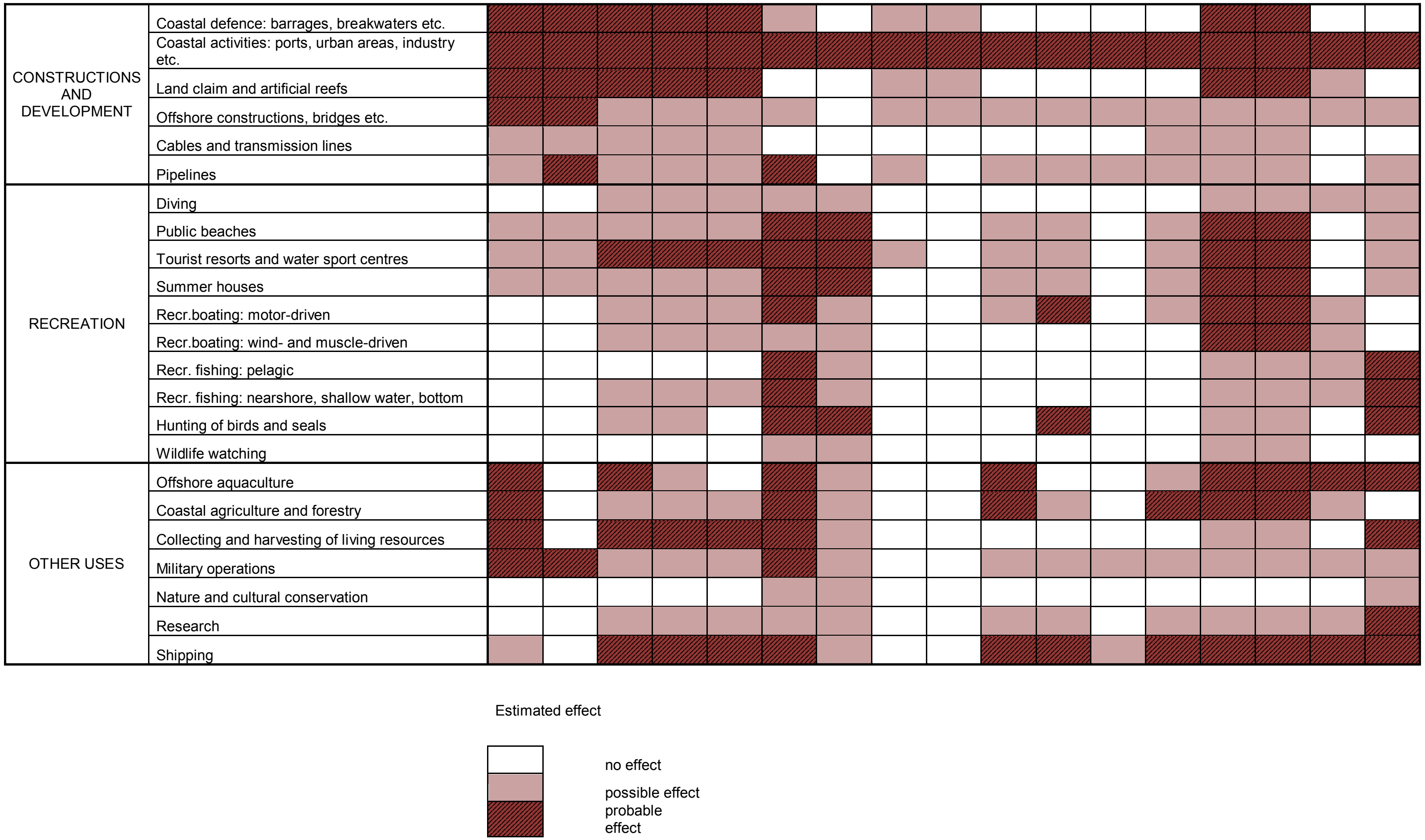


Figure 2. Table of pressures and their impacts on marine habitats and species

		HABITATS										SPECIES								
		1110 Sandbanks which are slightly covered by sea water all the time	1130 Estuaries	1140 Mudflats and sandflats not covered by seawater at low tide	1150 Coastal lagoons	1160 Large shallow inlets and bays	1170 Reefs	1180 Submarine structures made by leaking gases	1610 Baltic esker islands with sandy, rocky, and shingle beach vegetation and sublittoral vegetation	1620 Boreal Baltic islets and small islands	1650 Boreal Baltic narrow inlets	*Fucus vesiculosus*	*Furcellaria lumbricalis*	*Chara tomentosa*	*Zostera marina*	*Phragmites australis*	Bivalve molluscs	Clupeid fish	Birds	Marine mammals
COMMERSIAL FISHERIES	Hook and line fishing	○	○		○	○	○	○			○							○	○	○
	Pelagic nets	○	○		○	○	○	○			○				○			○	○	●
	Pelagic trawling and seine fishing	○	○			○	○	○			○	○	○		○		○	●	○	●
	Benthic trawling and dredging	●	●			●	●	●	●	●	□	●	●		●		●	●	○	●
	Traps, bottom nets, beach seines	○	○	●	●	○	○	○	○	○	○	●	●	●	●	●	●	●	○	●
EXTRACTION	Dredging and dredge spoil disposal	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●
	Oil and gas exploration and extraction	●	●		●	●	●	●	●	●	●	○	●		●		●	○	○	●
	Rock and mineral quarrying		●		●	●	●	●	●	●	●	●	●		●		●	○	○	●
	Sand and gravel extraction	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●	○	○	●
	Seawater abstraction	○	●	●	●	○	○	○	○	○	○	○	○	●	○	○	○	○		○
ENERGY GENERATION	Nuclear power plants	●	●	●	●	●	●	●	●		●	●	●	●	●	●	●	●	○	○
	Wave and current energy generation	●	●			●	●	●			□	○	○		○		○	○		○
	Wind farms	●	●	●	●	●	●	●	●	●	●	○	○	●	●	●	○	○	○	○
	Other power stations	●	●	●	●	●	●	●	●		●	●	●	●	●	●	●	●	○	○

Figure 2 (cont.)

CONSTRUCTIONS AND DEVELOPMENT	Coastal defence: barrages, breakwaters etc.	●	●	●	●	●	●	●	●		●	●	●	●	●	●	●	●	●	●
	Coastal activities: ports, urban areas, industry etc.	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●
	Land claim and artificial reefs	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●
	Offshore constructions, bridges etc.	●	●	●	●	●	●	●	●	●	●	●	●	●	●	○	○	○	○	○
	Cables and transmission lines	○	○	○	○	○	○	○	○	○	○	○	○	○	○	○	○	○		○
	Pipelines	●	●	●	●	●	●	●	●	●	●	○	○	●	●	○	●	○		○
RECREATION	Diving	○	○		○	○	○	●	○	○	○	○	○	●	○		○	○		○
	Public beaches		○		●	○	○		●		○	●	●	●	●	●	●	●	●	●
	Tourist resorts and water sport centres	●	●	●	●	●	●	●	●		●	●	●	●	●	●	●	●	●	●
	Summer houses	○	○	○	○	○	○	○	●		○	○	○	●	○	●	○	○	○	●
	Recr.boating: motor-driven	○	○	○	●	○	○	○	○	○	○	○	○	●	○	○	○	○	○	●
	Recr.boating: wind- and muscle-driven	○	○	○	○	○	○	○	○	○	○			○	○				○	○
	Recr. fishing: pelagic	○	○		○	○	○	○			○				○			○	○	○
	Recr. fishing: nearshore, shallow water, bottom	○	○	○	○	○	○	○	○	○	○	○	○	●	●	○	○	○	○	○
	Hunting of birds and seals		○	○	○	○	○		○	●	○			○	○	○	●	●	●	●
	Wildlife watching	○	○	○	○	○	○	○	○	○	○			○		○			○	○
OTHER USES	Offshore aquaculture	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●	○	○	○
	Coastal agriculture and forestry	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●	○
	Collecting and harvesting of living resources	○	○	○	○	○	○	○	○	○	○	○	●	○	○	○	●		○	○
	Military operations	●	●	●	●	●	●	●	●	●	●	○	○	○	○	○	○	●	○	●
	Nature and cultural conservation	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●
	Research	○	○	○	○	○	○	○	○	○	○	○	○	○	○	○	○	○	○	○
	Shipping	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●

Distance of influence

- No effect / not relevant
- On-site/local
- Areal
- Sub-basin
- Baltic-wide

Intensity of impact

- No impact
- ○ Moderate
- ● Heavy

Figure 3. Table of conflicts and compatibilities among human activities

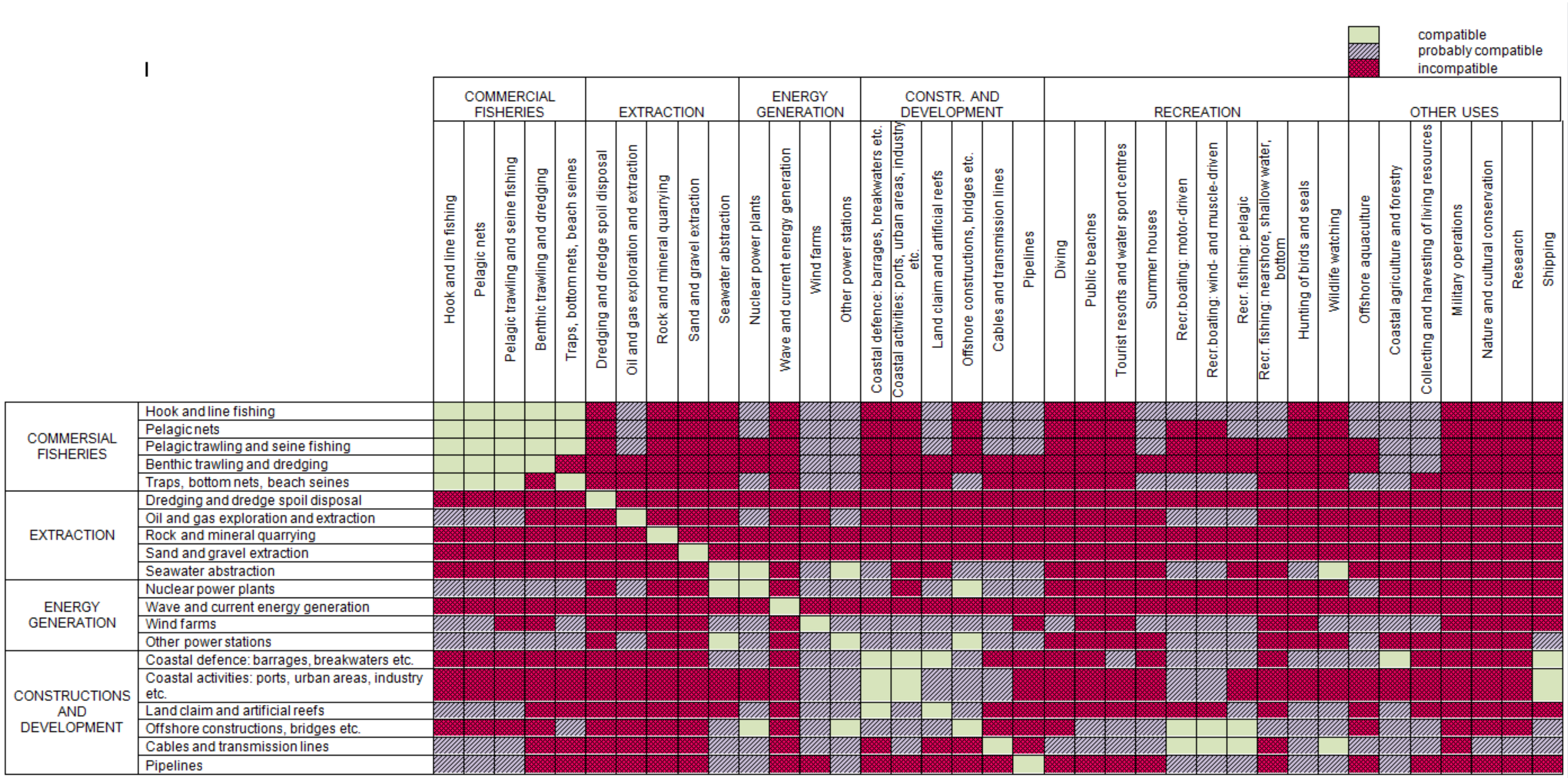
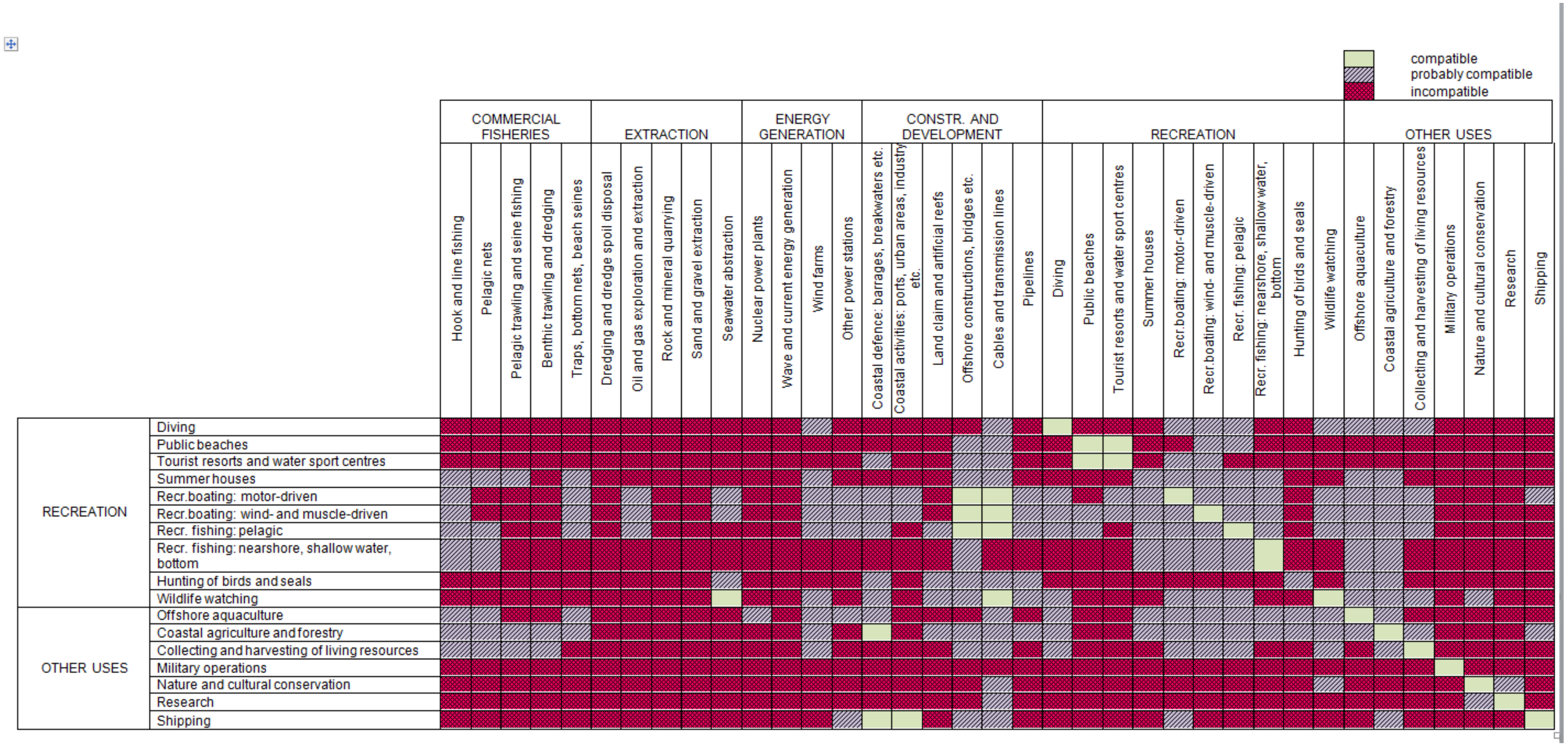


Figure 3 (cont.)



3. Harmonisation of the definitions of Annex I habitats

3.1 Introduction

Council Directive 92/43 EEC of 21 May 1992 or the Habitat Directive serves one of the most important legislative instruments, which provides a legal and common framework for the conservation of natural habitats. A draft list of habitat types listed in the Directive Annex I was based on the CORINE hierarchical classification of European habitats. Annex I listed 218 European natural habitat types, including 71 priority habitat types. In contrast to species list in the Annex II, the development of common agreed definitions appeared to be essential task for the different habitat types of Annex I. It is obvious, that understanding of habitat types in different biogeographic regions is crucial for development of consistent NATURA 2000 networks.

In the Baltic Sea, due to reduced salinity and its general effect on biodiversity, the number of biological criteria is decreasing from southern areas towards north. There are also substantial geological differences along the northwest – southeast gradient, which also influence the list of biological criteria to be applied for identification of habitat types. The northwestern part is primarily shaped by rocky shores and numerous islands and archipelagos protecting the coast, whereas the eastern side is exposed to wave action with typical sand and stony areas in the underwater slope. Due to such different environmental constraints countries report individual criteria or focus on different factors in characterization of Habitat Directive Annex I habitat types. On the other hand, in spite of diverse criteria on habitat type structure, countries typically share the same view concerning its functional properties. It is obvious, that the same habitat type contains different sub-types in different parts of the Baltic Sea, and hence both common as well as region–specific criteria have to be developed for different sub-types of the same habitat. It is not possible to develop the same criteria for the Quark area in the Bothnian Sea and Kattegat, however according to functional role and some basic physical characteristics different habitats in both of these areas may belong to the same habitat type.

Different official documents related to the Habitat Directive (e.g. Guidelines for the establishment of the Natura 2000 network in the marine environment 2007) recognise, that Member States will need to use different scales and apply expert judgment when appraising a habitat type at national level. However, in order to construct a scientifically sound and consistent NATURA 2000 network, habitat variation, which often correlates with the distribution of habitats, has to be considered and all major sub-types of habitats should be incorporated into network.

The aim of this report is to review current knowledge on diversity of selected habitat types in the Baltic Sea: reefs, sandbanks and lagoons, in order to support further development of criteria for identification of these habitat types. The case studies included in the report demonstrate the application of interpretations for selected habitat types in different areas of the Baltic Sea region.

3.2 Reefs (1170)

"Interpretation Manual of European Union Habitats" (2003) provides the following definition of reefs:

> Submarine, or exposed at low tide, rocky substrates and biogenic concretions, which arise from the sea floor in the sublittoral zone but may extend into the littoral zone where there is an uninterrupted zonation of plant and animal communities. These reefs generally support a zonation of benthic communities of algae and animals species including concretions, encrustations and corallogenic concretions.
>
> In northern Baltic areas, the upper shallow water filamentous algal-zone with great annual succession is normally well developed on gently sloping shores. *Fucus vesiculosus* is submerged at depth of 0.5-6 m in the sublittoral zone. A red algae zone occurs below the Fucus zone at depths of about 5 to 10 m.

The following species and groups of organisms have been used to demonstrate reef fauna and flora (Interpretation Manual of European Union Habitats, 2003):

Plants: Brown algae (species of the Fucus, Laminaria and Cystoseira genus, *Pilayella littoralis*), red algae (e.g. species of the Corallinaceae, Ceramiceae and Rhodomelaceae families), green algae. Other plant species: *Dictyota dichotoma, Padina pavonica, Halopteris scoparia, Laurencia obtusa, Hypnea musciformis, Dasycladus claveformis, Acetabularia mediterranea.*

Animals: mussel beds (on rocky substrates), invertebrate specialists of hard marine substrates (sponges, Bryozoa and cirripedian Crustacea for example).

Later on, the original definition of reefs was further specified in order to enable identification in offshore waters (Guidelines for the establishment of the Natura 2000 network in the marine environment 2007). This modification mainly resulted in extended list of substrate types, but also clarified other features.

Reefs can be either biogenic concretions or of geogenic origin. They are hard compact substrata on solid and soft bottoms which arise from the sea floor in the sublittoral and littoral zone. Reefs may support a zonation of benthic communities of algae and animal species as well as concretions and corallogenic concretions.

Clarifications:

- "Hard compact substrata" are: rocks (including soft rock, e.g. chalk), boulders and cobbles (generally >64 mm in diameter)
- "Biogenic concretions" are defined as: concretions, encrustations, corallogenic concretions and bivalve mussel beds originating from dead or living animals, i.e. biogenic hard bottoms which supply habitats for epibiotic species
- "Geogenic origin" means: reefs formed by non biogenic substrata
- "Arise from the sea floor" means: the reef is topographically distinct from the surrounding seafloor
- "Sublittoral and littoral zone" means: the reefs may extend from the sublittoral uninterrupted into the intertidal (littoral) zone or may only occur in the sublittoral zone, including deep water areas such as the bathyal
- Such hard substrata that are covered by a thin and mobile veneer of sediment are classed as reefs if the associated biota are dependent on the hard substratum rather than the overlying sediment
- Where an uninterrupted zonation of sublittoral and littoral communities exist, the integrity of the ecological unit should be respected in the selection of sites
- A variety of subtidal topographic features are included in this habitat complex such as: Hydrothermal vent habitats, sea mounts, vertical rock walls, horizontal ledges, overhangs, pinnacles, gullies, ridges, sloping or flat bed rock, broken rock and boulder and cobble fields

Main criteria for a general guidance in defining reefs at the European scale are important when considering reefs in the Baltic Sea:

- submerged
- formed by hard compact substrate of non-biogenic or biogenic origin
- topographically distinct and raised above the surrounding seabed
- may support uninterrupted zonation of benthic communities

Biogenic reefs. Holt et al. (1998) define biogenic reefs as "solid, massive structures which are created by accumulations of organisms, usually rising from the seabed, or at least clearly forming a substantial discrete community or habitat which is very different from the surrounding seabed. The structure of the reef may be composed almost entirely of the reef building organism and its tubes or shells or it may to some degree be composed of sediments stones and shells bound together by the organisms." In many cases the following criteria have been defined to identify biogenic reefs due to frequent continuous gradation between communities which are clearly not reefs and those which clearly are (Holt et al., 1998): i) the unit should be substantial in size in order to disqualify nodule like aggregations (somewhat raised and generally of the order of a metre or two across as a minimum, and; ii) should create a substratum which is reasonably discrete and substantially different to the underlying or surrounding substratum, usually with much more available hard surfaces and crevices on and in which other flora and fauna can grow.

OSPAR (2008) defines mussel *Mytilus edulis* as binding agent of the substrate and habitat provider for many infaunal and epibiota species at high densities, which account for at least 30% cover. Three structural elements have been described for mussel beds (Suchanek, 1979), which seems to be applicable for the Baltic biogenic reefs:

- a physical matrix of living and dead shells
- a bottom layer of accumulated sediments, mussel faeces and pseudofaeces, organic detritus and shell debris
- an assemblage of associated flora and fauna

OSPAR description (OSPAR 2008) of *Modiolus modiolus* horse mussel beds indicates cover threshold of 30% or more on a range of substrata, from cobbles to muddy gravels and sands, where mussels tend to have a stabilising effect due to the production of byssal threads. In the North Sea dense Modiolus reefs are generally very stable in the long term and are consistently found in the same place over long time periods (Holt et al.

1998). Two distinct types of Modiolus reefs have been described (Holt et al. 1998): semi-infaunal and infaunal. Semi-infaunal reefs are characterized by large accumulations of faecal mud and shell build up visible on echosounders. The living mussels in this case form an irregularly clumped layer over the mound, with the largest individuals living with about two thirds of their length embedded in the deposit and small individuals find refuge amongst the byssal threads of the clumps of larger ones.

Infaunal reefs, usually occur on coarser grounds and in strong currents, where the mussels bind together banks of gravel and live virtually as nested infauna within the coarse deposit. They can form wave-like mounds or bioherms which in the Bay of Fundy have been reported as up to 3 m high and hundreds of meters long (Holt et al. 1998). In both cases Modiolus beds have been considered as biogenic reefs, if substantial mounds are formed due to retention of sediment and fecal faeces and pseudofaeces by threats of horse mussels.

Apart from the infauna, the Modiolus community in Strangford Lough has been described as consisting of mainly three components (Magorrian, 1996 cited from Holt et al. 1998), which probably applies to any Modiolus reef community: a) very dense aggregations of living and dead Modiolus shells which form the frame work in a single or multiple layers; b) rich community of free living and sessile epifauna and predators; c) very rich and diverse small community which seeks shelter in the crevices between the Modiolus shells and byssus threads and flourishes on its rich sediment.

Geogenic reefs. Understanding of geogenic reefs is more complicated than that of biogenic origin due to larger variation and complexity of physical structure. Generally these reefs are formed of solid compact substrate of non-biogenic origin, which arise from surrounding seabed. There is limited knowledge on which features of benthic life have to be considered as characteristic for geogenic reefs next to their sedimentary and topographic features. Numerous environmental features, e.g. boulder and cobble fields, are included in this habitat complex (see clarifications for definition above), however none of the biological features such as presence of rare/endangered species, relatively high density of species or characteristic/unique biological assemblages are mentioned to be important. Most of the characteristic species listed in the Directive are irrelevant for the Baltic, therefore they only provide a general guidance on type of characteristic organisms i.e. habitat engineers.

The Habitat Directive provides the list of species in Annex II, however many Baltic areas are naturally devoid of true marine species and many Annex II species due to naturally harsh environmental conditions. On the other hand, these areas still serve significant sites for maintenance of local biodiversity and ecosystem functioning. Therefore environmental features, which primarily determine high biological diversity and ecosystem scale processes should be selected as criteria for identification of the Baltic Sea reefs. In this context it should be stressed that boulder reefs in the Baltic Sea often are a habitat for epifauna and macroalgal vegetation. In some areas and depth intervals some of those species are so dominant that they are often referred to as habitat forming species.

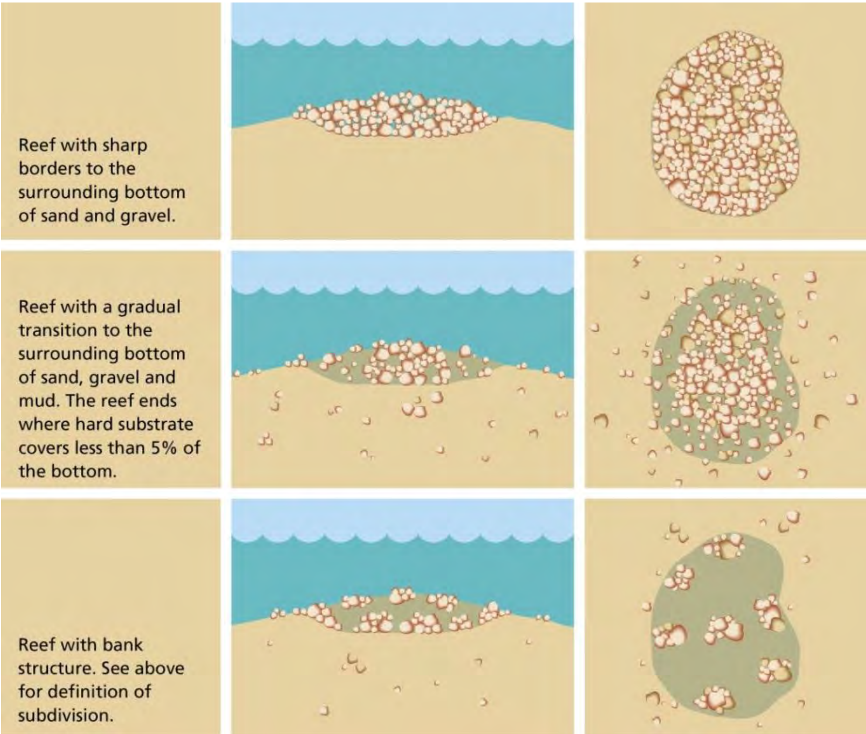
Differentiation of reef habitats into sub-features has been performed for Danish waters by dividing reefs into 6 different types based on depth, stability and group of structuring organisms (Dahl et al. 2004):

1. deep water stable reefs with structuralizing algae
2. deep water stable reefs with structuralizing fauna
3. shallow water stable reefs with structuralizing algae
4. shallow water stable reefs dominated by *Mytilus edulis*
5. shallow water unstable reefs
6. deep water biogenic reefs

Discrimination between deep and shallow reefs mainly accounts for light conditions, whereas the division of reefs into stable and unstable refers to substrate stability. Hard stable substrate is typically characterized by long living species and higher biomasses whereas unstable substrate is frequently colonised by opportunistic species, newly settled perennial species and in general lower biomasses.

When mapping is carried out the following question has to be answered: what is to be considered as a hard bottom and how much hard bottom is "needed" to form a reef? Although these issues were discussed among the EU member states preparing the interpretation manual no agreement was met, probably because the amount of reef and need for protection differed substantially among the member states. In Denmark such definition was developed for reefs made up by glacial deposits (boulder reefs). It includes a description of hard substrate and sets a minimum cover of hard stable substrate and a minimum size of the area (BOX 1)

BOX 1. Definition of reefs and sketch of different reef types and their borders to other types of seabed habitats



The left column shows a vertical cut and the right column shows the reef seen from above. From Dahl et al. (2003) and Dahl et al. (2004)

A reef is an area rising from the surrounding sea floor. The hard substrate made by pebbles, gravel, boulders, cliffs or biogenic concretions have to cover at least 5% of the sea bottom and the size of this area must be at least 10 m². If the reef is subdivided into smaller banks, i.e. composed of separated aggregations of hard substrate, the border of the reef is limited by a line around all subsection which each meets the requirements of 10 m² size and 5% cover of hard substrate. If the reef is sharply or gradually changing into a sandy or gravel dominated seabed, the border of the reef is defined by the cover of 5% hard substrate.

Hard substrate is defined as: Geological or biogenic material on the sea bottom with more than 10% of the surface covered by characteristic hard-substrate fauna and flora at least once a year.

3.2.1 Diversity of reefs in the Baltic Sea

In contrast to many other seas the Baltic contains only few species capable of forming biogenic substrate: horse mussels *Modiolus modiolus* and blue mussels *Mytilus edulis trossulus.*[1] Both have restricted distribution in the Kattegat, Belt Sea or south-western Baltic. Geogenic reefs, however, are relatively widespread and have a relatively complex structure defined by substrate, geomorphology and biological community (and/or dominant species).

In the western Baltic, the Belt Sea area and Kattegat the algal vegetation on geogenic reefs is multi-layered with different algal species forming the lower, mid and top vegetation layers at least in the depth interval down to 12–15 m (Figure 4). Additionally, changing salinity, physical stress and light climate often alter the species dominance from reef to reef at the same depth, therefore dominance level is highly variable. Below 15–20 m fauna elements dominate the biomasses, however a wide range of different combinations of dominant species may also occur. The communities are structured by a complex interaction of salinity, light and physical forcing including waves and current speed that are changing over a short spatial scale from Kattegat to the sills of Gedser-Dars and Drogden in Øresund.

Figure 4. Different macroalgal communities from 6–8 m depth in Kattegat

The reef Læsø Trindel in the northern part (left) and reef Briseis Flak in the southern part (right). Photo K. Dahl.

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[1] Relatively high densities and biomass of zebra mussels *Dreissena polymorpha* are found on soft bottoms in the Curonian lagoon (Daunys et al. 2006) and in the Szczecin Lagoon (Wolnomiejski, Wozniczka, 2008). On hard substrates the species is found in the Gulf of Riga and in the eastern Gulf of Finland (Orlova, Panov, 2004). So far it remains unclear if such habitats qualify for Annex I criteria.

In the south-western, central and northern Baltic, however, geogenic reefs are typically dominated by a relatively few species (frequently defined as habitat forming species) (Figure 5). Such reefs can be classified into sub-types according to the dominant species and substrate type, which determine general appearance of a habitat. Origin of a reef (biogenic or geogenic) is only relevant for classification of reefs in the Kattegat, Belt Sea and south-western Baltic, however high species diversity and generally low species dominance level will require other features (next to the origin and substrate type) to be used for division of reefs into sub-types.

Figure 5. Boulder reefs dominated by Polysiphonia fucoides (left) and Furcellaria lumbricalis (right) in the south-eastern Baltic Sea at 4-5 m depth (Palanga reef).

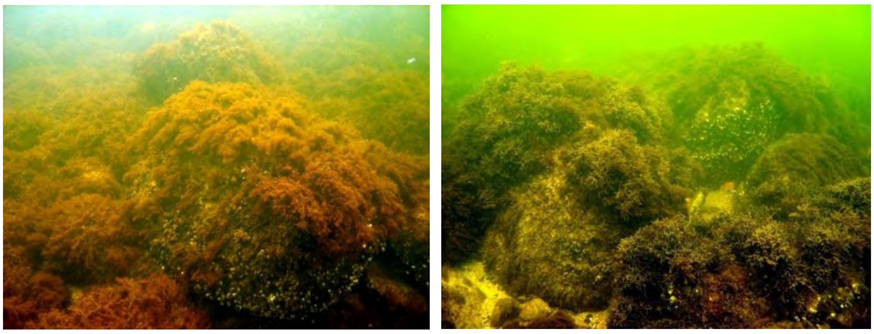


Photo M. Bucas.

Examples of sub-types of geogenic reefs common in the central and northern Baltic are listed below:

- *Mytilus edulis trossulus* on rocks – widespread in the north and north-western Baltic (along the Swedish and Finnish coasts)
- *Mytilus edulis trossulus* on hard clay (till) – recorded occasionally in the eastern Baltic (Lithuanian and Latvian coastal waters)
- *Mytilus edulis trossulus* on boulders and cobbles – widespread, Belt Sea, Arkona Basin, Eastern and Western Gotland Basin
- *Dreissena polymorpha* on boulders and cobbles – recorded locally, present in the Gulf of Riga
- *Dreissena polymorpha* beds – biogenic origin, present locally in enclosed water bodies with freshwater input (e.g. Curonian lagoon, Vistula lagoon)
- *Furcellaria lumbricalis* on boulders and cobbles – geogenic origin, distributed in the eastern Gotland Basin, Gulf of Riga (Kihnu Strait)

- *Fucus vesiculosus* on rocks – geogenic origin, widespread, Belt Sea waters, western Gotland Basin
- *Fucus vesiculosus* on boulders (geogenic origin) – Irbe Strait, Gulf of Riga, Gulf of Finland
- *Delleseria sanguinea* reefs – geogenic origin, distributed in the Belt Sea (Staberhuk, Fehmarn Belt, Sagas Bank), Eastern Gotland Basin

Sub-types with restricted distribution in the Baltic Sea will have lower variability in structure and function than those, which occur along the wider part of the Baltic salinity gradient (e.g. Mytilus or Fucus reefs). Therefore widespread sub-types in some cases are treated differently in different countries. For example blue mussels on rocks in Finnish coastal waters are not considered as fulfilling criteria of reefs, whereas boulder fields with mussels have been recognized as reefs in the eastern Baltic coastal waters (Latvia and Lithuania).

Some of listed sub-types may be a part of uninterrupted vertical zonation and occur in a distinct combination with other habitats. Typical zonation of macroalgae on rocky slopes is usually found with green algae dominating near the surface, brown algae in the shallow sublittoral part, and red algae at the greatest depths of the photic zone. So far, none of the countries reported reef as a zonation of habitat sub-types along the sublittoral slope, therefore sub-types based on different zonation patterns cannot be reviewed.

3.2.2 Geographic range

According to National summaries of the Article 17 reports reefs have been reported to be present in all Baltic Sea countries. According to Fact Sheet developed for "Reefs" by HELCOM HABITAT working group (Boedeker, 2007), this habitat type occurs in all HELCOM sub-regions of the Baltic Sea area. All countries reported reefs present in their EEZ.

In German waters reefs are mostly found in the areas of the Adlergrund, Rönnebank, the Kadet Channel and the Fehmarn Belt (Federal Agency for Nature Conservation, 2008). Other sources also indicate reefs in Kriegers Flak area (Zettler et al., 2006). All reported reefs are found outside the 12 nm zone. Denmark reported extensive areas covered by reefs within a total of 51 SAC's (Dahl et al., 2004). However there are more sites where reefs may occur and still most of the seabed in Danish waters is not properly mapped.

In the eastern Baltic, except Polish waters, reefs have been mapped in coastal waters only. Recorded reefs are distributed locally in the sublittoral, mainly along the stretches of erosion coasts, where they explicitly support zonation of benthic communities. There are no biogenic reefs reported for the eastern and northern Baltic, however beds formed by invasive zebra mussel present in river mouths (e.g. Nemunas river delta in the Curonian lagoon and Daugava mouth are in the Gulf of Riga) have not been discussed in a context of reef.

3.2.3 Physical features

The distribution of reefs in the northern Baltic Sea depends first of all on the presence of bed rocks, while in the southern parts, Danish Straits and most of Kattegat it depends mainly on erosional processes of glacial deposits and on hydrodynamic processes. The depth range of reefs depends on several interlinked processes, first of all presence of suitable substrate, and secondarily sedimentation and erosional processes. Salinity and light are additional modifying factors which play an important role for the structure and function of reef communities.

Shape. Two distinct shapes of reefs can be identified in the Baltic Sea: irregular shapes in the sublittoral slopes and elongated ridges or round shaped elevations completely or partly making up bank areas. The latter is more common in open waters.

Depth. Identified reefs are typically distributed in the depth range from 5 to 25 m, but deeper reef areas are also identified in Kattegat (Dahl et al. 2007) (Table 2.1). Biogenic reefs formed by horse mussels are found at depths below 15–18 m (Belt Sea area area), whereas biogenic blue mussel beds usually occur in depths between 10 and 27 m (Belt Sea, Arkona Basin). Mussels on boulders and cobles form colonies down to 25 m depth, whereas at greater depth the density decreases. In most areas of the Baltic macroalgae communities develop in depths down to 15 m but in the central and western Baltic, the Belt Sea and Kattegat it occurs down to 20–25 m depth. In general, high resolution maps of seabeds at larger depths are rare so only few reefs have been identified in deep waters. However, hard substrates on reef-like seabed elevations are known to occur in the Eastern Gotland Basin in depths of 40–50 m and in Kattegat and likely also in the Belt Sea.

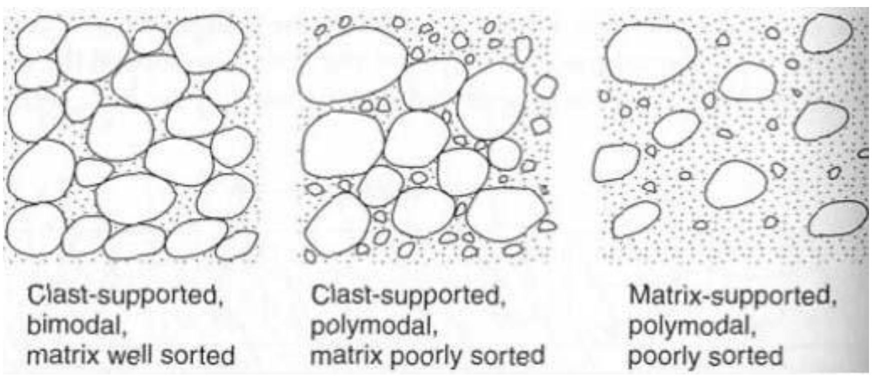
On the large scale, geogenic and topographically distinct reefs are typically formed on the elevated seabed, where depth differences from the top to the foot of the reef are between 1 and 20 m. Less frequently depth differences reaches 10 m (e.g. Walkyrien Ground, Kim's Top, Herthas Flak, Schultz's Grund, Møn's Klint) or even 2–6 m (e.g. Sagas Bank, Plantagenet Ground).

Substrate. According to German Federal Agency for Nature Conservation, reefs and reef-like structures in the German EEZ of the Baltic Sea occur mainly as boulder fields on moraine ridges and generally are described as hard mineral substrates such as rocks, till, or stones, primarily moraine ridges with block and stone cover in gravel/sandy surroundings. Reefs on partly mobile sediment should be classified as reefs if the associated biota is dependent on the rock rather than overlying sediment (Boedeker et al., 2006).

In Danish waters major parts of reefs consist of stable boulders and gravel (Dahl et al., 2004). Unstable substrates consisting of gravel and pebbles are found on most reefs and dominate a few reefs, therefore the 5% rule was fixed for hard substrate coverage to define a border between reef and non-reef areas (BOX 1) (Dahl et al., 2004).

In the eastern Gotland Basin and Gulf of Riga reefs are mainly found in boulder and cobble fields and hard substrate coverage threshold of 80% is well valid for identification of reefs at the spatial scale of 50–100 m or larger. Only locally, specific sub-types of reefs have been identified on hard glacial till steeply rising from the surrounding bottom by 3–5 m. Additional criteria of substrate composition and distribution has been discussed for "stony reefs" in the Eastern English Channel of the UK waters (Figure 7). Division of clast-supported and matrix-supported sea bottom types was based on drawing the limit between the bottoms where cobbles are touching each other and form consolidated seabed from those with finer and poorly sorted sediment in between (JNCC Report No. 432, 2009). Such distinction along with the average number of cobbles (within a still video image) and maximum clast size has been proved as unsatisfactory, however it is used for guiding during identification.

Figure 7. Interpretation of constituents of a "stony reef" (after Ceri James, cited from JNCC Report No. 432, 2009)



Exposure. Exposure is another important factor which influences both the distribution of reefs as well as type and diversity of reef communities. At the Baltic scale eight exposure classes have been modeled using Simplified Wave Model based on adjusted fetch and average wind speed data, which were ecologically proved for a number of coastal areas with complex coastlines (Wijkmark, Isaeus, 2010). Along the exposed eastern Baltic coasts fetch estimates based on seabed topography and wave orbital velocity have been proven as reliably describing the distribution of dominant reef species at the local scale (Muller-Karulis et al., 2007; Daunys et al., 2008).

Table 1. Dominant species and physical features of some reef sub-types present in the central and northern Baltic Sea

Dominant species	Substrate	Depth	Geoegraphic location	Salinity
Mytilus edulis Belt Sea	biogenic	5–25		~20
Mytilus edulis north and north-western Baltic	rock	5–25		
Mytilus edulis trossu-lus	hard clay (till)	15–20	Eastern Baltic	~7–8
Mytilus edulis trossulus	biogenic, boulders, cobble	10–25	South-western Baltic, eastern and western Gotland Basin: Hoburgs and Northern Midsjö Banks, Fehrmarn Belt, Kadet Trench; Gulf of Riga, Irbe strait, eastern Gulf of Finland	5–16
*Dreissena polymorpha	boulders, cobble	?	Gulf of Riga	~3–5
*Dreissena polymorpha	biogenic	<5–7	enclosed water bodies (e.g. Curonian lagoon)	<0,5
Furcellaria lumbricalis	boulders, pebble	2–10	eastern Gotland Basin, Gulf of Riga, Gulf of Finland	5–10
Fucus vesiculosus	rock	<6, max. cover at 1–2 m	mainly Swedish and Finnish coasts	>4
Fucus vesiculosus	boulders	<5	Gulf of Riga, Irbe Strait	~5
Fucus serratus	rock	>2	eastern and western Gotland Basin (mainly Swedish and Finnish coasts)	>7

Scale. In Danish waters a threshold of 10 m^2 is applied to determine minimum size of individual reef (Dahl et al., 2004). The same rule has been applied in the large habitat mapping program currently completed in Estonian, Latvian and Lithuanian coastal waters. Other sources related to the Baltic Sea do not provide habitat scale when reporting findings. OSPAR (2008) defines an area of at least $25m^2$ for a habitat to occur at a site, but this threshold may need to be higher in offshore areas due to limitations of surveys and sampling.

3.2.4 Biological features

Habitat forming and dominant species. In comparison to other seas there is relatively low diversity of benthic species which are structuring reef habitats in the Baltic. Only 2–3 macrofauna species (*Modiolus modiolus, Mytilus edulis, Dreissena polymorpha*) have been reported as forming biogenic reefs, while 5–6 macroalgae species (*Fucus vesiculosus, Delleseria sanguinea, Furcelaria lumbricalis, Laminaria sp*.) have been documented as habitat forming species on geogenic reefs. However, the low number of reported structuring species is partly attributed to the lack of habitat perspective in earlier studies.

Table 2. Biological features of selected Baltic reefs

Species	Fehmarn Belt1	Kadet Trench1	Kriegers Flak1	Adler Ground1	Slupsk Bank2	Palanga3
Salinity	15–23 psu	9–19 psu	7–15 psu	1–10 psu	1–10 psu	7–8 psu
Depth	15–25 m	15–30 m	8–35 m	7–35 m	8–20 m	4–20 m
Diversity	250 species	160 species	80 species	90 species	32 species	30 species
Abundance ind m-1	10.122	4.734	2.554	7.427	na	17.500
Characteristic benthic macrofauna species	Halisarca dujardini, Leucosolenia sp., Metridium senile, Sertularia cupressina, Musculus spp., Buccinum undatum, Lamellidoris muricata, Neptunea antiqua, Flabelligera affinis, Nicolea zostericola, Streptosyllis websteri, Idotea granulosa, Carcinus maenas, Callipallene brevirostris, Mucronella immer-sa, Psammechinus miliaris, Ciona intestinalis, Dendrodoa grossularia	Halichondria panicea, Metridium senile, Mytilus edulis, Nereimyra punctata, Balanus crenatus, Nymphon brevirostre, Eucratea loricata, Dendro-doa grossularia	Mytilus edulis, Balanus improvisus, Clava multicornis, Saduria entomon, Gammarus spp., Electra crustulenta	Mytilus edulis, Theodoxus fluviatilis, Balanus improvisus, Clava multicornis, Saduria entomon, Gammarus spp., Electra crustulenta	Harmothoe sarsi Hediste diversicolor Pygospio elegans Fabricia sabella Piscicola geometra Lochmanella falcata Balanus improvisus Idotea balthica Saduria entomon Jaera syei) Calliopius laeviusculus Gammarus zaddachi Gammarus salinus Gammarus oceanicus Mysis mixta Praunus inermis Hydrobia ulvae Theodoxus fluviatilis Embletonia pallida Mytilus edulis trossulus Electra crustulenta	Mytilus edulis Balanus improvisus Bathyporea pilosa Corophium lacustre Corophium volutator Electra crustulenta Fabricia sabella Gammarus locusta Gammarus salinus Gammarus zadachi Idothea baltica Jaera albifrons Leptocheirus pilosus Theodoxus fluviatilis
Charateristic benthic macroalgae species				Fucus sp., Chorda tomentosa,	Delesseria sanguinea Furcellaria lumbricalis Coccotylus truncatus, Phyllopho-ra brodiaei	Furcellaria lumbricalis, Polisyphonia sp.,

Data from: [1] Zettler and Gosselck (2006); [2] Andrulewicz et al. (2004); [3] Martin et al. (in prep.)

Spatial distribution is relatively well studied for some habitat forming species such as bladder wrack *Fucus vesiculosus* or blue mussels *Mytilus edulis*. This knowledge was transferred to simple species distribution maps based on presence or presence/absence data (e.g. Torn et al. 2006), which can be used for spatial prediction of potential reefs. However, species dominance level which is related to its capacity to play a role of habitat engineer needs to be modeled in order to understand distribution of habitat sub-types. Similarly, Mytilus can form biogenic reefs on soft bottoms or may have an important structuring role on hard bottoms in the south-western Baltic, whereas only later species life mode is realized in the central and northern Baltic. There is little known about factors which determine shift from soft bottoms to hard bottom life mode, and this gap considerably limits prediction of habitat sub-type distribution at the Baltic Sea.

Associated species. It is well recognized that reefs have typical associated species, occurrence of which is determined either by topographic features (e.g. lower depths), influence of reef dominant species (e.g. increase of geogenic substrate complexity, source of fecal material etc.) or substrate characteristic (complexity of the seabed supported by substrate composition). Composition and diversity of associated species is changing along the Baltic salinity gradient, however remains relatively stable in the south-western and central Baltic (Table 2.2).

3.2.5 Proposed definition of reefs and recommendations

The current definition is based on features common to sub-types of the central and northern Baltic reefs in order to cover diversity of this habitat type in the Baltic Sea. Baltic reefs are predominantly of geogenic origin and most often located in the coastal areas. However they may also occur in offshore areas and sometimes in association with the sandbanks. Existing data suggest that reefs can be a part of lagoons (Natura 2000 type 1150) and they are located within a number of Danish fjords as well (Natura 2000 type 1160).

Sublittoral topographically distinct area dominated by bedrock, boulders, cobble or biogenic substrate, which arise from the surrounding seafloor. The reefs generally support diverse and/or specific epifaunal and macroalgal vegetation communities. Typical depth range is from the sea surface and down to 25 m. Thresholds for hard substrate coverage vary in range from 5% in Danish waters up to 50% in the eastern Gotland Basin. Minimum reef area has been set in few studies earlier and varies between 10 and 25 m^2 in coastal waters. Reef communities in the upper photic zone (depths down to 15-25 m) are mostly dominated by habitat forming algae, whereas deeper reefs are predominantly shaped by epifaunal species. In all cases reefs have much higher species diversity than adjacent seabed, but also total number of species is higher in the southern Baltic reefs than in the northern ones.

Two examples of biogenic reefs and eight examples of geogenic reefs have been described from various sources as containing features characteristic for reefs of the central and northern Baltic Sea.

Sub-types of biogenic reefs:

- *Modiolus modiolus* reefs – southern Kattegat and the Belt Sea (Danish waters)
- *Mytilus edulis* beds – fjords connected to Kattegat, the Belt Sea (Walkyrien Ground)
- *Dreissena polymorpha* beds – present locally in enclosed water bodies with freshwater input (e.g. Curonian lagoon, Vistula lagoon)

Sub-types of geogenic reefs:

- *Mytilus edulis trossulus* on rocks – north and north-western Baltic
- *Mytilus edulis trossulus* on hard clay (till) – eastern Baltic (Lithuanian and Latvian coastal waters)
- *Mytilus edulis trossulus* on boulders and cobbles – Belt Sea, Arkona Basin, Eastern and Western Gotland Basin
- *Furcellaria lumbricalis* on boulders and cobbles – eastern Gotland Basin, Gulf of Riga (Kihnu Strait)
- *Fucus vesiculosus* on rocks – Belt Sea and western Gotland Basin
- *Fucus vesiculosus* on boulders – Irbe Strait, Gulf of Riga, Gulf of Finland
- *Delleseria sanguinea* reefs – Belt Sea (Staberhuk, Fehmarn Belt, Sagas Bank), Eastern Gotland Basin
- *Dreissena polymorpha* on boulders and cobbles – locally in the Gulf of Riga

3.3 Sandbanks which are slightly covered by sea water all the time (1110)

Interpretation Manual of European Union Habitats (2007) provided the following definition of sandbanks:

> Sandbanks are elevated, elongated, rounded or irregular topographic features, permanently submerged and predominantly surrounded by deeper water. They consist mainly of sandy sediments, but larger grain sizes, including boulders and cobbles, or smaller grain sizes including mud may also be present on a sandbank. Banks where sandy sediments occur in a layer over hard substrata are classed as sandbanks if the associated biota are dependent on the sand rather than on the underlying hard substrata.
>
> "Slightly covered by sea water all the time" means that above a sandbank the water depth is seldom more than 20 m below chart datum. Sandbanks can, however, extend beneath 20 m below chart datum. It can, therefore, be appropriate to include in designations such areas where they are part of the feature and host its biological assemblages.

Primarily sandbanks could be determined based on substrate and topography. Broad-scale data from e.g. geological seabed maps and bathymetric assessments can be used to locate potential sandbanks. Shallow parts of sandbanks can also be identified using aerial photographs (Ekebom and Erkkilä 2003) whereas SCUBA transect-technique is recommended to map sandbanks in more detail (Swedish Environmental Protection Agency 2004a).

Habitat types presented in the *Interpretation Manual of European Union Habitats* (2007) are frequently associated or overlap with other Annex I habitat types. Sandbanks (1110) could be separated from other habitat types according to the following criteria:

- *Estuaries* (1130) are separated from *sandbanks* because *estuaries* are influenced by freshwater. *Sandbanks* may be part of *estuaries*, although they are never mixed
- *Mudflats and sandflats not covered by seawater at low tide* (1140) are exposed at low tide, while *sandbanks* are permanently submerged
- *Sandbanks* (1110) in close connection to *Baltic esker islands with sandy, rocky and shingle beach vegetation and sublittoral vegetation* (1610) shall be treated as 1610

- Mussel banks shall generally assign to *reefs* (1170) and not to *sandbanks*
- *Sandbanks* (1110) may be part of *large shallow inlets and bays* (1160)
- *Coastal lagoons* (1150) and *boreal Baltic islets and small islands* (1620) are prioritised *sandbanks*
- Mobile shallow sand bars in exposed environments of the eastern Baltic Sea coastal waters (up to 8–10 m depth) are not classified as *sandbanks*

3.3.1 Diversity of sandbanks in the Baltic Sea

According to the physical environment the sandbanks in the Danish waters were divided into 3 sub-features (Dahl et al. 2004): i) non-exposed sandbanks in shallow water; ii) exposed sandbanks in shallow water; iii) sandbanks in deep water. At the scale of the Baltic Sea sandbanks have not been classified so far, however, presence of vegetation and origin can be useful next to exposure and depth factor. Following sub-types are therefore suggested, however will need further judgement by experts:

- sheltered sandbanks in shallow waters formed by land upheaval
- sheltered sandbanks in shallow waters dominated by vegetation
- sheltered sandbanks in shallow waters with scarce vegetation or without vegetation
- exposed sandbanks in shallow waters dominated by vegetation
- exposed sandbanks in shallow waters with scarce vegetation or without vegetation
- deep water sandbanks

Photic depth is likely to be the most feasible to discriminate between deep and shallow sandbanks, although 20 m depth limit has been mentioned (though criticised) as typical for sandbanks. For conservation purposes additional criteria of habitat homogeneity can be used in order to differentiate pure sandbanks from those mixed with other habitat types (e.g. reefs).

3.3.2 Geographic range

According to National summaries of the Article 17 reports sandbanks have been reported to be present in all the Baltic Sea countries. According to Fact Sheet developed for *sandbanks* by HELCOM HABITAT working group, this habitat type occurs in all HELCOM sub-regions of the Baltic Sea area. All countries reported sandbanks present in their Exclusive Economic Zone (EEZ). However, at later stages of evaluation and MPA designation Lithuania and Latvia have declared sandbanks in their coastal waters as of little value in respect to seabed vegetation and macroinvertebrates, as well as for fish and birds. In Denmark, 40 Natura 2000 sites have been designated solely or partly due to the occurrence of the sandbank habitat type (Dahl et al. 2004). Of these 40 sites, 18 do have existing data of coastal vegetation and benthic fauna and only 5 do have accompanying data on e.g. water quality.

Three offshore banks of the southern Baltic Sea are at least partially situated within the Polish EEZ (Andrulewicz and Wielgat 1999, Andrulewicz et al. 2004). Odra Bank is a sandbank of particular interest due to its large extension, habitat characteristic and high colonisation of macrozoobenthos. The Odra Bank is a shallow-water habitat (with average depth of 15 m and the most shallow parts on 7–8 m depth) comprised of sandy substrata (mainly fine sand, but also lots of shell gravels), which is situated in the northern part of the Pomeranian Bay (offshore in the border of Poland and Germany). The Slupsk Bank is situated in the southern Baltic proper (about 25 nautical miles north of the Polish coast) and consists of a diverse bottom habitat with sand, gravels, stones and boulders. In general, the Slupsk Bank is dominated by deposits of sand and gravel, although the north-western part of the bank, at depths of 10–20 m, consists of stones and boulders. The third important sandbank area of Poland is the Southern Midsjö Bank, which is located in the middle Baltic proper and shared with Sweden (an area surrounded by the Bornholm Basin, Eastern and Western Gotland Basins and the Slupsk Furrow to the south). This bank is quite deep (minimum depth of 14 m) and has a typical bottom comprised of sand and gravel.

In the Finnish and Swedish EEZ, sandbanks are distributed throughout the coastline and consist mainly of sand, even though other grain-sizes also occur (e.g. mud, gravel and boulders). In Sweden, large sandbanks are found within the southern region, around Öland and south of Gotland (e.g. Hoburg Bank as well as Northern and Southern Midsjö Bank in the middle Baltic proper), and along the coasts of Blekinge (Hanö Bay) and Skåne. The most northern sandbanks are found north of Luleå, among which the Nordström Ground and the Falken Ground are the largest. In Finland, sandbanks are mainly found along the southern and northern coasts and the largest sandbank area is found within the eastern part of the Archipelago Sea. In Finland, sandbanks are mainly results of land upheaval and therefore fairly specific in the context of other banks in the Baltic Sea.

3.3.3 Physical features

In general, sandbanks in the Baltic Sea are limited by various physical factors such as topography, degree of exposure, depth and salinity. Moreover, the structure and function of sandbanks are influenced by e.g. water quality, currents and waves, light penetration and sedimentation. Mutually, all these factors do have important role in determining dominant habitat and species composition.

Sediment type, substrate and degree of exposure: Sandbanks are primarily made up by sand grains (with a particle diameter of 0.06–2 mm), but do also consist of mud grains, cobbles, boulders or sediment of other grain sizes, although in smaller quantities. Banks where sandy sediment occurs as a layer over hard substrata are classed as sandbanks if the associated organisms are dependent on the sandy sediment rather than on the underlying hard substrata. The sediment may consist of glaciflu-vial deposits. According to German interpretation, sandbanks are differentiated from other habitat types by the dominance (in area) of the sandy substrate which is at least 40 cm in depth (Federal Agency for Nature Conservation, 2010). Eelgrass (*Zostera marina*) occurs mainly on exposed sandy bottoms where the sediment has a low organic content (1–3%), but may occasionally occur on sheltered soft bottoms with a higher content of organic sediment. Where bottom substrate is pure sand, macrophyte vegetation is scarce, but e.g. *Ruppia cirrhosa* and *Potamogeton pectinatus* can be found in these areas.

Depth range: Sandbanks are usually found at depths of 0–20 m, but can extend down to an approximate maximum depth of 36 m. *Zostera marina* (eelgrass) and *Ruppia* spp., two important key plants, are distributed from about 1 to 8 m depth with their main extension at about 3 to 5 m depth.

Topography: Morphology of submerged, sandbanks rather than grain size profile are emphasised in determining whether a submerged sandbank qualifies for the European Submerged Sandbanks dataset (Jones, 2001).

Sandbanks must be topographically distinct from the surrounding seabed. The sandbank height criteria used in the selection of banks for the European Submerged Sandbank Database was set to a minimum of 5 m with respect to the surrounding seabed. The slope, however, has not been determined as quantitative criteria and may vary considerably depending on type of a sandbank in the Baltic. More gentle slopes can be characteristic for sandbanks formed in the process of land upheaval (e.g. in Finnish waters) in comparison to open Baltic sandbanks.

Salinity range: Sandbanks are common at surface salinities with a maximum of about 12–14 psu in the southernmost Baltic proper to almost freshwater (about 2 psu) in the north part of Bothnian Bay. The main part of sandbanks is found in the salinity range of 3 to 7 psu. The northern limit of eelgrass coincides with the limit of 5 psu. Large eelgrass meadows are found in the Baltic proper where the surface salinity ranges from 6 to 8 psu.

Scale: No scale has been determined for sandbanks, which means that they can range from very small to the scale of a glacial moraine formation. However, as the variation of natural conditions occurs at a much larger scale in offshore environments compared to inshore coastal areas the scale to identify and define sandbanks is typically superior in offshore banks.

Table 3. Physical features of selected Baltic sandbanks

Sandbank area	Substrate	Depth	Topography	Salinity	Size	Reference
Kriegers flak [Southern Baltic Sea, off the coasts of Denmark, Sweden and Germany]	Mainly sand and gravel, but also moraine *	Min. depth: 8–15 m Max. depth: 35–37 m	Raised above surroun-ding bottom	~7–8 psu		Swedish Environmental Protection Agency (2006), Salinity data source: National centre for Environmental and Energy, Århus University/ Denmark
Odra Bank [Pomeranian Bay, off the coasts of Germany and Poland]	Well-sorted fine, medium and coarse sand	Min. depth: 7–8 m Max. depth: 20 m Mean depth: 15 m	Raised above surroun-ding bottom	~7–8 psu	approx. 480 km2 (35 km x 25 km)	Andrulewicz and Wielgat (1999), Zettler et al. (2006), Zettler and Gosselck (2006), Salinity data source: National centre for Environmental and Energy, Arhus University/ Denmark
Slupsk Bank [southern Baltic proper]	Dominated by deposits of sand and gravel. The bottom in north-western part comprises stones and boulders *	8–20 m	Raised above surroun-ding bottom	~7 psu		Andrulewicz and Wielgat (1999), Andrulewicz et al. (2004), Salinity data source: National centre for Environmental and Energy, Arhus University/ Denmark
Southern Middle Bank [central Baltic proper]	Sand and gravels	14–34 m	Raised above surroun-ding bottom	~7 psu	26 km2 (covering depths between 16 and 30 m)	Andrulewicz and Wielgat (1999), Swedish Environmental Protection Agency (2006). Salinity data source: National centre for Environmental and Energy, Arhus University/ Denmark
Northern Middle Bank [central Baltic proper]	Sand (moraine and glacial clay beneath)	15–36 m	Raised above surroun-ding bottom	~7 psu		Swedish Environmental Protection Agency (2006), Salinity data source: National centre for Environmental and Energy, Arhus University/ Denmark
Hoburg Bank [central Baltic proper, southeast of Gotland, Sweden]	Moraine and sandy sediment	15–35 m	Raised above surroun-ding bottom	~6–6.5 psu	122 673 ha – about 50% of the bank areal represent sandbanks (1110) and 50% reefs (1170)	Swedish Environmental Protection Agency (2006), Gotland Municipality (2005), Salinity data source: National centre for Environmental and Energy, Arhus University/ Denmark
Finn Ground/Eastern Bank [Offshore bank in the Bothnian Sea, Sweden]	Dominated by sorted moraine. The bottom in western part comprises sand and gravel *	Min. depth: 4–8 m Max. depth: 30 m	Raised above surroun-ding bottom	5.25–5.5 psu		Swedish Environmental Protection Agency (2006), Salinity data source: National centre for Environmental and Energy, Arhus University/ Denmark
Vänta Litets Grund [Offshore bank in the Bothnian Sea, Sweden]	Sorted moraine covered by sandy sediment	Max. depth: >30 m	Raised above surroun-ding bottom	5–5.5 psu		Swedish Environmental Protection Agency (2006), Salinity data source: National centre for Environmental and Energy, Arhus University/ Denmark
Marakallen [Offshore bank in the Bothnian Bay, Sweden]	Ice-river material and sandy sediment	Max. depth: 18 m	Raised above surroun-ding bottom	2.5–3 psu		Swedish Environmental Protection Agency (2006), Salinity data source: National centre for Environmental and Energy, Arhus University/ Denmark

* = Defined as both Sublittoral sandbanks (1110) and Reefs (1170)

3.3.4 Biological features

Submerged sandbanks are considered an important habitat type in the Baltic Sea, not least since they provide habitat for both soft- and rocky-bottom species. The banks could consist without vegetation or be covered by seagrass and/or macroalgae, which depend on water depth as well as on exposure to waves and currents. Banks far from the coast have good water exchange and do often function as refuges for marine species that have been forced away (due to e.g. competition) from the nearshore coastal zone. In addition, sandbanks provide spawning and nursery grounds for many fish species as well as functioning as wintering habitat for numerous bird species.

Three ecological subgroups have been identified for sandbanks according to the assessment of typical species (i.e. species functioning as indicators for conservation status): (1) sandy bottoms almost without vegetation but with a large versatility in the sediment, (2) eelgrass (*Zostera marina*) meadows and other macrophytes with less versatility in the sediment, and (3) mussel banks with more than 25% coverage.

Habitat-forming species: The Baltic Sea comprises a unique mix of marine and freshwater species adapted to contemporary brackish water conditions and a few true brackish-water species. Despite this, there is a limited biodiversity of animal and plant species compared to other seas. In general, the diversity is decreasing towards the northern and eastern parts of the Baltic as fewer species can thrive in the lower salinity levels. In the marine habitats of these low-saline areas freshwater species are dominating, especially in estuaries and coastal waters. For sandbanks, some examples of habitat-forming species are described below:

- *Zostera marina* (eelgrass) is an important species within the entire Baltic region and the dominant species on shallow sandbanks of the southern Baltic proper. Its northern limit coincides with the limit of 5 psu, i.e. close to the northern archipelago of Stockholm and Åland. The production season stretches from May to October. Eelgrass occurs predominantly on exposed sandy bottoms (at 2 to 6 m depths) while many *Chara*-species and macrophytes of limnic origin are the habitat-forming species of sheltered soft bottoms. The only exception to this is the eastern Baltic (which is the most exposed in the Baltic) where eelgrass does not occur in exposed areas. Occasionally small meadows of eelgrass are found within sheltered soft bottoms in other areas of the Baltic as well. Accordingly, eelgrass meadows in the Baltic proper differ from meadows in Skagerrak (e.g. on the Swedish west coast), which are mainly found on sheltered soft bottoms at depths of 1 to 4 m. The reason for this difference is not known, but it could be due to less competition from other plant species (like *Chara*-species and freshwater macrophytes), which seems to be the case in the Baltic Sea. The distribution of eelgrass along the southern and eastern Swedish coasts is not entirely mapped. The largest continuous meadows of eelgrass in the south of Sweden are likely those found from Sandhammaren to Västervik and in Kalmarsund along the west coast of Öland. In addition, large eelgrass meadows are also found near semi-exposed sandy beaches of Gotland. In Finland, eelgrass communities are found in the outer and middle archipelago of Åland, Archipelago Sea and Uusimaa area (from the town of Uusikaupunki on the west coast to the town of Sipoo on the southern coast)

- *Potamogeton pectinatus* is found on sand and mud bottoms of nutrient-rich waters. The production season stretches from June through August
- *Ruppia cirrhosa* is common on shallow mud bottoms of the Baltic proper, having a northern limit in the Bothnian Bay. The production season stretches from July to September
- *Zannichellia palustris* is found on shallow soft bottoms all along the Swedish and Finnish coasts. The production season stretches from July to September
- *Chara aspera* is an alga mainly found on sand and mud bottoms down to a few meters depth. In Shallow areas it may form dense meadows
- *Tolypella nidifica* is an alga mainly found on exposed sand and mud bottoms down to about 6 m depth

Characteristic species

- *Plants*: Potamogeton pectinatus, Ruppia cirrhosa, Zannichellia palustris, Zostera marina
- *Algae*: Chara aspera, Tolypella nidifica
- *Birds*: Clangula hyemalis, Melanitta nigra, Cepphus grylle
- *Fish*: Callionymus, Nerophis ophidion, Platichthys flesus, Pleuronectes platessa, Pomatoschistus microps, Pomatoschistus minutus, Syngnathus typhle
- *Invertebrates*: Sagartiogeton viduatus, Hediste diversicolor (Nereis diversicolor), Cerastoderma edule, Cerastoderma glaucum, Hinia reticulata, Macoma balthica, Mya arenaria, Crangon crangon, Saduria entomon, Astropecten irregularis, Brissopsis lyrifera

Typical species (i.e. species functioning as indicators for conservation status)

- *Plants*: Potamogeton filiformis, Potamogeton pectinatus, Ruppia cirrhosa, Ruppia maritima, Zannichellia palustris, Zostera marina
- *Algae*: Chara aspera, Chara baltica, Chara canescens, Chara tomentosa, Tolypella nidifica
- *Birds*: Clangula hyemalis (winter period), Melanitta nigra (winter period), Cepphus grille (winter period)
- *Fish*: Ammodytes marinus, Ammodytes tobianus, Anguilla anguilla, Clupea harengus, Gadus morhua, Gasterosteus aculeatus, Platichthys flesus, Pleuronectes platessa, Pomatoschistus microps, Pomatoschistus minutus, Psetta maxima, Sprattus sprattus
- *Chordates*: Branchiostoma lanceolatum
- *Echinoderms*: Astropecten irregularis, Echinocyamus pusillus, Psammechinus miliaris, Spatangus purpureus
- *Molluscs*: Acanthocardia echinata, Chamelea striatula, Pecten maximus, Spisula elliptica
- *Crustaceans*: Crangon crangon, Palaemon adspersus, Palaemon elegans
- *Cnidarians*: Virgularia mirabilis

3.4 Lagoons (1150)

"Interpretation Manual of European Union Habitats" (2003) provided following definition of lagoons:

> Lagoons are expanses of shallow coastal salt water, of varying salinity and water volume, wholly or partially separated from the sea by sand banks or shingle, or, less frequently, by rocks. Salinity may vary from brackish water to hypersalinity depending on rainfall, evaporation and through the addition of fresh seawater from storms, temporary flooding of the sea in winter or tidal exchange. With or without vegetation from *Ruppietea maritimae*, *Potametea*, *Zosteretea* or *Charetea* (CORINE 91: 23.21 or 23.22).
>
> - Flads and gloes, considered a Baltic variety of lagoons, are small, usually shallow, more or less delimited water bodies still connected to the sea or have been cut off from the sea very recently by land upheaval. Characterised by well-developed reedbeds and luxuriant submerged vegetation and having several morphological and botanical development stages in the process whereby sea becomes land
> - Salt basins and salt ponds may also be considered as lagoons, providing they had their origin on a transformed natural old lagoon or on a saltmarsh, and are characterised by a minor impact from exploitation

In the Habitats Directive lagoons are defined as priority habitat type, which can be separated from other associated Annex I habitats according to the following criteria:

- The essential difference between *estuaries* (1130) and *lagoons* is that lagoons have a sedimentary barrier that to some degree restricts water exchange with the surrounding sea. Frequently estuaries are considered as parts of rivers which come into contact with the sea, whereas lagoons are "pond- or lake-like" bodies of water (Oliver 2007)
- The presence of a barrier restricting water exchange between the surrounding sea to some degree also separates *lagoons* from the *large shallow inlets and bays* (1160)

For preliminary identification of lagoons, topographical maps at different scales can be used. However, identification of a sill can often be limited on available large scale topographical and bathymetric maps and therefore visiting the potential sites in the field is necessary. Identification of lagoons is also possible from aerial photographs (Numminen 1999). Different GIS techniques have been also applied in identifying lagoons, e.g. in Finland and Sweden, distance from the shoreline (<30 m), location of 5 m below average seawater level and the total area less than 30 ha were applied as criteria for mapping lagoons with GIS (Wennberg et al. 2008). However, the principles in delineating lagoons may differ to some extent between countries. In Germany presence of the sill is important qualifying feature and the habitat includes the waterbody as well as the shorelines with their reed swamps, tall herbaceous perennial vegetation, and pioneer communities (at mean water level) (Anon. 2007). In Finland the border of a lagoon is drawn where the mineral soil begins. The presence of typical vegetation is also one of the criteria that a lagoon needs to fulfill in order to be qualified as an Annex I habitat type (1150).

3.4.1 Diversity of lagoons in the Baltic Sea

Baltic lagoons are mainly differentiated according to the type of connection with the sea and the stage of geomorphological succession. Following sub-types of lagoons are mentioned in different Baltic Sea countries:

- classic lagoon types **–** former marine areas, which have been entirely or partially cut off from the sea by an isthmus formed by alluvial deposits from the sea
- estuarine lagoons – water bodies with freshwater inflow and separated from the sea by narrow sandy spit (e.g. Vistula Lagoon, Curonian Lagoon)
- boddens – distinct water bodies of the Pleistocene formed coast flooded during the Littorina transgression, currently with only a minor water exchange rate with the sea. If distinct through-flow of riverine water can be observed or seagrass meadows and a distinct water exchange rate with the Baltic Sea are present, such boddens are classified as estuaries or large shallow inlets and bays respectively (Boedeker, 2001)
- flads – small enclosed bays with narrow openings to the sea that are often becoming blocked by a growing sill. The flads in the Quark area are formed in the shallow depressions between the De Geer and Rogen moraines (Rinkineva and Molander 1997). Flads are unique to Finland and Sweden and usually are very shallow (less than 4–5 m depth) with no freshwater input. For a thorough description of flads see Munsterhjelm (1997)
- gloes – lakes formed by flads that have recently lost contact with the sea, but still have an intermittent saline water inflow during high water events. Flads and gloes form a successional series of different morphological stages. Gloe develops into a lake, when it loses its intermittent connection with the sea after land uplift raises the lake above the maximum high water level. Gloes have no freshwater input and are found only in Finland and Sweden (for a thorough description of gloes see Munsterhjelm 1997)
- salt basins and salt ponds (rare in the Baltic Sea) – considered as lagoons, if their origin is associated with a transformed natural old lagoon or a saltmarsh
- landlocked lagoons (coastal lakes) – former lagoons now completely cut off from the sea and salt-water incursion only occurs through permeation
- coves (classified in Denmark only) – water bodies enclosed from the sea behind spits of land, which still maintain an open connection to the sea (can be considered as small fjords)
- embanked areas (classified in Denmark only) – water exchange is often regulated by a sluice gate, only allowing saltwater incursion through leaks in the sluice and by permeation through the dam

3.4.2 Geographic range

According to National summaries of the Article 17 reports lagoons have been reported to be present in all Baltic Sea countries. According to HELCOM list of threatened and/or declining habitats/biotopes of the Baltic Sea area (HELCOM 2007) this habitat type occurs in all HELCOM sub-regions of the Baltic Sea area.

In Denmark lagoons are common especially in the Limfjorden area as well as on the coasts of Funen and Zealand islands (Pihl et al., 2001). These lagoons represent different stages of geomorphological succession and have been classified into four major successional types: coves, classical lagoons, landlocked lagoons and embanked areas (Pihl et al., 2001).

In Germany four large lagoons or lagoon complexes are reported along the Baltic coast: Saltzhaff, the Darß-Zingst Bodden, representing a chain of four connected lagoons, Großer Jasmunder Bodden (belonging to the Northern Rügener Boddens) and the Greifswalder Bodden (Eggert et al., 2005, Schiewer, 2008, Schumann et al., 2005). Lagoons are also present along the Polish coast (Pawlaczyk et al., 2004): Odra/Szczecin Lagoon and Vistula Lagoon, the later being shared with the Kaliningrad region of Russia. Both water bodies are within Natura 2000 sites (Pawlaczyk et al., 2004). There is also one smaller lagoon, Puck Lagoon comprising the inner part of the Puck Bay. In addition there are several coastal lakes in Poland that belong to this habitat type.

In Lithuania, the Curonian lagoon is the largest lagoon in the Baltic region. This lagoon is enclosed freshwater body separated from the sea by narrow sandy spit and its inner part is shared with Kaliningrad region of Russia. Due to large freshwater input lagoon has estuarine features and therefore is classified as estuarine lagoon. The part of the Curonian lagoon which belongs to Lithuania has been declared as NATURA 2000.

In Estonia lagoons are also present. For example, on the island of Hiiumaa there are lagoons Kirikulaht and Käinalaht and on the island of Saaremaa there are lagoons Laialepa laht and Suurlaht. The lagoon of Mõisalaht is on the mainland (Timm et al. 2007).

In Latvia the large fresh water bodies such as Lakes Pape, Liepajas, Engure, Babites, and Kaniera are ancient lagoons and they were cut off from the sea long ago. Currently these water bodies have been classified as lagoons (Aunins et al., 2010).

In Sweden and Finland lagoons (flads and gloes) are distributed throughout the coastline. Flads and gloes are particularly common along the Bothnian Bay (in the area of the Kvarken Archipelago), where the land uplift is the highest (8 mm per year), but also in the Archipelago Sea (Airaksinen & Karttunen 2001), in Åland and in the western part of Gulf of Finland (Uusimaa region).

3.4.3 Physical features

In general, lagoons are quite distinct in different parts of the Baltic Sea due to remarkable differences in geological history and hence in coastline physiography, topography, substrate and other features. However, a common physical feature of the lagoons across the Baltic Sea is the presence of a barrier between the lagoon and adjacent sea, which restricts water exchange. The barriers may be shaped by e.g. moving sand or gravel or by land upheaval. Although processes of a barrier development considerably differ between lagoon sub-types, typically lagoons have high residence time and relatively high rates of sedimentation and accumulation of organic matter. Most of the lagoons are flooded during storms and other high water events and are influenced by forces that shape the barriers between the lagoon and adjacent sea areas.

Substrate. In the southern Baltic Sea, lagoons are often found on wave-eroded sandy shores, therefore various sand fractions and silt are predominant on surface sediment. In contrast, in the northern Baltic Sea, lagoons are characteristic features of rocky archipelagos and are formed due to land uplift. Bottom substrate in such lagoons (primarily in gloes and flads) may partly or completely consist of stones, gravel, sand or bedrock.

Since lagoons are often more or less standing water bodies and the sill at the opening of the lagoon prohibits water movement to some extent, organic matter accumulates to the bottom of the lagoons. An extent of soft bottoms often gets higher towards inner parts of the lagoon and this may also lead to zonation of vegetation and changing communities.

Depth range. Lagoons are usually shallow, with their depth varying from less than 1 m to about 5 m. However, in some large lagoons maximum depths may exceed 10 m.

Topography. The lagoons are dynamic habitats over geological time scales and their topography is continuously changing due to freshwater inputs, interaction with the Baltic Sea or land uplift. Generally the lagoons in southern Baltic are topographically flat with gently sloping shores. In contrast, flads and gloes in the archipelago areas of the northern Baltic may be surrounded by steeper rocks.

Salinity range. The salinity of the Baltic lagoons varies to a large extent, both in space and in time. Depending on the exchange with the Baltic Sea, salinity in the lagoons may range from 17 psu (e.g. in the southern Baltic: Eggert et al. 2005) down to freshwater in the eastern Gotland Basin and northern Baltic. Stability of developing salinity gradients (if any) depends on nature of freshwater input and seawater inflows. Salinity of a flads and gloes is usually the same as in the outside waters, since they do not have major freshwater influx from rivers or streams. However, gloes may be almost freshwater in some cases.

Size. The size range of lagoons is highly variable in the Baltic: large coastal lagoons may have a surface area of up to several hundreds km^2 (the Curonian lagoon being 1584 km^2 in size) and small lagoons may cover only a few hectares. Gloes and flads are usually smaller than 25–30 ha.

3.4.4 Biological features

Lagoons are biodiversity hotspots providing an unique environment for aquatic and coastal flora and fauna. Due to dynamic nature of the lagoons, vegetation is in a continuous succession. Vegetation binds nutrients effectively and prohibits mixing of the sediments, therefore as a result, water might be clear despite the soft bottom. The luxuriant vegetation of the lagoons provides also habitat for many aquatic invertebrates, which in turn support wide spectra of food for fishes or birds. Being extremely heterogeneous lagoons provide high diversity of biotopes for spawning and nursery of juvenile fish as well as important feeding areas for many non-breeding and breeding waterbirds.

Biological diversity of lagoons is primarily shaped by geological history, which determines their physiographic features and degree of exposure to river basins and the Baltic Sea. If connected to the larger river basins and/or being of relatively low water exchange rates with the sea, lagoons are typically affected by eutrophicaton. On the other hand, flushing by seawater modifies eutrophication effects and creates physiological boundaries for many organisms. Therefore species diversity and composition in the Baltic lagoons is highly variable and can be generalized to the limited extent only.

Bottoms of the lagoons are often covered by abundant and characteristic macrophyte vegetation that includes plants e.g. *Ruppia maritime* and *Najas marina* and charophytes, e.g. *Chara* spp. and *Tolypella nidifica*. In addition to this, common species like *Potamogeton* and *Myriophyllum* are also found. Lagoons are often characterized by well-developed surrounding reed (e.g. *Phragmites australis*) stands.

Nine communities of rooted macrophytes have been defined in the German lagoons Salzhaff, Darss-Zingster Boddenkette, Rügensche Binnenbodden and Greifswalder Bodden (Selig et al. 2007a; 2007b): *Chara tomentosa, Chara aspera–Chara canescens,* Charophytes–*Ruppia cirrhosa, Ruppia cirrhosa, Charophyte-Zostera marina, Zostera marina, Zostera noltii-Ruppia cirrhosa, Najas marina, Myriophyllum-Potamogeton.*

Flads and gloes are characterized by a highly developed reed zone and lush submerged vegetation. Sequence of botanical types has been described for the southern Finland (but also present in Sweden) according to the developmental stages of the archipelago flad- and glo-type (Munsterhjelm 1997): *Chaetopmorpha linum, Vaucheria, Ceratophyllum demersum-Myriophyllum, Potamogeton pectinatus-Chara tomentosa, Chara tomentosa, Chara tomentosa-Najas marina, Najas marina, Najas marina-Ruppia maritime, Ruppia maritime, Chara aspera (paragymnophyllous)* and *v*egetation poor.

In the Curonian Lagoon the large littoral zone is covered by the macrophyte beds, dominated by *Potamogeton perfoliatus*, *P. pectinatus* and *Cladophora.* Typical dominant benthic taxa of this lagoon are unionids (*Unio tumidus*), oligochaets and chironomids, *Dreissena polymorpha*, *Marenzelleria neglacta* and Ponto-Caspian amhipods (*Chaetogammarus* and *Pontogammarus*).

Characteristic species:
Plants:

- Submerged plants: Najas marina, Potamogeton pectinatus, Potamogeton perfoliatus, Potamogeton filiformis and other Potamogeton species, Ruppia maritima, Zannichellia spp., Ceratophyllum demersum, Myriophyllum spp., Ranunculus baudotii, Callitriche hermafroditica, Lemna trisulca, Eleocharis acicularis
- Plants on the waterside: Subularia aquatic, Phragmites australis, Schoenoplectus lacustris, Typha angustifolia, Sagittaria sagittifolia, Butomus umbellatus

Algae:

- Charophytes as Chara tomentosa, Chara baltica, Chara aspera, Chara cansescens, Chara connivens, Tolypella nidifica
- In some lagoons can occur macroalgae as Fucus spp., Furcellaria lumbricalis, Cladophora aegagropila
- Birds: Podiceps cristatus, Bucephala clangula, Aythya fuligula, Grus grus, Phalacrocorax carbo, Ardea cinerea and other migratory species

- Fish: Clupea harengus membras, Lucioperca lucioperca, Esox lucius, Rutilus rutilus, Perca fluviatilis, Gasterosteus aculeatus, Pungitius pungitius, Phoxinus phoxinus, Abramis brama, Blicca bjoerkna, Scardinius erythrophthalmus and many migratory species as Salmon salar, Salmo trutta trutta, Osmerus eperlanus, Coregonus lavaretus, Lampetra fluviatilis, Petromyzon marinus and Anguilla anguilla

Invertebrates:

- Polychaetes: Marenzelleria neglecta., Nereis diversicolor
- Molluscs: Macoma baltica, Mya arenaria, Dreissena polymorpha, Mytilus edulis, Cerastoderma glaucum, Unionids, Hydrobia spp., Bithynia spp., Theodoxus fluviatilis, Radix spp
- Crustaceans: Gammarus spp., Asellus aquaticus, Idothea spp., mysids as Paramysis lacustris and Limnomysis benedeni, Balanus improvisus
- Insects: Chironomidae
- Cnidarians: e.g. Cordylophora caspia

3.4.5 Proposed definition of the habitat type for the Baltic Sea

The proposed definition is based on features common to all Baltic sub-types of lagoons in order to fully cover diversity of this habitat type in the Baltic Sea.

> Baltic lagoons are expanses of shallow coastal waters, wholly or partially separated from the sea by a barrier: sand spit, shingle, gravel or by rocks and bedrocks or by land upheaval. The salinity of the Baltic lagoons varies depending on the freshwater input and water exchange with the sea and may range from the brackish water to the freshwater. Topographically flat lagoons with gently sloping shores and surface area of up to several hundreds km^2 are typically situated in the south-western and south-eastern parts of the Baltic Sea, whereas small lagoons surrounded by steeper rocks and/or formed by land upheaval are covering less than 30 ha and are characteristic for the northern Baltic. Depth of the Baltic lagoons rarely exceeds 5 m.
>
> Baltic lagoons are mainly differentiated according to the type of connection with the sea and the stage of geomorphological succession. Following sub-types have been classified as lagoons in different Baltic Sea countries:

- classic lagoon types – former marine areas, which have been entirely or partially cut off from the sea by an isthmus formed by alluvial deposits from the sea
- estuarine lagoons – water bodies with freshwater inflow and separated from the sea by narrow sandy spit (e.g. Vistula Lagoon, Curonian Lagoon)
- boddens – distinct water bodies of the Pleistocene formed coast flooded during the Littorina transgression, currently with only a minor water exchange rate with the sea[1]
- flads – small enclosed bays with narrow openings to the sea that are often becoming blocked by a growing sill. Flads are unique to Finland and Sweden and usually are very shallow (less than 4-5 m depth) with no freshwater input
- gloes – are lakes formed by flads that have recently lost contact with the sea, but still have an intermittent saline water inflow during high water events. Flads and gloes form a successional series of different morphological stages. Gloe develops into a lake, when it loses its intermittent connection with the sea after land uplift raises the lake above the maximum high water level. Gloes have no freshwater input and are found only in Finland and Sweden
- salt basins and salt ponds (rare in the Baltic Sea) – considered as lagoons, if their origin is associated with a transformed natural old lagoon or a saltmarsh
- landlocked lagoons (coastal lakes) – former lagoons now completely cut off from the sea and salt-water incursion only occurs through permeation
- coves (classified in Denmark only) – water bodies enclosed from the sea behind spits of land, which still maintain an open connection to the sea
- embanked areas (classified in Denmark only) – water exchange is often regulated by a sluice gate, only allowing saltwater incursion through leaks in the sluice and by permeation through the dam

Development of measures and threshold values for individual sub-types of lagoons is needed in order to support a common understanding of this habitat type. Due to the high variability of the coastal benthic environment, the contribution of regional experts comprehensively covering different Baltic Sea regions will be of utmost importance in generating common “reefiness” measures.

Table 4. Physical features of selected Baltic lagoons

Lagoon	Area (km2)	Separation from the sea	Connection with the open sea	Mean depth (m)	Substrate	Salinity (psu)	Freshwater inflow	Reference
Curonian lagoon	1584	Sand spit	Enclosed, narrow inlet	3.8	Mainly sand and, silt	0–7	Nemunas	Gasiunaite et al. 2008
Vistula Lagoon	838	Sand spit	Enclosed, narrow inlet	2.6 (max 5.2)	sand, mud	0.5–6.5 (mean 3.5)	Pregola	Andrulewicz 1997, Chuba-renko 2008
Puck Lagoon (Inner Puck Bay)	104	Sand ridge	Enclosed, wide inlet	3 (max 8)	mainly coarse sand, also stony, muddy and peat	7.6	Reda	Andrulewicz 1997, Scymczak & Piekarek-Jankowska 2007
Odra/Szczecin Lagoon	635		Enclosed, narrow inlet	1–2	silt, sandy silt, silty sand and sand	4	Odra	Andrulewicz 1997
Greifswalder Bodden	510		Enclosed, wide inlet	5.8 (max 13.5)		3.2–11.2 (mean 7)	Ryck, Ziese	Selig et al. 2007b
Grosser Jasmunder Bodden	58.6		Enclosed, narrow inlet	4 (max 10)		7.0–9.6 (mean 8.2)		Eggert et al. 2006, Schiewer 2007
Saaler Bodden			Enclosed, narrow inlet	2.3 (max 9.5)		0.4–5.4 (mean 3.5)	Recknitz	Eggert et al. 2006, Selig et al. 2007b
Salzhaff	27	Sand ridge	Enclosed, narrow inlet	2 (max 10)		9.3–17.9 (mean 12.4)	Hellbach	Eggert et al. 2006, Selig et al. 2007b

4. Modelling key biological elements of reefs and other benthic habitats

When reporting the reference characteristics and status of the EU Habitat Directive's Annex 1 habitats, not only range and occurrence of these habitats, but also the status of their structure and function need to be evaluated.

Benthic algal vegetation and invertebrate communities play an important role, forming habitats for fish and fauna (structure) and as primary producer of organic material (function) for important part of the marine foodweb. For those reasons the state of the macroalgal vegetation and invertebrate communities is relevant for Annex 1 habitats having hard bottom seabed's like "reefs", "bobbling reefs and in some cases "shallow bays and lagoons."

In addition, macroalgal vegetation and invertebrate communities are mandatory biological quality elements (BQE) in assessing the quality of coastal waters according to the Water Framework Directive and finally they are also biological features explicitly mentioned in Annex III of the new EU Marine Strategy Framework Directive.

All in all, macroalgal vegetation and invertebrate communities form important elements assessing the quality status of marine ecosystems in the photic zone in both coastal and open waters.

Currently, in many countries, the factors and characteristics used to evaluate the structure and function of these habitats remain unspecified. This causes major difficulties in evaluating the reference conditions and conservation status of the habitats and also decreases the quality of the reporting. Therefore, one of the aims of this project has been to identify key factors that are important characteristics which can be used both to describe the functional status of different benthic habitats and to indicate state of the habitats.

Based on the available experience we anticipate that many of the key characteristics are related to the coverage of biological key elements like vegetation cover or cover of more specific habitat-forming species.

Mapping of habitat-forming species has so far largely relied on surveys and remotely sensed technologies without attempting to capitalize on the development of empirical and statistical spatial modelling techniques to extrapolate survey results to wider areas. During the past 10 years, participants in MOPODECO have been engaged in testing and applying empirical models as well as deterministic and GIS-based statistical models for use in estimation of coastal and offshore habitats. The results of these activities have been very encouraging in showing how spatial statistical models can be used as a cost-effective means for estimating the potential extent and state of habitat-forming benthic species of sub-merged vegetation and musselbeds. By developing these tools in co-operation across the Nordic and Baltic Sea countries, this project has helped to establish common data sets and harmonize the practices used in different countries. The project has provided the first attempt to estimate the coverage of habitat-forming species for a whole region on the basis of co-ordinated spatial models using state-of-the-art techniques and a combination of structural and dynamic parameters. The developed models should provide a useful showcase for the follow-up to the recommendations of the Nordic Forum on MPAs in Marine Spatial Planning (Blæsbjerg et al. 2009) and the work carried out on mapping and modelling of marine habitats in the Baltic Sea region under the Interreg III B project BALANCE (Hamdani & Reker 2007, Skov & Dinesen 2007).

Chapter 4.1 summarises the efforts to develop total macroalgal cover as a common metric of reef habitats in the Skagerrak and the Baltic Sea, which can be used as an indicator of both the reef and water quality over a wide geographic range. In chapter 4.2 the experiences from 4.1 have been applied to model the coverage of macroalgae in selected areas in the Kattegat for which detailed geo-morphological data are available using spatial statistical models. In chapter 4.3 the experiences using spatial presence only models to estimate the coverage of the red algae *Fucus vesicolus* in different regions of the Baltic Sea are detailed. In chapter 4.4 the experiences from using a deterministic model to estimate the coverage of blue mussel *Mytilus* are given. Finally, in chapter 4.5 the results of the modeling activities are summarized, especially with respect to the potentials and limitations for applying the models at the regional scale as a tool to provide statistics on the status and quality of the structure and functions of reefs and other benthic habitats.

4.1 Modelling total macroalgal cover

4.1.1 Monitoring submerged vegetation

The national marine monitoring programmes of the Nordic countries cover a wide geographic range from Skagerrak to the Baltic Sea. The vast majority of data is collected in fjords and shallow coastal areas. The monitoring of submerged vegetation in the Baltic Sea as well as the monitoring carried out in the Skagerrak does not follow a common standardized guideline. One could argue that the national monitoring programs can be allocated to two different "schools" each including a suite of different sampling methodologies. One school describes the vegetation cover according to the overall seabed (habitat description) and is carried out in many countries in the Cental Baltic Sea. This methodology often have information on the seabed composition given "side by side" with the information on cover of macroalgal and rooted plants like eelgrass. The other school used by Norway, Denmark and on the Swedish west coast on the other hand includes the seabed composition as a higher hierarchic layer and describe macroalgal vegetation as cover on only stable hard substrate and rooted plants as cover on soft sediments.

The basic difference with regard to macroalgal vegetation is that the "habitat description" methodology describes the average vegetation cover on all hard substrat types including unstable gravel and other substrate-types like mussels, whereas the methodology used by Norway, Denmark and west coast Sweden focus on vegetation on hard stable substrate only, ignoring what is growing on the unstable hard substrate. Within those two schools the sampling methodology vary with regard to use of line transects versus point investigations, use of smaller frames (<½ m^2) versus larger sampling areas, replicates versus no replicates, cover estimates in fixed classes, in percentage or abundance and use of in situ diver estimates or estimation of coverage based on photos or even sequences taken by drop video. A review of methodologies used can be seen in Moy et al. 2010 for Norway, part of Sweden, Finland and part of Denmark.

4.1.2 Factors controlling algal growth

The distribution and composition of macroalgal communities on hard bottom habitats depends on chemical, physical and biological factors controlling growth and mortality. The changing salinities from Skagerrak to the Botnian Bay are known to have a strong influence on the biodiversity in seaweed forests (Nielsen et al. 1995). Studies also show that salinity influence the ratio between opportunistic and per-annual algal species in the Danish coastal areas favouring the fast growing species in less saline waters (Carstensen et al., 2008). Light is another important factor that controls the macro algal cover. Light attenuation is caused by phytoplankton and other particles in the water, as well as dissolved organic matter and the water itself (Kirk 1994, Aksness et al. 2009). Drifting algae and particles sedimenting on the surface of the algae may also contribute to shading of the macroalgal community (May et al., 2008). Eutrophication increase the level of these light attenuation components thereby constituting a major pressure of macroalgal communities (Duarte 1995, Krause-Jensen et al. 2007, Dahl & Carstensen 2009). Investigations from Danish waters using larger datasets have documented positive relationships between nutrient and chlorophyll concentrations (Nielsen et al. 2002 and Carstensen et al. 2003). Reduction in water clarity as a result of increased chlorophyll concentration is also well documented (Petersen et al. 2005).

The stability of substrate is also an important factor controlling the species composition and cover on hard substrate. Stable substrate host a higher fraction of per-annual species cover, compared to unstable substrate at the same depth and locality. The community on unstable substrate is dominated by opportunistic algal species (Dahl et al., 2001). The size of the stones needed to form a stable substrate depends on exposure of the locality combined with the actual water depth were they are located.

Grassing by crustaceans and sea urchins effect the growth rates of algal species, cause mortality and overall reduce the vegetation cover. Grassing can develop to be the most important controlling factor. Devastating grassing due by mass occurrence of the northern sea urchin (*Strongylocentrotus droebachiensis*) changing seaweed forest to "barren grounds" is documented from several areas on the Northern hemisphere as reviewed by Steneck et al.. 2002, and it also an increasing issue on some Danish reef areas (Ærtebjerg et al. 2007). Lack of fish predation on sea-urchins due to overexploitation of fish stocks is probably the major cause of incidence of mass occurrence of Strongylocentrotus (Steneck et al. 2002).

In the southern part of the Danish Straits the salinity drops to a level were starfish cannot survive anymore and stop controlling the blue mussel population. Space composition between *Mytilus* and seaweed vegetation might be an important factor deducing the algal cover on hard substrate. Danish and German data collected east and west of the Gedser-Darss Sill indicate that this might be the case, but so far relationships between mussel and vegetation cover has not be proven in statistical analysis on Baltic Sea data.

Physical disturbance of the seabed caused by dredging fishing gears, and exploitation of stone and gravel or sediment spill from dredging operations are other anthropogenic disturbances that might interfer with the quality of hard bottom habitats. Chemical pollution might also have an effect on the algal cover e.g. from antifouling paint used on ships near harbour sites.

4.1.3 Algal indicators and international calibrations

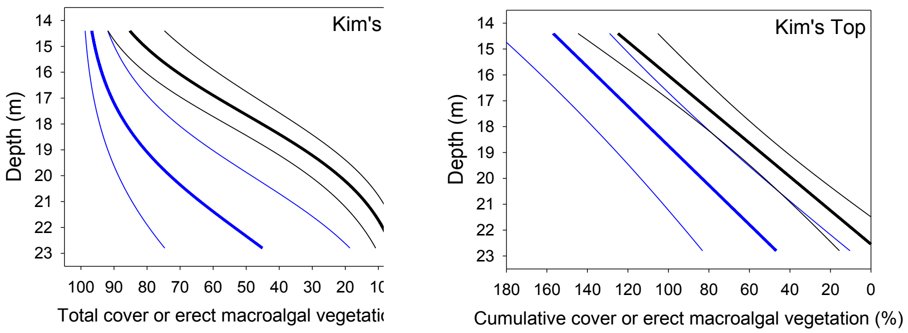
Problems with intercalibration of data describing macroalgal communities in Kattegat and Skagerrak between Norway, Sweden and Denmark were well documented in the NMR project RETRO (Reference conditions and EQOs for aquatic vegetation and macro-zoobenthos). In this case differences in algal communities was clearly demonstrated which could not be justified by natural or human impact alone (Petersen et al. 2006). The intercalibration work done so far in relation to the Water Framework Directive for the NE Atlantic and the Baltic coastal and transitional waters has focused on depth distribution of selected species. Based on historical data and expert judgement reference values of lower depth distribution limit for 8 selected macroalgae species has been set for defined water types of intercalibration. The species chosen are common in Skagerrak-Kattegat, are easily recognisable, and monitoring and historical data were available. But still, intercalibration has shown an urgent need for harmonizing monitoring methods and defining the lower depth limit. Even though the same species exists at Danish reef areas it was not possible to extract robust depth distribution data at all. Establishment of depth distribution borders has a weakness due to the fact that the methodology is based on finding something getting extinct. Search time and "stochastic" occurrences might play an important role defining the depth distribution now and in historic time.

In the Baltic Sea attempts have been done to intercalibrate bladderwrack (Fucus vesiculosus) between Sweden and Finland. This work is still in progress. Reference depth distribution of this Fucus species is based on literature surveys of historical data.

In Denmark other algal metrics (indicators) and methodologies have been developed and used to describe reference conditions and the present ecological status. Empirical models have been developed for NATURA-2000 reef sites in the open part of Kattegat describing "Total vegetation cover" and "cumulative vegetation cover" as a function of locality, solar radiation, depth, grassing pressure of sea-urchin and total load of nutrient to Kattegat (Dahl & Carstensen, 2008). Both models are statistically well founded. Lacking historic information on the vegetation cover in Danish waters the two models have been used to describe reference conditions and other scenarios of vegetation cover at different estimates of nitrogen loads to Kattegat, as seen in the example in Figure 8.

Empirical modelling on algal datasets collected on hard stable substrate in fjords and shallow coastal areas of Denmark has also been done recently (Carstensen et al., 2008). More or less all datasets collected as part of the Danish national monitoring program in 2001, 2003 and 2005 have been included in the work. Six different indicators were tested and important structuring factors identified and quantified. Common for all indicators are that data has been normalised for differences in sampling depth, spatial variation within a water body and time within the summer period by an "underlying model." The resulting algal indicators are expressed as an average value and variance representing 7 m water depth in each selected water body. Each of the six indicators and the factors that significantly structured the vegetation in the analysed dataset are shown in Table 5.

Figure 8. Total cover (left) and cumulative cover (right) of macro-phytes at different depth and at different nutrient load scenarios at the reef Kim's Top in the central part of Kattegat.



The thick blue line describes a reference load scenario with 10.000 tons from rivers and point sources in January–June and the thick black line describes an average load scenario equal to the period 1999–2007 on 48.000 tons in the same 6 month. The thin lines describe the upper and lower 95% confidence intervals on the estimated covers. From Dahl & Carstensen (2008).

Table 5. Macroalgal indicators and factors significantly structuring each indicator, as well as the overall model correlations

Variable	TN	Salinity	TN " Salinity	R2
Total algal cover		↑	↓	0.68
Cumulated algal cover		↑	↓	0.70
Cumulated cover of per-annual species	↓	↑		0.71
Cumulated cover of opportunistic species			↓	0.69
Fraction of opportunistic algalspecies	↑		↓	0.63
Number of per-annual species makroalgearter		↑	↓	0.79

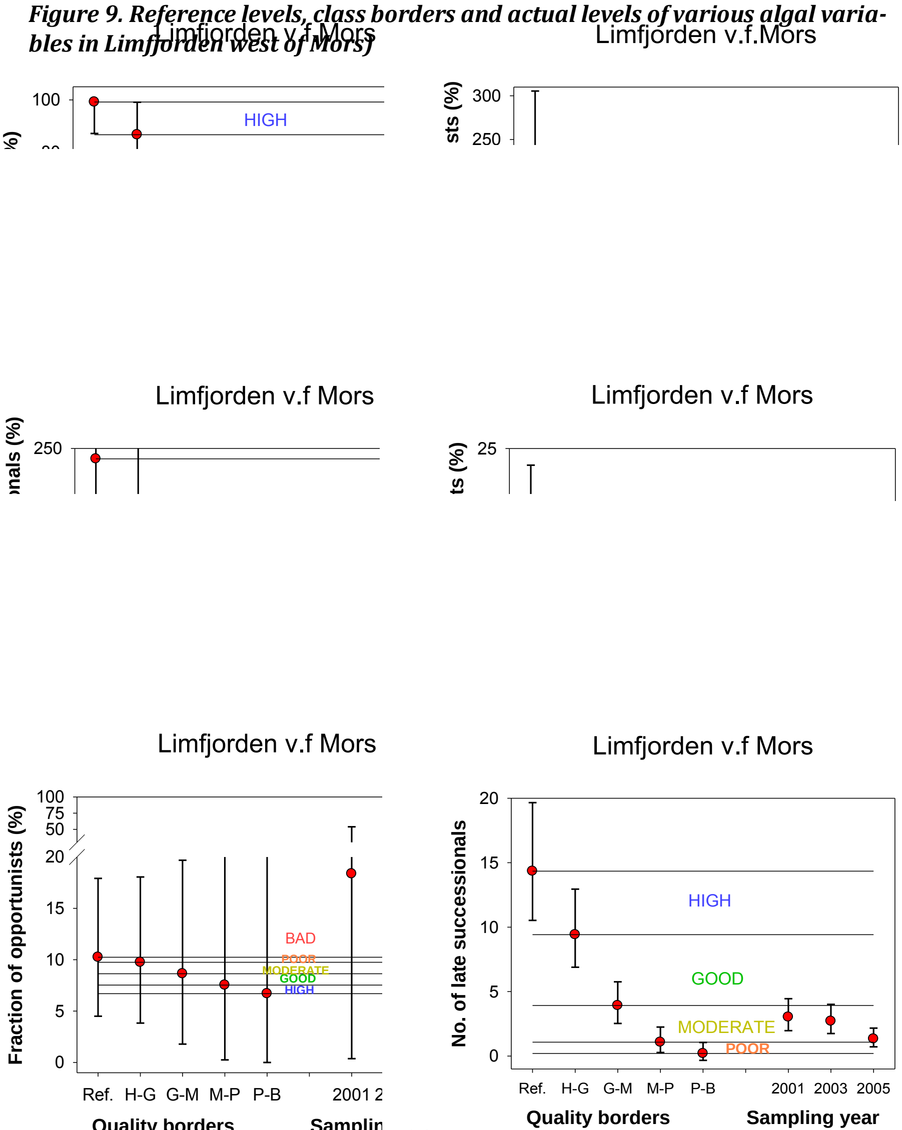
TN =Total Nitrogen TN*salinity means that the effect of TN was dependent on salinity. The arrows indicate positive (up) or negative (down) effects of TN and salinity (from Carstensen et al. 2008).

All macroalgal variables responded significantly ($p>0.05$) to a combination of changes in total nitrogen and to changes in salinity which emphasises the need for setting different targets depending on salinity. The strongest responses to changes in nitrogen concentration and the least variability were found for the indicators “total algal cover,” “number of late-successional species” and “fraction of opportunists” in less saline waters.

As was the case with reef vegetation in open waters no reference data is available for macroalgal vegetation in coastal Danish waters. Reference conditions for each algal indicators and ecological status class boundaries were established for all the macroalgal variables in a large number of waterbodies, considerably smaller than prescribed in the Water Framework Directive. The boundaries were established based on estimates on pristine load scenarios and site-specific relations between load and concentrations in the recipient waterbodies. Figure 9 gives an example of all indicators from the north-western part of Limfjorden, a water body with excellent datasets of both hydrography and algal stations.

The methodology of linking macroalgal covers to water chemistry used in Danish waters has also been tested on a dataset from Finland. Krause-Jensen et al. 2009 found that Finnish and Danish coastal monitoring data of cumulative cover (sum of all species-specific cover) had a similar functional relationship to Secchi depths. In this study we hypothesise that a common functional relationship of total cover to Secchi depths can be obtained across differences in the national monitoring programs.

Figure 9. Reference levels, class borders and actual levels of various algal variables in Limfjorden west of Mors)



Total algal cover (A), Cumulated algal cover (B), Cumulated cover of late-successionals (C), Cumulated cover of opportunists (D), fraction of opportunists (E) and number of late-successional algal species (F). Algal variables are modelled for a water depth of 7 m.

Recognizing a huge need for a common metrics for intercalibration of conservation status for reef habitats and ecological status according to the Water Frame Work Directive a first attempt to develop a common indicator across all the different national monitoring programs was made within the ALGAMONY project, financed by NMR (Moy et al. 2010). The Danish metric “total cover of erect macroalgal on hard stable substrate” was chosen as a possible additional variable that could relatively easily be sampled as part of the different national monitoring programs and then used to inter-calibrate among the participant countries. Attempts was also made to estimate "Total cover" data from existing time series of the Swedish and Finish Baltic monitoring programs (Moy et al. 2010).

The resulting dataset was quite heterogeneously distributed between study sites. The first step was to normalize the overall data observation of vegetation cover to a standard depth by an underlying model. The parameters for the change in vegetation by depth varied by a factor of 2 between countries, with the steepest gradients observed at the Swedish west coast (Table 6).

Table 6. Summary of model results to estimate annual total cover means for the different study sites

Study site	#annual total cover means	# of areas	# of years	Depth range (m)	Depth estimate (a)	Residual variance
Finland archipelago	40	6	7	3.0 – 11.3	-0.056 (±0.0120)	0.08214
Askö	20	1	20	3.0 – 25.6	-0.048 (±0.0023)	0.1703
Denmark stone reefs	155	24	15	6.0 – 24.0	-0.080 (±0.0024)	0.05063
Sweden W coast	6	6	1	5.0 – 20.0	-0.105 (±0.0024)	0.05966
Norway coast	18	9	2	5.0 – 30.0	-0.060 (±0.0019)	0.04333

The parameter estimate and the residual variance are estimated for the angular transformed observations. (from Moy 2010)

The uncertainty of the Finnish depth gradient was substantially larger than for the other study sites due to lack of deeper observations and therefore a limited range for estimating the depth gradient. The residual variance was also large at Askö. Estimation of "total cover" on existing data on individual cover data and estimation of suitable substrate for algal vegetation at Askö probably introduce additional variation. The residual variance for the Danish stone reefs, the Swedish west coast, and the Norwegian coast had similar magnitude.

The total cover in three areas could be described by following relationship to Secchi depth:

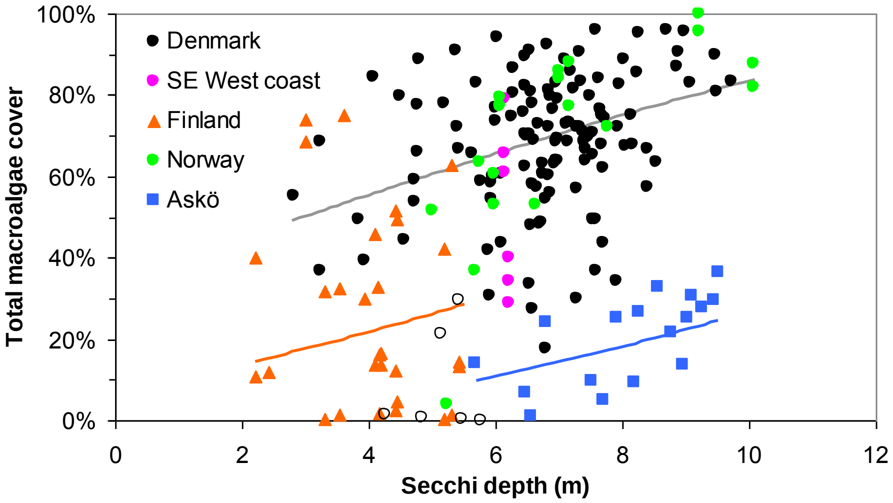
Skagerrak/Kattegat/Belt Sea: y=sin2(0.63+0.052x)
Askö area: y=sin2(0.28+0.052x)
Finnish archipelago: y=sin2(0.23+0.052x)

There was considerable scatter around the three estimated relationships between total cover and Secchi depths (Figure 10). The least scatter was found for the Askö data. This is most likely due to the problem of accurately characterising the water quality for the areas with macroalgae transects. For the Askö data there was only one area and therefore all scatter was principally temporal, i.e. uncertainty in the annual estimates.

For the other study sites the scatter also include spatial differences, i.e. differences in how well macroalgae transects are linked to water quality stations as well as the quality of data behind the estimated marginal mean values of water quality variables. Estimated marginal means on Secchi depth less than 5 m is for example hardly reflecting the average values from January to June in open inner Danish waters.

Comparable relationships for total macroalgae cover were obtained by substituting Secchi depths with total nitrogen (TN) or total phosphorus (TP) concentrations.

Figure 10. Estimated annual total cover at 15 m depth versus annual Secchi depths (Jan–Jun)



Gray solid line shows the combined model for Denmark, Norway and Swedish west coast, whereas separate models are shown for Finnish data and data from Askö. Open circles show annual means from Knudegrund and Lønstrup Rødgrund that were not included in the analysis. Total cover means and regression lines were found from the inverse angular transformation (From Moy 2010).

4.1.4 Aim

This study aims first of all to identify a key element of the algal vegetation, which can act as indicator of both the reef and water quality over a wide geographic range. At the same time we would like to identify and quantify the most important abiotic and biotic factors that structure the vegetation. We have chosen to focus our work on “total cover of erect algal vegetation.” This variable is integrative over time and available over a large depth range. At the same time total vegetation cover is highly relevant in an ecological perspective as the benthic macroalgal vegetation in general plays an important role in structuring hard bottom habitats and plays an important role as primary producer. We hypothesize that 1) high nutrient concentrations result in reduced macroalgal cover as a consequence of increased shading in the water colum and 2) that the large scale salinity gradient across the Baltic Sea as well as estuarine salinity gradients affect these relationships through positive effects of high salinity on water clarity, macroalgal diversity and eventually macroalgal cover.

4.1.5 Material and methods

Total cover of erect macroalgae

In this project we have focussed on data collected in the open part of the Norwegian south and south-western coast by Norwegian Institute For Water Research (NIVA), the open part of the Swedish west coast in Skagerrak by University of Gothenburg, all open inner parts of Danish waters by The National Environmental Research Institute, Aarhus University, Fjords and coastal areas inner parts of Danish fjords by the Danish counties (and later Environmental Centers), The Askö area near Stockholm by University of Stockholm, (Hoburgs Bank in the central Baltic Proper), the archipelago of Finland by Finnish Environment Institute and University of Helsinki, coastal areas of Estonia by the Estonian Marine Institute, University of Tartu, coastal areas of Lithuania by Klaipeda University, Lithuania and coastal areas of Germany by MariLim, Germany.

Total vegetation cover is not a mandatory variable collected in several national monitoring programs in the Baltic area. To achieve a common Baltic wide dataset describing this indicator some assumptions are needed and a set of estimation procedures has been applied as described below. Furthermore requirements of at least 10% hard bottom on the algal sampling sites have been used as a prerequisite for the joint dataset.

Danish data

The Danish monitoring programme has included a direct diver judgement of the percentage "total cover of erect macroalgal vegetation" of suitable stable hard substrate since 1993. This means that there exists a long time series on some of reef locations and more scattered information from a larger amount of other locations (table 3.1). Additional information of cover of sea-urchins, drifting alga mats and *Mytilus edulis* has also been available as pressure factors for the vegetation cover.

Norwegian data

The Norwegian hard bottom monitoring programme initiated sampling on total cover in 2007. This means that data exists for three years, 2007, 2008 and 2009 (Moy et al. 2010, Nordehaug et al. 2011). In some locations sampling has only been conducted one or two years. All data are given for hard stable substrate suitable for macroalgal vegetation. Locations severely affected by sea-urchin grassing have been omitted.

Swedish data

Data from the Swedish west coast in Kattegat-Skagerrak are restricted to 2007 and 2008 as "total cover" was not estimated in previous years. Total cover data is based on image analysis of a number of replicate photos taken of 0.25 m^2 of hard bottom areas. All data are given for hard stable substrate suitable for macroalgal vegetation.

Total algal cover on hard stable substrate from the Askö area on the Swedish east-coast has not been monitored. However total cover was estimated as the cumulative sum of individual species cover based on the experience (Krautsky pers. com.) that the vegetation is only slightly multilayered from water depth below 3 m (se later). The sum total vegetation cover on hard substrate was also estimated assuming different cover degrees on different substrate types based on experiences (Krautsky pers. com.). Assuming those approximations on total cover on suitable substrate resulted in dataset covering a long time span.

Finish data

The Finish dataset did not include "total cover" before the specific sampling in 2007, 2008 and 2009. It is however possible to estimate the "total cover" value for older data as the cumulative sum of individual species covers based on the experience that the vegetation is only slightly multilayered on water depth chosen for this analysis (Ari Ruuskanen pers com.). This assumption is identical to the one used for the Swedish Askö data. Registration of species specific vegetation cover has also changed from the overall seabed in within frames to suitable hard substrate in 1999. Total vegetation cover from the early period has to be recalculated based on the amount of hard substrate within the frame.

Estonian data

This dataset includes includes vegetation data from the period 1995–2008. Long time series were available from 5 locations. In the other locations rest sampling was made in one or two years. The Estonian total cover data include all vegetation, higher plants as well as macro algae, on the overall seabed. Total cover on hard stable substrate (TC) was estimated with the assumption that the sum of cover of stones, boulders, rock and limestone made up the suitable hard substrate:

*TC = (Total cover – higherplant/chara cover)*100/(stone%+boulder%+ rock/limestone*

Lithuanian data

In Lithuanian coastal waters total algal cover on hard stable substrate has not been monitored. However, total cover was estimated as the cumulative sum of individual species cover based on the experience (Bučas pers. com.) that the vegetation is only slightly multilayered from water depth below 3 m. Cobbles and boulders were considered as hard stable substrate at the exposed Lithuanian coast according to M. Bučas et al. (2007).

German data

Total cover of vegetation is a part of every site description/coverage estimation done in the German macrophyte monitoring program. Compared to the methods described above Germany follows the same procedure as Estonia. Total cover includes all vegetation parts of the overall seabed, higher plants as well as macro algae. Total cover on hard stable substrate (TC) was estimated with the assumption that the sum of cover of boulders, cobbles, pebbles, gravel and clay reefs made up the suitable hard substrate: it is not estimated substrate specific. Total cover on hard stable substrate (TC) was delivered to the project using following estimation procedure:

*TC = algal cover*100/(percentage cover of boulders, cobbles, pebbles, gravel and clay reefs)*

At stations with high mussel coverage, which is then the substrate for the macrophytes, this is used as the total hard substrate.

Selection of depth range

We focused the analysis exclusively on datasets from depth ranges where physical disturbance was no longer a major controlling factor for the algal vegetation. The coastward end of this depth range was estimated on Danish datasets (Carstensen et al. 2005 and Dahl & Carstensen, 2008) as the water depth with highest cumulative algal cover using non-parametric adjustment (LOESS, Cleveland 1979). This adjustment was made separately for a large number of areas and resulted in categorisation of the datasets in four exposure groups: weakly exposed areas where maximum cover was located at water depths of ~1 m, moderately exposed areas with maximum cover at water depths of ~3 m and more exposed coastal areas with maximum cover at water depths of ~5 m

(Carstensen et al. 2005) and open reef areas with a maximum cover of ~6 m (Dahl and Carstensen 2008).

The present dataset is restricted to total algal cover with a range from 0 to 100% and not cumulative cover of individual algal species that can exceed 100% by several folds. Fitting non-parametric spline functions describing total cover as a function of depth indicated that the exposure classes and depth ranges developed on Danish data were applicable to the larger dataset. The actual depth used to truncate the different national datasets is shown in Table 7.

Table 7. Minimum depth for data included in the analysis

Nationality	Depth
Norway	>5m
Sweden: Skagerrak	>5m
Baltic Sea	>3m
Denmark: Inner bays	>1m
Outer bays	>3m
Coastal waters	>5m
Reef areas in open waters	>6m
Germany	>5m
Finland	>3m
Estonia	>3m
Lithuania	>5m

Physio-chemical variables

Spatial variations in algal variables were related to the physio-chemical variables salinity, nutrient concentration, chlorophyll concentration and Secchi depth. The oceanographic stations were matched to nearby vegetation sampling stations based on expert judgement. In some case several vegetation stations refers to one hydrographical station like the Askö dataset. In other cases several hydrography datasets are used to characterize the water quality at one vegetation station.

Norwegian data

Norwegian physio-chemical dataset was collected from the oceanographic part of the Norwegian Coastal Monitoring Programme with 3 stations in Skagerrak and one station in the North Sea (Nordehaug et al. 2011). The sampling frequency was generally every second week, and sampling and chemical analyses were performed according to standard oceanographic guidelines.

Danish data

Danish algal data were matched with physio-chemical data stored in the National environmental database (MADS) at the National centre for Environment and Energy, Aarhus University (former NERI) and sampled by the former Danish counties, the newly established Environmental Centers, Aarhus University (Former NERI) as well as SMHI. Sampling and chemical analysis were performed according to common guidelines (Andersen et al. 2004). The typical sampling frequency range between weekly and monthly but some stations especially in the open waters has a lower frequency, especially on Secchi depth measurements that depends on daylight and suitable wind conditions.

Swedish data
Monthly measurements of hydrographical and water chemistry data from the National and the SMHI off-shore monitoring programs were retrieved from the SHARK database hosted by the Swedish Meteorological and Hydrological Institute. Sampling was conducted according to the national guideline: http://www.naturvardsverket.se/upload/02_tillstandet_i_miljon/Miljoovervakning/undersokn_typ/hav/vegbotva.pdf.

Finnish data
Finnish data sets were obtained from SYKE database. Samplings were carried out on permanent sampling stations nearby the algae monitoring stations. Water samples for phycio-chemical analyses were collected 1–2 times a month in January–March (if no ice occurred), and 2–4 times a month in April–August.

Estonian data
Estonian data were obtained from database of Estonian Marine Institute. Data were collected during the national monitoring programme. Nearest station matched with phytobenthos data stored were used. Water samples for phycio-chemical analyses were collected 1 times a month, sampling months vary in different years and stations.

Lithuanian data
Lithuanian physio-chemical dataset was collected from the oceanographic stations of the national monitoring programme in the Baltic Sea. The sampling frequency is four times per year and chemical analyses are performed according to standard guidelines.

German data
German physio-chemical dataset was collected from the monitoring programme of the state agencies for environment of Schleswig-Holstein and Mecklenburg Vorpommern with 50 stations along the German Baltic coastline. The sampling frequency is every month and sampling and chemical analyses are performed according to standard guidelines.

Data handling and harmonization

Data on Secchi depth and nutrient concentration show substantial and contra dictionary variation over the year as a result of the growth season of phytoplankton. A generic model was applied for water quality data taking into account variations between stations within the area (variable), months (January–June), and years of sampling. In Skagerrak, Kattegat and the Danish Straits longer time series existed for the water quality data and all these data were used to obtain better estimates of the seasonal variation (months) and variation between stations. The model employed was:

$$X_{ijkl} = station_i + year_j + month_k + e_{ijkl} \qquad (1)$$

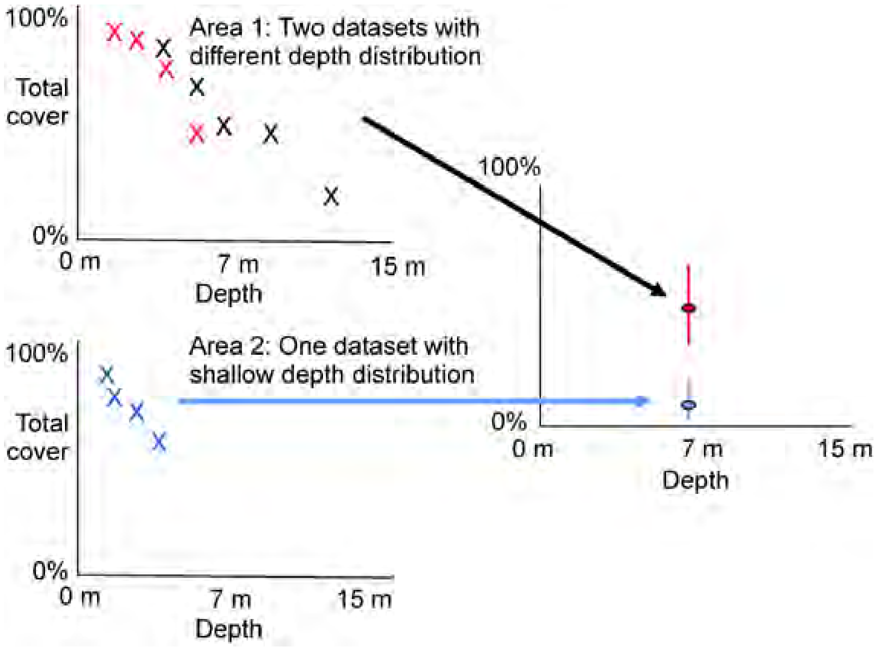
From this model marginal mean values of water quality data were estimated prior each sampling season on vegetation stations. Nutrient and chlorophyll-*a* concentrations were log-transformed prior to the analysis. For open waters the nutrients were estimated for depth 0– 15 m and salinity for depth 10– 20m, for inner waters the upper depth limit was relaxed to the surface. Annual means of water quality data are only shown in relation to macroalgae data. There were several cases were it was not possible to estimate marginal means due to insufficient data at appropriate depths and times.

Total cover exists from depths ranging from 1½ to 30m. In order to accommodate depth differences between transects and to group nearby transects when possible all data were transformed to a total cover at 7 and 15m water depth and a sea-urchin cover of near zero (cover =0,1%). This step facilitates a direct comparison between stations. This was done by an underlying model estimating the marginal means of total vegetation cover:

Arc-sin total cover = station + depth+ sea-urchin + year

As total cover values have fixed boundaries at 0 and 100, data were transformed by an arc-sin function prior to analysis. The model is described in details in Carstensen et al. 2008 and visualized in Figure 11. In this study it is further developed to account for differences in sea-urchine grassing as well. Total cover was expressed with a minor grassing effect in open water reefs in Kattegat represented by a sea-urchin cover of 0.1%.

Figure 11. Examples on the data harmonization procedure resulting in an estimated marginal mean value of the selected indicator for two selected areas with confidence level



In this example from Carstensen et al. (2009) a water depth of 7 m was chosen as output for further analysis.

Several vegetation transects were often collected within Danish fjords. In those cases marginal mean values were estimated for the fjord or part of the fjord (outer and inner). Lithuanian data were collected in grid pattern. In each case grids were considered as one sampling location, somewhat in parallel to some Danish fjords with several sampling transects. In all other cases marginal means of total vegetation were estimated on the given location.

The outcome of this exercise is a data set linking annual and area-specific total cover estimated for two chosen standard depths, 7 and 15 m with mean values of water quality variables in the 6 months period (January–June) prior to macroalgae monitoring in the summer season (typically between May and September). Data on vegetation for which it were not possible to estimate average values of salinity and total nitrogen or Secchi depth were discarded. An overview of the resulting vegetation dataset with is given in in Table 8.

Table 8. Sampling stations of macroalgal vegetation used in the analysis

Nation	Area	No. Locations	Depth range (m)	Data periode	Nr of sampling years	Main sampling month
Norway	Skagerrak	14	5–30	2007–2009	3	June and August
	North Sea	2	5–28	2007–2008	2	June
Denmark	Skagerrak/ Kattegat and Belt Sea area	30	6–24	1993–2009	17	June and August
	Adjacent fjords/ Western Baltic	18	1½–14	2001–2009	9	June and August
Sweden	Skagerrak	6	5–20	2007–2008	2	August
	Baltic Proper	34	3–26	1974–2008	21	June–September
Germany	W. of Gedser-Dars sill	10	5–7	2007–2009	3	July–September
	E. of Gedser-Dars sill	12	5–13	2006–2009	4	July–September
Lithuania	Baltic Proper	16 (*127)	5–19	2003–2006	3	May–October
Estonia	Baltic Proper	18	5–15	1995–2008	13	July–September
Finland	Baltic Proper /Gulf of Finl.	3	3–16	1998–2009	8	June–August

*Some sampling "stations" represent a mesh of sampling stations like in Lithuania. Number in brackets indicate the backgrounddata used and aggregated.

Based on preliminary assessment of algal cover the overall dataset was split in three sub-sets for further analysis. This was necessary because we would like to have predicted values of vegetation cover between 0 and 100% for the majority of the data as input for the analysis of the correlation to Secchi depth and water chemistry variables. A standardisation depth of 15m for cover predictions seemed to be an optimal choice for open water stations from the North Sea to the Belt Sea area. A depth of 7 meter seemed optimal for the Danish fjords and the eastern Baltic. At the same time we noticed from Danish and German data sets that the mussel *Mytilus edulis* became especially abundant at hard bottom habitats in open water east of the Gedser-Dars sill. Mytilus in those areas is most often entangled with the algal vegetation and this might interfere with the development of the vegetation cover. For this reason we chose the Gedser-Dars sill as a boundary for open water stations in the two sub-sets.

The three subsets of data include:

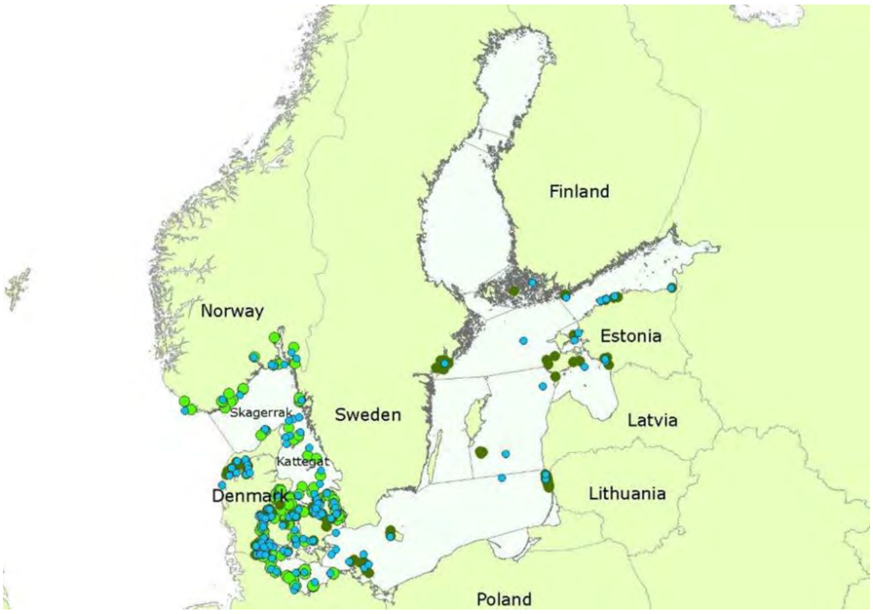
- North Sea-Skagerrak and Kattegat. This group represents surface salinity >12.2 psu, relative high water transparency (Secchi depth) and reduced presence of blue mussels (*Mytilus edulis*). Modelled macroalgal cover on 15m was used as predicted input for further analysis
- A subset of above data excluding German data, as these were found to behave differently from the rest (se below)

- Danish fjords and the majority of the Baltic Sea east of the Gedser-Dars sill. This group of data is characterised by low salinity and often deduced water transparency (Secchi depth). Blue mussels is often abundant. Modelled macroalgal cover on 7 m used as predicted output

The geographic distribution of sampling stations is visualized in figure 12.

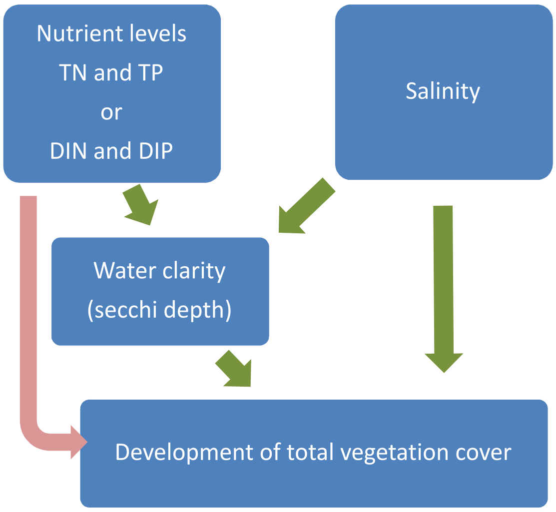
A suite of multiple regression analysis was carried out (table 9) in accordance to the conceptual model shown in Figure 13. Nutrients effects on water transparency expressed as Secchi depth were tested assuming an effect on the transparency by the nutrient effect on plankton production in the water column. Effects on total macroalgal vegetation of water transparency and salinity were tested as well as the indirect effects of nutrients on the vegetation cover. First we introduced all the potential independent variables in the regression, and then excluded variables one by one until only the significant variables remained. For predictive purposes random effects of country and location were left out of the model parameterisation. The nutrient data were all ln-transform in order to linearize the relationship to Secchi depth and algal cover.

Figure 12. Map of sampling stations



Light green circles represent aggregated macroalgae stations from open and more saline areas (group 1 and 2). Dark green circles represent aggregated macroalgae stations from low saline areas (group 3) Blue triangles represent oceanography stations.

Figure 13. Conceptual model for the GLM analysis



The green arrows represent investigated pathway of effects on nutrient and salinity on vegetation. The light red arrow illustrates an additional analyzed shortcut from nutrient to vegetation cover.

Table 9. The suite of regression models used is shown below. Country and locality are allowed to random variation. The initial models allows for combined effects of salinity and nutrients

Secchi depth as function of nutrients
E(Secchi) = b0 + b1ln(totN) + b2ln(totP) + b3salinity + b4(ln(totN)* salinity) + b5(ln(totP)* salinity) + B1locality+ B2country + e
E(Secchi) = b0 + b1ln(din) + b2ln(dip) + b3salinity + b4(ln(din)* salinity) + b5(ln(dip)* salinity) + B1locality+ B2country + e
Total algal cover as function of Secchi depth E(arsin(cover)) = b0 + b1Secchi + b2salinity + b3(Secchi*salinity) + B1locality+ B2country + e
Total algal cover as function of nutrients E(arsin(cover)) = b0 + b1ln(totN) + b2ln(totP) + b3salinity + b4(ln(totN)* salinity) + b5(ln(totP)* salinity) + B1locality+ B2country + e
E(arsin(cover)) = b0 + b1ln(din) + b2ln(dip) + b3salinity + b4(ln(din)* salinity) + b5(ln(din)* salinity) + B1locality+ B2country + e
Total algal cover as function of the nutrients found to control Secchi depth A special case of either model 4 or 5

4.1.6 Results

Water quality and vegetation cover from the North Sea to the Gedser-Dars sill in the western Baltic

This dataset cover a large timespan dominated by Danish Data from Kattegat and to some degree the Belt Sea area. Vegetation cover is expressed at 15 m depth and covers data with a salinity above 12.2 psu. Vegetation cover, nutrient concentration and Secchi depth measures all show distinct year to year changes within the period as well as variation between stations within each (Figure 11).

Relations between Secchi depth and nutrients (model 1+2)

In this area we found a significant and negative effect of TN on Secchi depth (model 1, P=<.0001). The Secchi depth model (1) could be reduced to:

Secchi depth (m) = 13.3307 – 2.7994 “ lnTN (µmol/l)

In both cases there were effects of both country and locality on the estimated Secchi depth.

Secchi was also found to correlate significantly with dissolved nutrients and salinity (model 2). In this case the Secchi depth could be described by dissolved nitrogen in interaction with salinity (p=0.1843 for DIN. DIN is included due to the following combined effect of DIN*salinity p=0.0354), dissolved phosphorous in interaction with salinity (p=0.0417 for DIP and p=0.0127 for DIP*salinity) and salinity (p=0.0160)

The parameter for the model with nutrients is:

Secchi depth (m) = 1.3366 – 2.1955 “lnDIP (µmol/l) + 0.09546 “lnDIP(µmol/l) “ Salinity (psu) + 0.7222 “ lnDIN(µmol/l) – 0.04581 “ lnDIN (µmol/l) “ salinity (psu) + 0.2011 “ salinity (psu)

Also in this case we found an effect of country as well a station.

Relationship between total vegetation cover and Secchi depth (model 3)

Total cover normalized to 15 m water depth was significantly strongly correlated with both Secchi depth (P=0.0004) and salinity (p=0.0059). Increasing values of salinity and Secchi depth both resulted in higher total covers.

The model was reduced to:

*Arcsin Total cover (%) = 0.3835 + 0.03962 " Secchi depth (m) + 0.01192*salinity (psu)*

Again, the total cover was dependent of both country as well as locality. The distribution of cover as function of Secchi depth and salinity is shown in figure xx and xx. Although there is a significant correlation between vegetation cover and both Secchi and salinity the scatter is substantial.

Relation between total vegetation cover and nutrients (model 4, 5 and 6)

In this case we found a significant relationship between total vegetation cover and total nitrogen (p=0.0022), salinity (p= 0.0417 and a combined effect of salinity and TN (p=0.0232) (model 4)

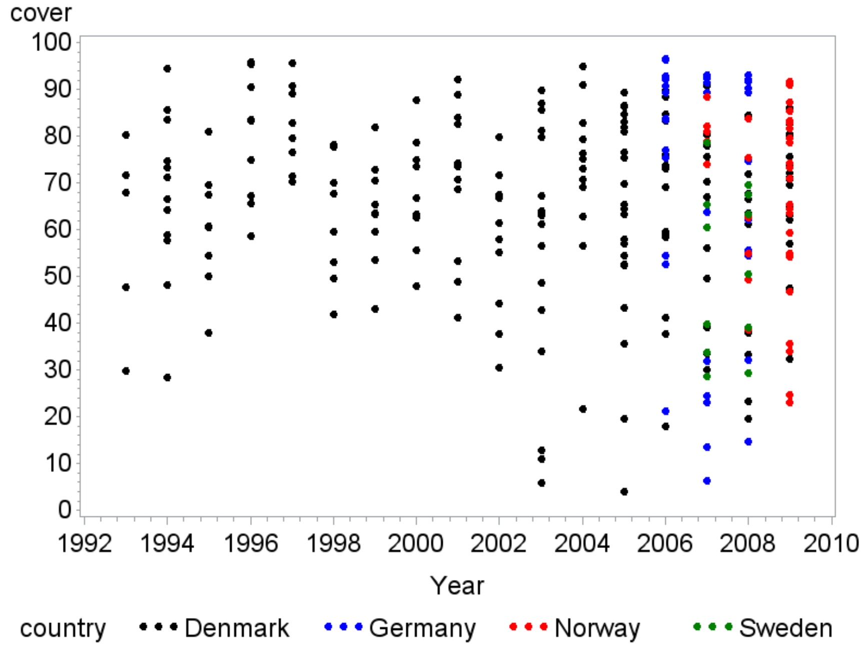
Arcsin Total cover (%) = 3.8741 -1.0935 " lnTN (µmol/l) – 0.03247 " lnTN (µmol/l) " salinity (psu) – 0.08463 " salinity (psu)

The distribution of cover as function of Secchi depth and salinity is shown in Figure 14. Country as well as locality did have an influence on the level on total cover. The correlation between total vegetation cover on one hand and nutrient concentration and salinity on the other (model 5) could be described by the following reduced model:

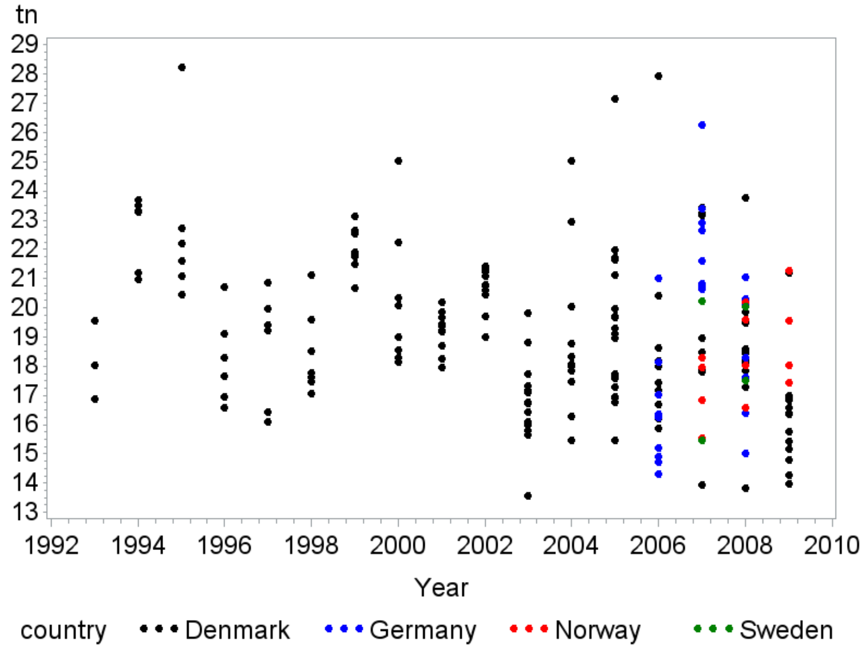
*Arcsin Total cover (%) =0.03968 – 0.03968*lnDIN (µmol/l) + 0.01110 " salinity (psu)*

The concentration of dissolved nitrogen had a significant negative effect on vegetation cover (p=0.0011) and salinity had a positive effect (p=0.0048). In this case we did not find an effect of country, only location. As we found the same nutrient variables controlling Secchi depth as was significant for the vegetation cover model 6 gave the same results as model 4.

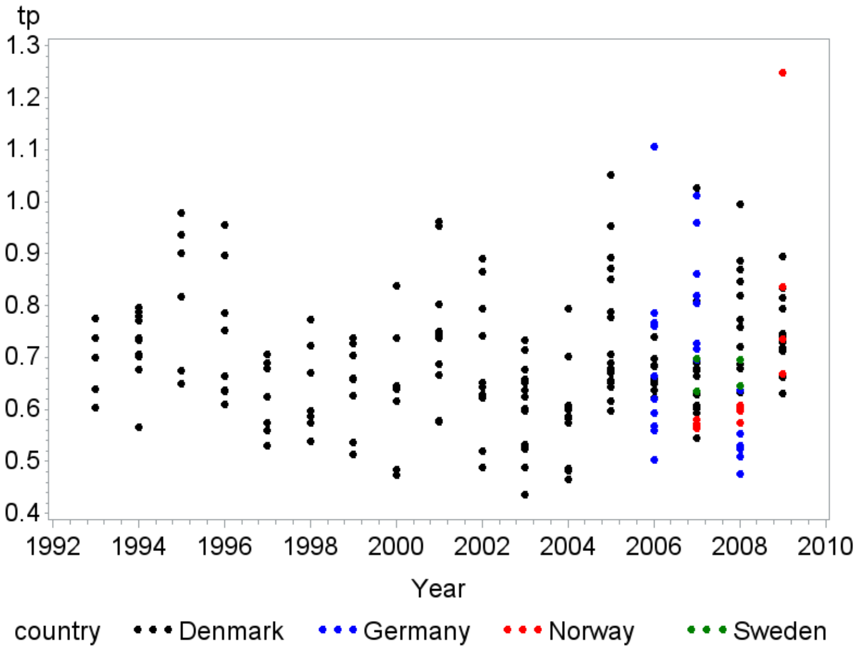
Figure 14. Vegetation cover from 1993 to 2009 back-transformed from arc-sin used in the model. Data from the North Sea to the Baltic Sea west of the Gedser-Dars sill



Total nitrogen concen-tration (µmol/l) from 1993 to 2009



Total phosphorous con-centration (µmol/l) from 1993 to 2009



Secchi depth measures in meter from 1993 to 2009

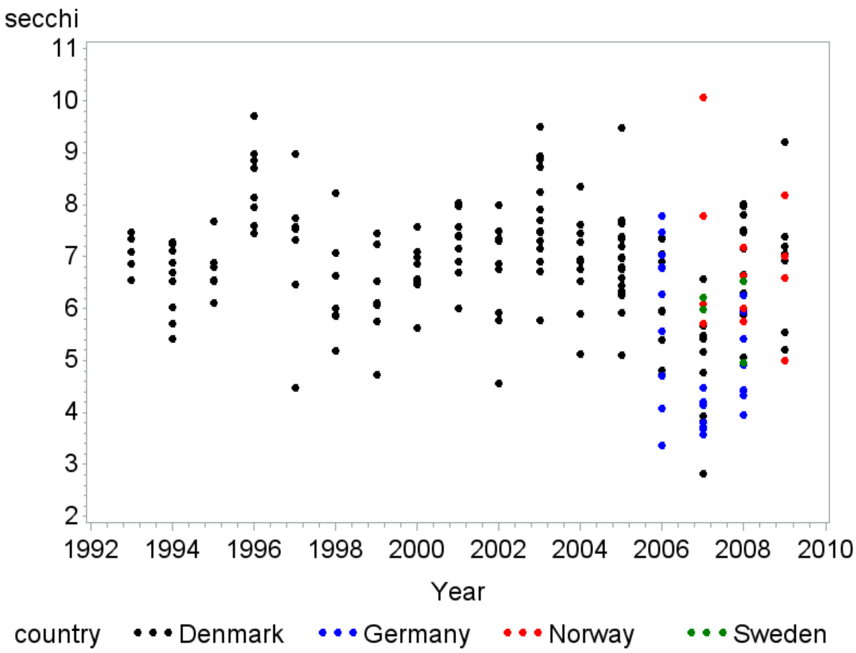
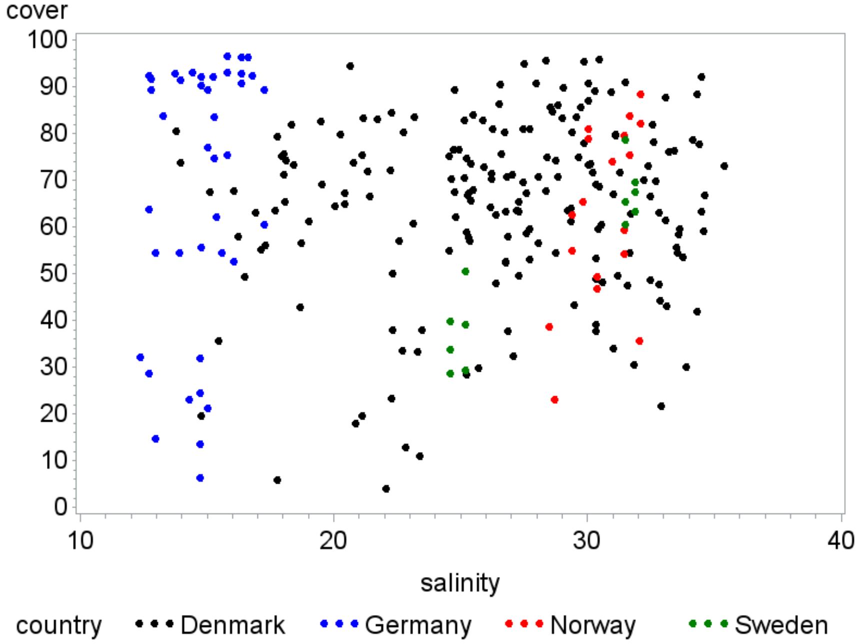
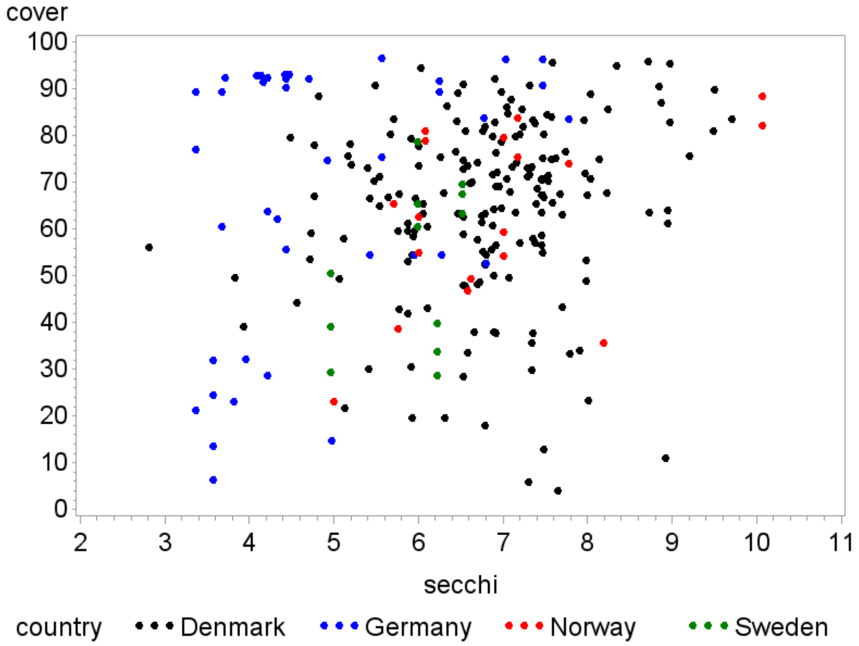


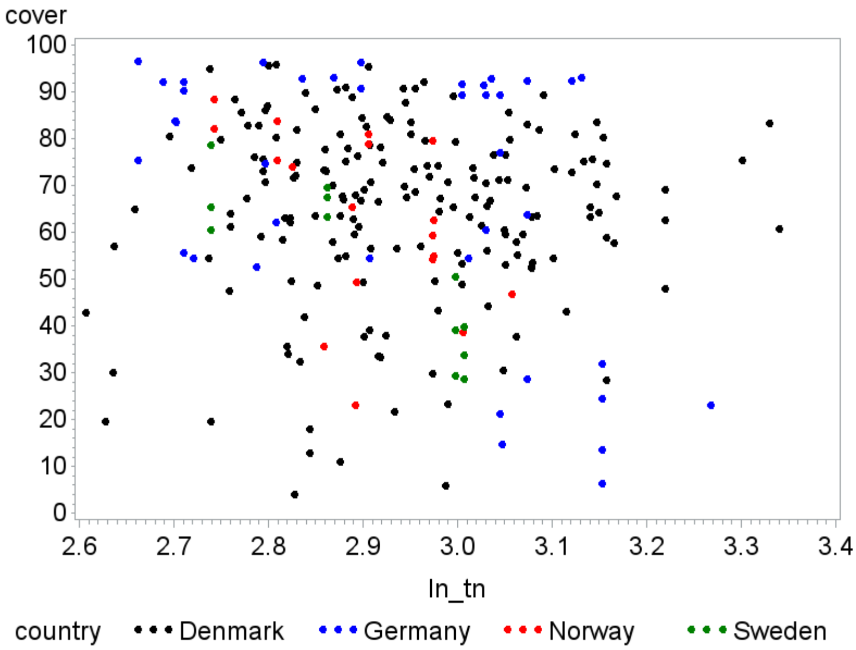
Figure 15. Total cover in% estimated at 15 m water depth (back-transformed from arc-cin) as function of salinity (psu). Data from the North Sea to the Baltic Sea west of the Gedser-Dars sill



Total cover in% estimated at 15 m water depth (back-transformed from arc-cin) as function of Secchi depth (m)



Total cover in % estimated at 15 m water depth (back-transformed from arc-cin) as function of total Nitrogen (µmol/l)



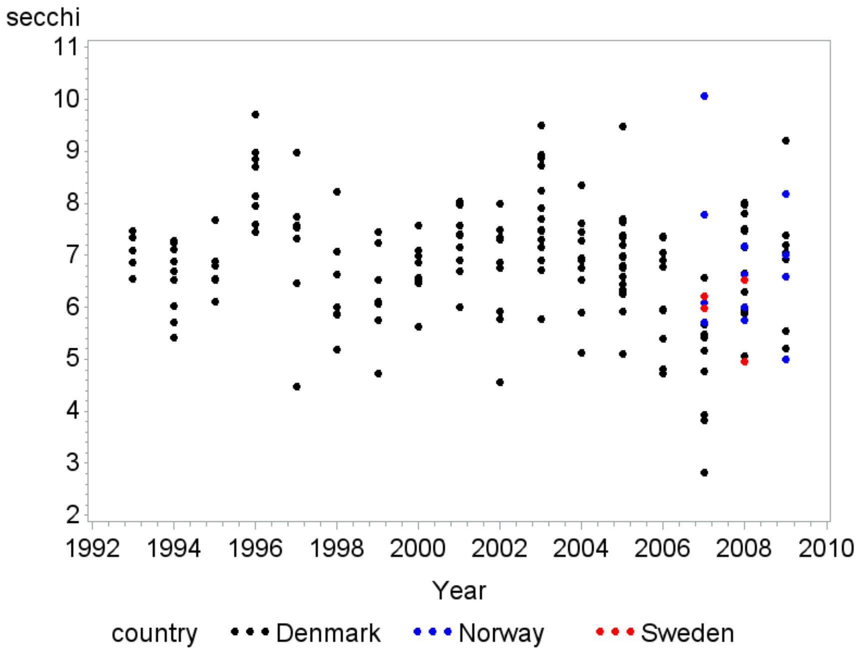
Water quality and vegetation cover Norwegian, Swedish and Danish open and coastal waters

We identified that the German dataset was the reason for the random effect of country in the overall analysis of the medium to high saline dataset. For this reason we made repeated the analysis excluding the German dataset.

Relations between sechi depth and nutrients (model 1+2)

The Secchi depth within this area showed substantial year to year variation (Figure 16).

Figure 16 Modelled average Secchi depth (m) from Norway, Denmark and Sweden based on observations from January to June



In this area we still found a significant and negative effect of TN on Secchi depth (P=0.0038). The Secchi depth model (1) could be reduced to:

Secchi depth (m) = 12.7087 – 2.0406 “lnTN (µmol/l)

There was no effect of country but an effect of locality.

Secchi was also found to correlated significantly with both dissolved nitrogen (P=0.0009), dissolved phosphorus (p=0.0015), salinity (p=0.0012) and a salinity combined with dissolved phosphorus (p=0.0009) Secchi depth decreased with increasing nutrients levels. There were no effects of country, only between stations. This model could be reduced to:

Secchi depth (m) = -1.1696 – 0.5323 “ lnDIN (µmol/l) – 4.9082 “ lnDIP (µmol/l) +0.3147 “ salinity (psu) + 0.1857 “ lnDIP (µmol/l) “ salinity (psu)

Relationship between total vegetation cover and Secchi depth (model 3)

Total cover normalized to 15 m water depth was significantly correlated with Secchi depth (P=0.0017) and salinity (p=0.0036). Increasing salinity and Secchi depth values both resulted in higher total covers.

The model was reduced to:

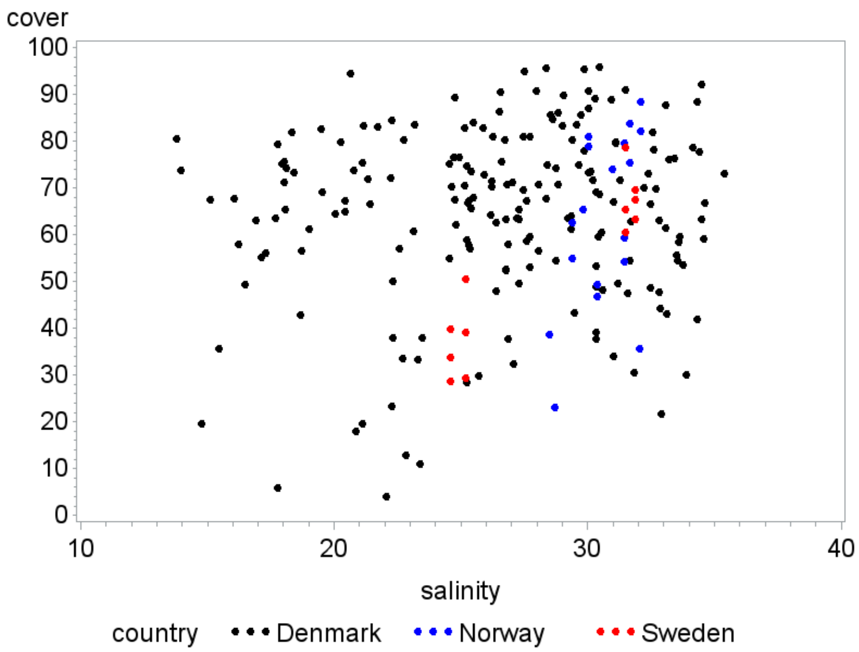
*Arcsin Total cover (%) = 0.3860 + 0.03257 "Secchi depth (m) + 0.01112*salinity (psu)*

We found no effects of country but an effect of locality. The distribution of cover as function of Secchi depth and salinity is shown in Figure 17.

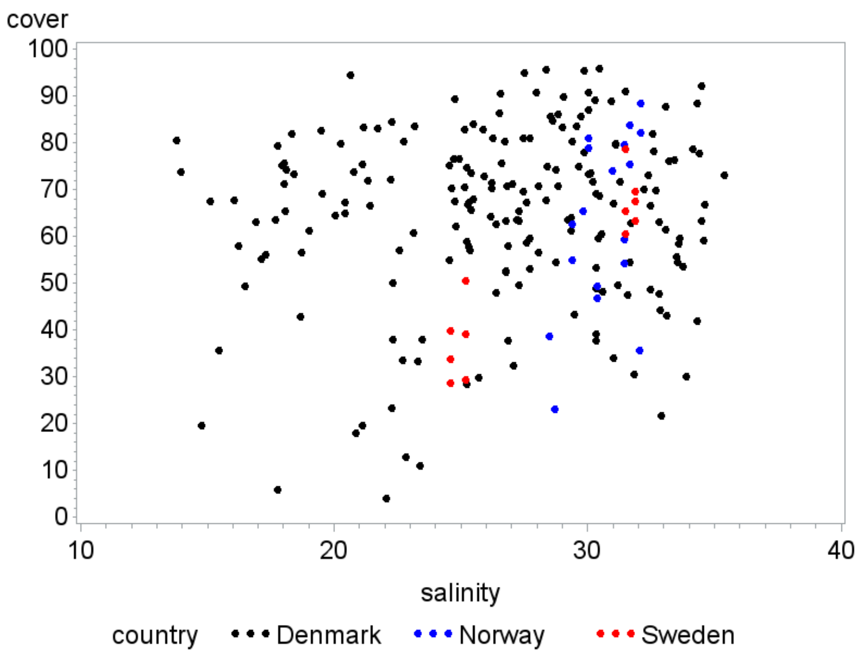
Relation between total vegetation cover and nutrients (model 4, 5 and 6)

In this case we found a significant negative relationship between total vegetation cover and TP (model 4, p=0,0029) and a positive correlation with salinity (p=0,0379)) whereas there were no effect on dissolved nutrients (model 5). Figur 12 show the distribution of vegetation cover in relation to TP. Although we found significant correlations between TN and Secchi depth on one hand and Secchi depth and vegetation cover on the other, TN was not found to correlate significantly with total cover (model 6, P=0,2946).

Figure 17. Total cover in % estimated at 15 m water depth (arc-sin transformed) as function of salinity (psu)



Total cover in % at 15 m water depth (back-transformed from arc-sin) as function of Secchi depth (m)

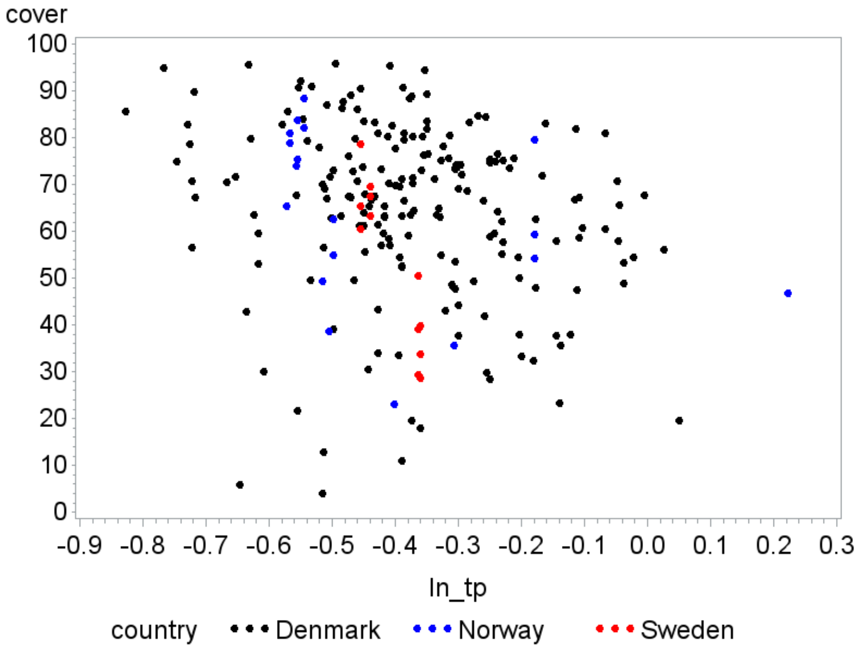


The correlation between total vegetation cover and TP could be described as:

*Arcsin Total cover (%) =0.6006 – 0.2242*lnTP (µmol/l) + 0.008125 " salinity (psu)*

Again we found no effect of country but an effect of locality.

Figure 18. Total cover in % at 15 m water depth (back-transformed from arcsin) as function of lnTP (µmol/l)



Water quality and vegetation cover in Danish fjords and the Baltic Sea east of Gedser-Dars sill

The data set from this area represented 6 countries: Denmark, Estonia, Finland, Germany, Lithuania and Sweden. A few observations of algal cover were available from Swedish waters in the 1970s and 1980s but the majority of observations represented the period after 1990 and particularly after 2000. Physicochemical data associated with the algal observations were also most abundant after 2000. Algal cover as well as nutrient concentrations and Secchi depth showed considerable variability between years and areas (Figure 16).

Relations between Secchi depth and nutrients (model 1+2)

In this area we found a significant effect of both TN (p=0.0005, TP (p=0.0116), salinity 0.0016 and a combined effect of TN and salinity (P=0.0006) on Secchi depth. Secchi depth decreased with increasing nutrients however an interaction term between TN and salinity had a significant impact on the model. The Secchi depth model (1) could be reduced to:

*Secchi depth (m) = 38.7514 – 11.7725*lnTN (µmol/l) – 1.7444*lnTP(µmol/l) – 4.6545*salinity (psu) + 1.6341 " lnTN µmol/l)*salinity (psu)*

There was no effect of country but an effect of locality.

The model describing Secchi depth as function of dissolved nutrients and salinity could be reduced to a model only including dissolved nitrogen only. Increased DIN decreased the Secchi depth (p=0.0282):

Secchi depth (m) = 5.4232 – 0.5574 lnDIN (µmol/l)*

In this case there was not only additional effect of location but also of country.

Relationship between total vegetation cover and Secchi depth (model 3)

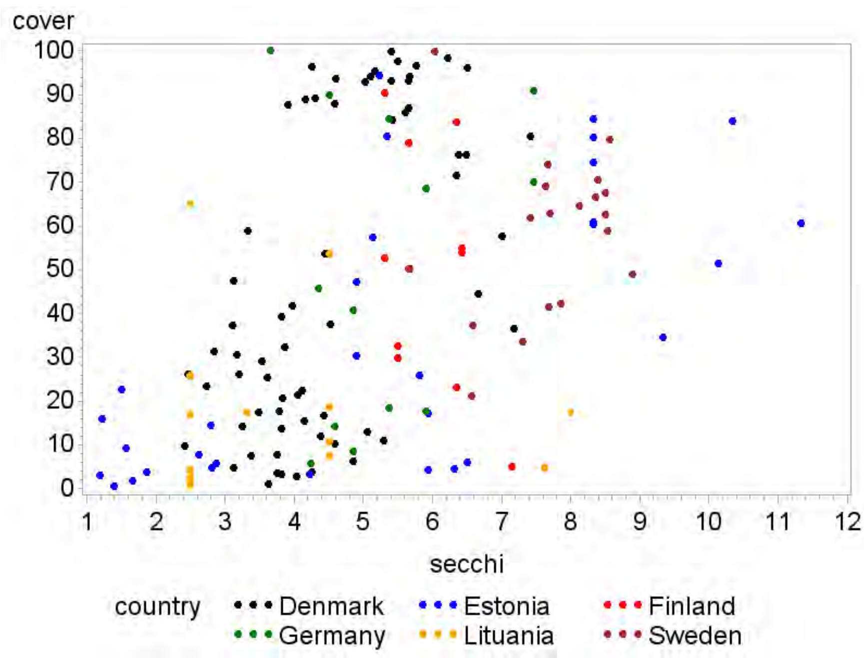
Total cover normalized to 7 m water depth was positively correlated with Secchi depth (P=0.0012). But in this dataset we did not find a significant effect of salinity.

The model was reduced to:

Arcsin Total cover (%) = 0.2344 + 0.06628 " Secchi depth (m)

We found no effects of country but an effect of locality. The distribution of cover as function of Secchi depth and salinity is shown in Figure 19.

Figure 19. Total cover in% estimated at 7 m water depth (back- transformed from arc-sin) as function of Secchi depth (m)



Relation between total vegetation cover and nutrients (model 4, 5 and 6)

In this case we found a significant relationship in model 4 were total vegetation cover correlated negatively with TN (p=0.0029) and salinity (p=0.0062) and positively with a combined effect of TN and salinity (p=0.0019) Vegetation cover at 7m depth could be described as:

*Arcsin Total cover (%) =5.7571 – 1.8677*lnTN (µmol/l) – 0.7488*salinity (psu) + 0.2766 lnTN (µmol/l) " salinity (psu)*

This time we found both an effect of country and location.

There were no correlation between vegetation cover and dissolved nutrients (model 5) and the outcome of model 3, 4 and 5 made run of model 6 unnecessary.

4.1.7 Discussion and conclusion

Our study documents significant negative effects of eutrophication on total macroalgal cover (TC) across the open Norwegian North Sea to the inner Baltic Sea and thereby confirmed our primary hypothesis. In the entire region, Secchi depth declined in response to increasing TN, and in more brackish areas also in response to increasing TP. The large data set with multiple combinations of e.g. levels of nutrients and salinity improves the possibility to distinguish between the various effects and in this case attribute the effect to nutrients. Moreover, Secchi depth generally declined in response to increasing DIN, DIP or both. Total macroalgal cover (TC) declined in response to declining Secchi depth and in response to increasing TN or TP and DIN or DIP, but the analyses were not conclusive regarding N versus P limitation. The combined role of N and P in regulating the algae was particularly apparent in the more saline region where TN caused reduced Secchi depth which again caused reduced TC, but where TP, rather than TN correlated directly with algal cover. This result differs from earlier studies of macroalgae in Danish open and coastal waters which have more unequivocally identified N as the most limiting nutrient (Krause-Jensen et al. 2007, Carstensen et al. 2008, Dahl and Carstensen 2008). Recent reviews highlight the importance of a balanced control of nitrogen and phosphorus even though nitrogen is generally the major limiting nutrient of coastal and marine waters (Conley et al. 2009, Howarth et al. 2011).

Our hypotheses regarding a positive effect of salinity on TC and Secchi depth were only partly confirmed. In open waters with relatively high salinity, salinity did show a positive correlation with Secchi depth and TC whereas interaction effects of salinity and nutrients could be either positive or negative. However, in the more brackish coastal waters and fjords, salinity showed no direct correlation with Secchi depth, a negative correlation with TC and positive interactive effects with nutrients on both Secchi depth and TC. The positive effect of salinity on TC in the open, most saline waters is likely related to the high diversity of algal species in the most saline waters (Nielsen et al., 1995) which, by representing various life forms and forming a multi-layered community, should be able to exploit the incoming light more efficiently and thus be more productive and dense than a less diverse community (Spehn et al., 2000). A large diversity is not universally a prerequisite for high cover, however, as, for instance, *Fucus vesiculosus* as a monoculture may exhibit high cover in a restricted depth interval.

In the more brackish areas, the negative effect of salinity on TC may be associated with interactions with blue mussel. Blue mussels may affect TC through a combination of shading by newly settled shells and sedimenting faeces, competition with algae for substrate, and by constituting an unstable algal-substrate supporting less algal cover (Albrecht 1998). The negative effect of salinity on total algal cover in the areas of low salinity may also be related to the fact that green algae, which are not as firmly fixed to the substratum as brown and red algae, constitute an increasing fraction of the algal community as salinity declines (Nielsen et al. 1995). The special hydrographic characteristics of the study area may contribute to explaining the diverging effect of salinity between regions. In the study area, the traditional estuarine gradients of nutrient-rich freshwater from land mixing with less nutrient rich saline water towards the open coasts is thus overlaid by a large-scale gradient of brackish, less nutrient-rich Baltic mixing with saline, more nutrient-rich North Sea and Skagerrak waters.

All the identified relationships between macroalgal cover and physico-chemical factors showed considerable variability. We identify four factors as major contributors to this variation which we discuss below: 1) omission of important regulating factors in the analyses, 2) methodological differences in data acquisition between countries and 3) imperfect coupling between algal sites and physico-chemical sites 4) scattered data for estimation of mean values of Secchi depth and nutrient concentrations.

Macroalgal communities are regulated by a complex of anthropogenic pressures and environmental factors which are not all included in the analysis. Eutrophication, taken into account in the analysis, is a major factor acting upon the attached macroalgal community through effects on e.g. water clarity, growth of epiphytic and drifting opportunistic algae, and oxygen conditions (e.g. Cloern et al. 2001, Kemp et al. 2005, present results). The large salinity gradients of the region, also taken into account in the analysis, equally have fundamental effect on the algal communities (Nielsen et al. 1995, Middelboe et al. 1997, present results). However, other pressures not considered in the analysis, also affect the macroalgal community and thereby contribute to the variability in the identified relationships between eutrophication, salinity and macroalgal cover. Fishing activity acting through top-down control is expected to exert a major regulating effect on the macroalgal community which may also interact with effects of eutrophication (e.g. Jackson 2001, Steneck et al. 2002, Baden et al. 2010). Physical disturbance of the sea bottom and coastal construction works may also play a local regulating role, both directly through burial or destruction of the macroalgae and indirectly by reducing water clarity (e.g. HELCOM 2010). Water temperature is also a major regulator of metabolic activity, growth and geographical distribution of seaweeds (Lüning 1990) which is, unfortunately, not included in the present analysis. The geographical gradient covered by the present study indeed includes a significant variation in seawater temperature which is likely to explain some of the spatial variation in data. Projected temperature increases are also likely to affect large-scale patterns of distribution and abundance of macroalgal communities globally (Müller et al. 2009) as well as in the study region. Increased temperatures may also accentuate the negative effects of eutrophication on marine vegetation as higher temperatures tend to stimulate respiration rates and thereby increase the light demand of the vegetation. The recent decline in the distribution of sugar kelp along the Skagerrak coast of Norway may be due to increased temperature in combination with increased run-off from land and interaction with eutrophication (e.g. Moy et al. 2008, Moy & Christie submitted). Inclusion of water temperature in future analyses of the large-scale dataset may, thus, reduce the unexplained variation. Physical exposure expressed as fetch or wave indices would also be relevant to include in future analyses as it is an important modulating factor in macroalgae communities (Wernberg and Connell 2008).

Methodological differences in data acquisition among the national monitoring programs cause major variation in the results since we had to apply many assumptions in order to make the various estimates of total cover comparable (see methods). The significant effect of country indicates that the data harmonization was not completely successful. The German data set, in particular, differed from the rest most likely due to the many estimation procedures needed to obtain data on total cover on hard substratum. Moreover, only few of the German samples represented water depths where the algal community was light limited and the "algal cover versus depth model" was therefore weak. The exclusion of German data from the analysis eliminated the significant effect of country and thereby increased the comparability of the remaining data. The fact that we conducted separate analyses for the saline, open areas in the west and the more brackish waters of the Danish fjords and the Baltic Sea east of the Darß sill also helped increase the comparability among the data included in each set of analyses. The open more saline areas had deeper algal communities which were best described by mean values at 15 m water depth while the less saline and generally more protected stations of the fjords and the Baltic Sea had shallower algal belts which were better described by mean values at 7 m water depth. Natural variability in algal cover between subsamples, stations, divers, years and seasons also contributes to the uncertainty in the estimated mean values of algal cover; a variability which can be reduced by optimizing design and intensity of monitoring programs and conducting training and intercalibration exercises among divers.

Imperfect coupling between algal stations and physico-chemical stations is another major methodological reason for the considerable variability of the relationships between algal cover and environmental variables. In some cases physico-chemical stations were located at quite some distance from algal stations. The estimation of average values of nutrients and Secchi depth could probably be improved substantially by increasing the sampling frequency of physico-chemical stations to better match the highly dynamic seasonal variation and by ensuring that the physico-chemical stations actually reflect the conditions in the near-score locations where the algae grow. This all together imply that the actual physico-chemical conditions at the algal sampling sites were not well described.

In spite of large variability in the identified relationships, our results demonstrate that it is possible to describe a key element for hard bottom habitats, in this case "total algal cover" as a function of eutrophication pressure and salinity over wide geographical ranges. The study also highlights the advantage of applying models which allow harmonization and comparison of data sampled at different depths, years, seasons and by different divers and subsequently relating the data set to physico-chemical regulating factors. A future adjustment of national monitoring programs to include a direct measurement of total algal cover on hard stable substrate could easily be implemented and would reduce the random variability in data and thereby improve the predictive power of future models. Harmonised large-scale monitoring programs with coupled information on macroalgal cover and a range of governing environmental variables also have perspectives with regard to establishing maps of the macroalgal habitats and their potential cover. Such data would allow a better quantification of this important habitat and thereby also make it easier to protect it.

4.2 Modelling of seaweed forests on reef habitats

4.2.1 Introduction

Data on detailed mapping of reef areas in Nature 2000 sites are rare in the Baltic. In most cases mapping has not been done and in some cases access to existing data are restricted. A proper mapping needs high resolution data on seabed composition as well as bathymetry. Information on depth is necessary to predict the biological component of the habitat. The Danish N-2000 sites Kim's Top and Hatter Barn are examples of reef areas with available high resolution mapping data provided by Danish Maritime Safety Organisation and the Geological Survey of Denmark and Greenland. In both cases it has been possible to add a key biological component to the description of the hard bottom habitats if form of cover of macroalgal vegetation. (Dahl et al. 2007 and Dahl et al. in prep).

4.2.2 Aim

The aim of this part of the project is to investigate the applicability of existing vegetation models to produce habitat maps for seaweed forests on reef areas in neighbouring NATURA-2000 sites. This work is done by extrapolate existing site-specific total and cumulative cover models to other local reef areas with available high resolution data on bathymetry and substrate. Lilla Middelgrund in the Swedish part of Kattegat was chosen as a case story.

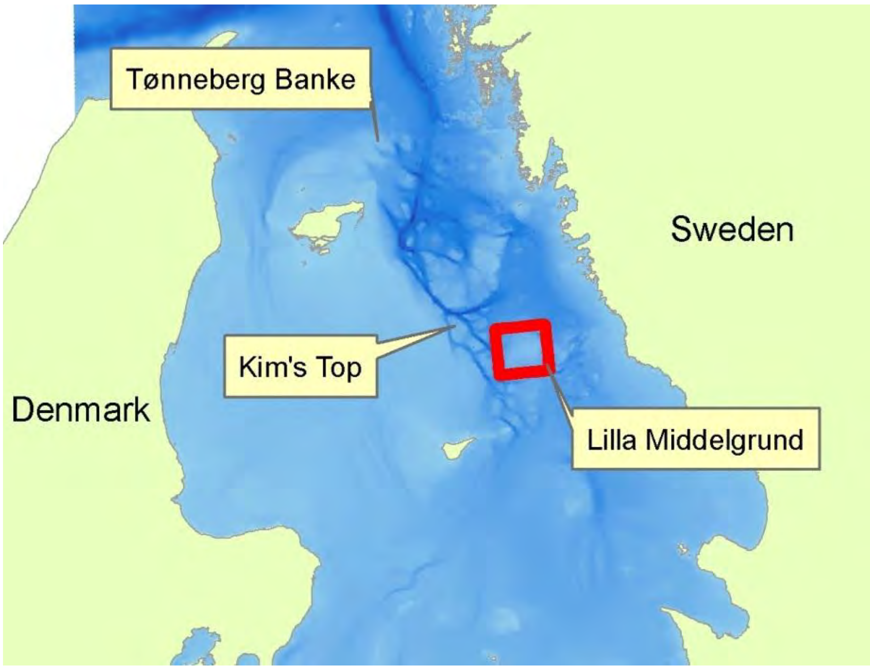
4.2.3 Data and methodology

Vegetation models for habitat modelling and site specific quality assessment

Site-specific statistical validated models describing total and cumulative vegetation cover exists for a number of reefs in the Danish part of Kattegat describing the vegetation cover as function of depth, nitrogen load from Danish and Swedish rivers and point sources, solar radiation and grassing pressure of sea-urchins (Dahl & Carstensen 2008). The models have been developed based on data collected as part of the Danish national marine monitoring program (Bilj 2007) The site specific models have also proven to be useful to nearby reefs (Dahl et al. 2007).

The existing models have also been intrapolated and extrapolated to include the Swedish reef Lilla Middelgrund as well (Figure 20). Lilla Middelgrund is a shallow area in the open central part of Kattegat. Lilla Middelgrund is located nearby the reef Kim's top but due to difference in the vertical distribution of hard bottom between the two reefs is was necessary supplemented with modelled algal distribution data from the more northern Tønneberg Banke for the shallow parts.

Figure 20: Reefs in Kattegat with suitable assessment models. Models from Kim's Top and to some extend Tønne Berg Banke is used for the Swedish Lilla Middelgrund



As input for the vegetation model we have used two scenarios of nitrogen load to Kattegat. The first scenario is based on a total nitrogen load on 48,000 tons from representing the average load to Kattegat from 1993 to 2006 in the first half year (January–June). This scenario has been used to test the validation of the extrapolation of model data from near-by Danish reefs to the Swedish Lilla Middelgrund.

Vegetation data

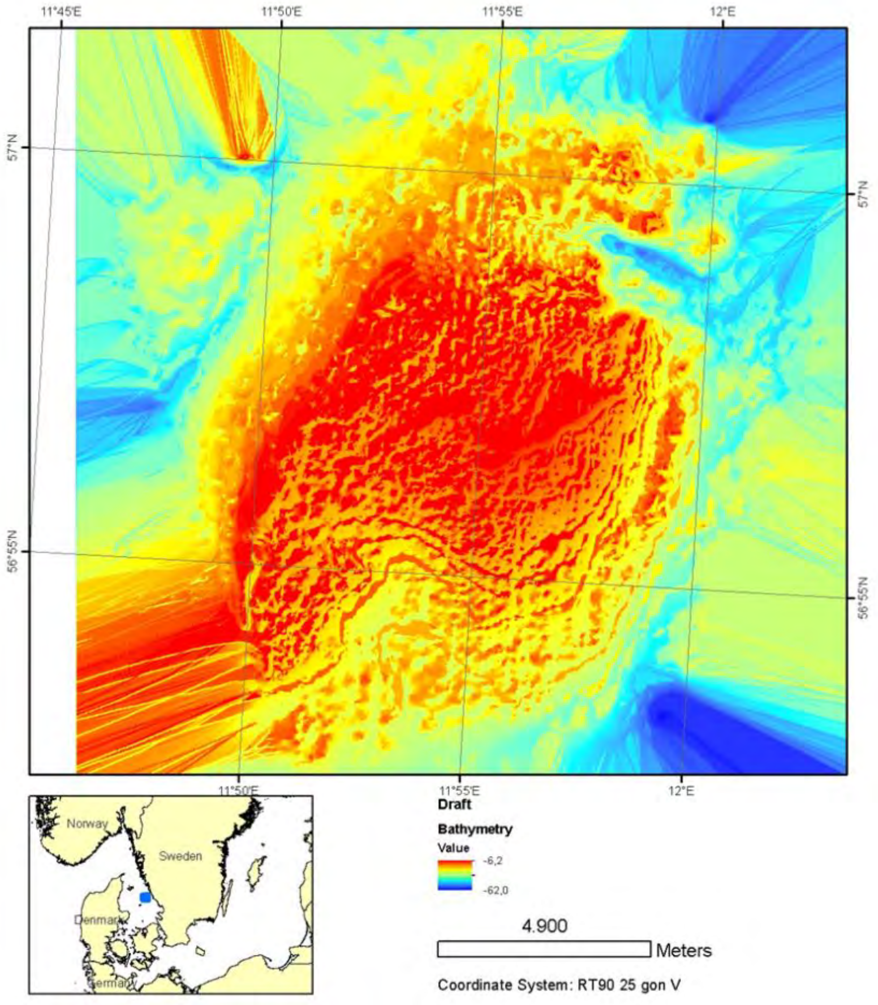
The vegetation at Lilla Middelgrund has been studied at several occasions. In 1997 Karlson (1997) visited the reef at several sampling stations and in 2006 DMU also made observations as part of the Balance project (Dahl et al. 2007). The investigation in 1997 described the species specific vegetation cover according to 4 classes (<5%, 5–25%, 25–75 and >75%) and no data on total cover was sampled. The investigation in 2006 used in most cases a submerged video camera with focus on seabed description and collection of total cover data and data on other large visible species. Only one dive was carried out on relative shallow waters. In this case species specific algal cover data was collected in percent cover of hard stable substrate.

Data on total cover of vegetation of the Danish reefs has been collected at as part of the National monitoring program.

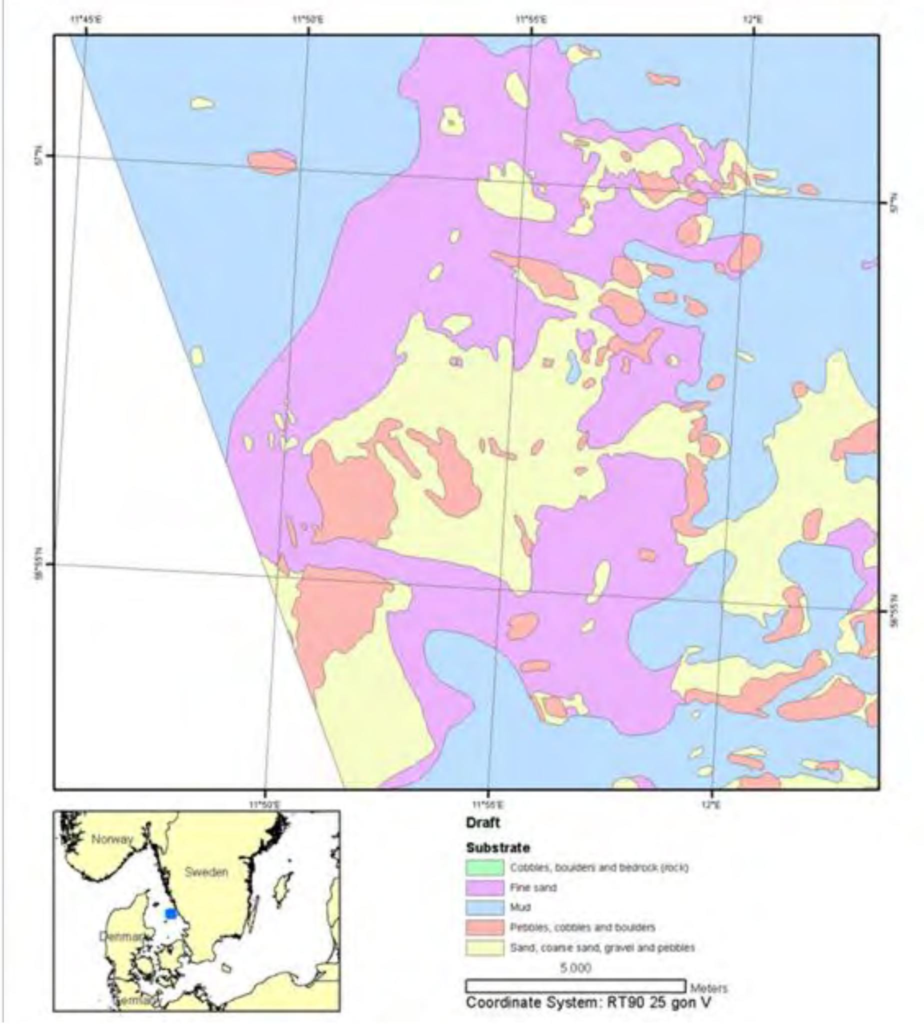
Seabed maps for seaweed forest habitat modeling

The Swedish Geological Survey, SGU) have identified mapped hard substrate on Lilla Middelgrund from 6 to 50 m water depth (Naturvårdsverket 2010) and bathymetry data is also available for the area although the spatial resolution was not very high.

Figure 21. Bathymetry of Lilla Middelgrund. Data from Naturvårdsverket 2010



Seabed classification of Lilla Middelgrund. From Data from Naturvårdsverket 2010



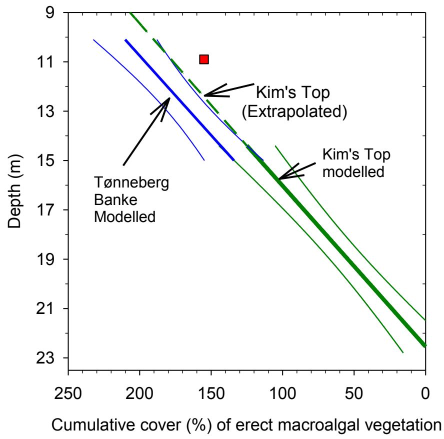
Verification of use of existing models

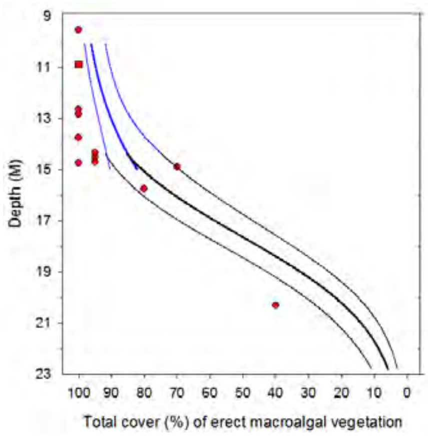
The modeled cumulative vegetation cover at Kim's Top and Tønneberg Banke almost had the same slope but with a 20% higher vegetation cover at the northern Tønneberg Banke (Figure 19). The choice of using an extrapolation of the vegetation model from Kim's top to more shallow areas at Lilla Middelgrund is thus justified. The difference in total cover between the two reefs was small and in this case a merged model was used.

The actual observations of cumulative cover and total cover on Lille Middelgrund are compared with the predicted models (Figure 22). It is obvious that the model prediction of total cover close to 100% cover is not perfect but on deeper water the match is acceptable. The logistic model used to describe total cover is asymptotic getting closer to 100% and for this reason it is not suitable to predict the vegetation cover at this range.

There only exists one sampling (from 2006) that has used percentage cover of individual species, which is a prerequisite for calculation of cumulative cover. However this observation is also comparable with the predicted observation based on the model from Kim's Top with a deviation from the estimated value of approximately 20% cover.

Figure 22





Left: Modelled cumulative cover at different depth at Kim Top and Tønneberg Banke with a total nitrogen load on 48.000 tons to Kattegat from January to June (solid thick lines) with 95% confidence levels (solid thin lines). The green stippled line represents extrapolated cumulative cover at Kim's top to more shallow waters. The red square represents observed cumulative cover at Lille Middelgrund in 2006.

Right: Modelled Total cover at different depth at Kim Top and Tønneberg Banke with a total nitrogen load on 48.000 tons to Kattegat from January to June (solid thick lines) with 95% confidence levels (solid thin lines). The red square represent diver observed total cover at Lille Middelgrund and the red dots represent total cover observed by drop video.

Habitat mapping

Habitat maps have been made describing the vegetation cover on Lilla Middelgrund combining information on bathymetry, seabed sediments and the vegetation models. Figures 23 and 24 illustrate the development of vegetation in a scenario with a total nitrogen load to Kattegat of 23,000 tons equal to the two dry years in 1996 and 1997. The vegetation covers shown on the maps are restricted to the depth intervals available from the models (10–23 m). Please note that the maps show the vegetation cover only on the hard stable fraction of the classified seabed sediment which can also include a large fraction of sand and other mobile sediment.

Figure 23: Estimated total vegetation cover on stable hard substrate at Lilla Middelgrund in Kattegat in a scenario with a total nitrogen load of 23,000 tons from diffuse land sources and point sources from January to June

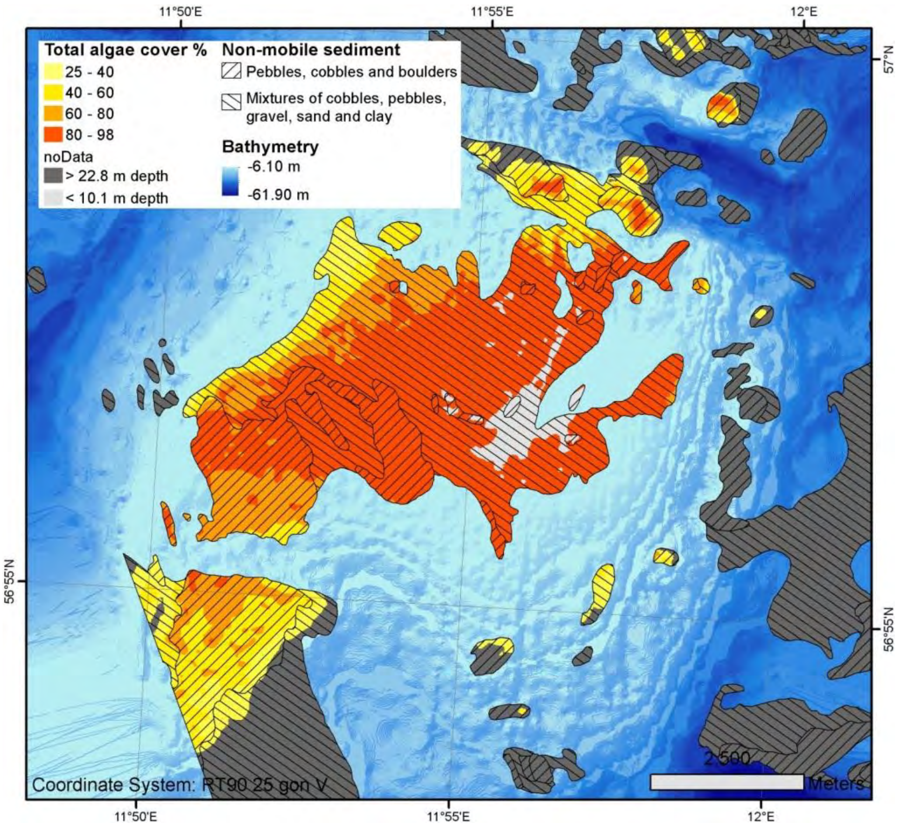
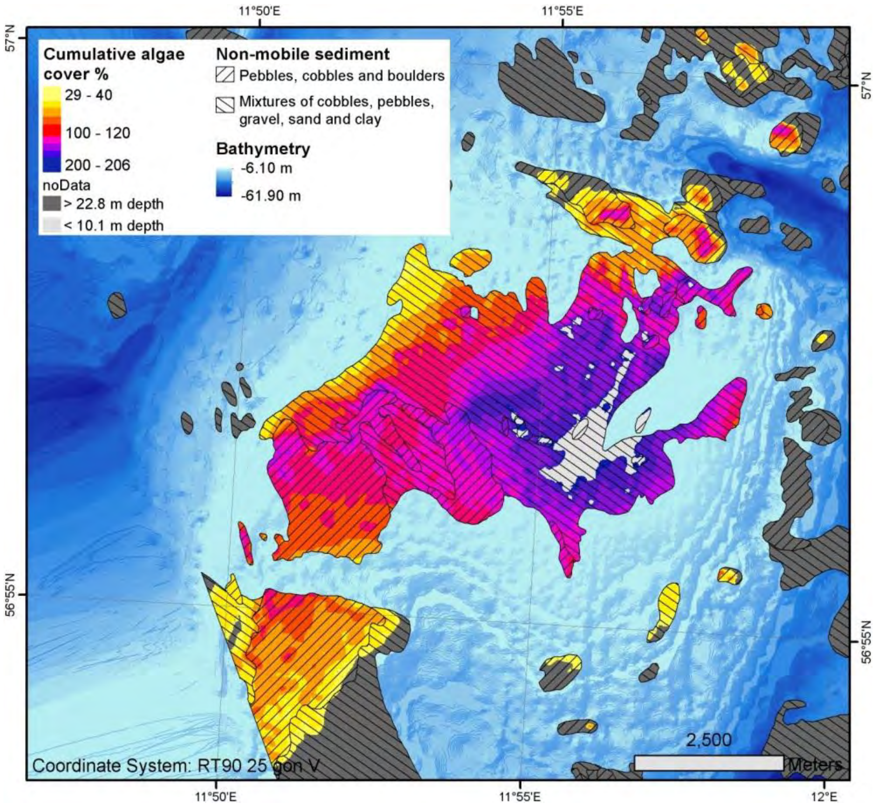


Figure 24: Estimated cumulative vegetation cover on stable hard substrate at Lilla Middelgrund in Kattegat in a scenario with a nitrogen load of 23,000 tons from diffuse land sources and point sources from January to June



4.3 Habitat Model for bladderwrack *Fucus vesiculosus*

The brown seaweed bladderwrack constitutes a true key species in the very shallow part of the Baltic marine ecosystem. Not only is the species characteristic of many reef areas, but the bushes of bladderwrack also serve as shelter and feeding habitat to many crustaceans, a wide array of invertebrates, such as snails and crustaceans, and many fish species with their offspring. Most monitoring data on the coverage of bladderwracks are biased towards areas of known presence of the species. For this reason a presence-only statistical model technique MaxEnt was applied.

4.3.1 Methodology

The prediction was created through a Maximum entropy model using the software MaxEnt (Phillips et al. 2006, available at www.cs. princeton.edu/~schapire/maxent). This is a statistical model using presence data only which has shown to perform well in comparison with other models (Elith et al. 2006).The model was transferred to predictor variables in grid-format using ArcGIS thereby creating a spatial prediction. In total 3927 presence training data points were used to build the model. The dataset for external validation consisted of 3,056 presence/absence data points. The external validation was made using the area under the ROC curve (AUC) which is a standard measure for evaluation of habitat maps. Many different models were produced based on different variable composition and model complexity. The model with the highest AUC score in the external validation was chosen as final model. The probability layer that initially is produced by the model was classified into suitable and not suitable habitat based on a cut-off value that maximizes sensititvity and specificity. The predictor variables used in the model were depth, salinity, fetch and a categorical variable sand/other. The biological field data and predictor variables are described in more detail below.

4.3.2 Biological data

Biological data were provided from 7 different countries around the Baltic Sea. The data consisted of diving transects, video transects and point data. The different datasets were as follows:

- Sweden – Diving transects and video transects. Evenly geographically distributed. Biased towards areas with hard substrate. Most video transects come from offshore bank inventories
- Finland – Diving transects and video transects. Mainly from 3 different areas along the Finnish coast. Unevenly geographically distributed data
- Estonia – point data, evenly geographically distributed
- Lithuania – point data. Only absence data since bladderwrack is not found along the Lithuanian coast
- Denmark – transects and point data. Evenly geographically distributed
- Norway – point data from stations within the national monitoring program. Few, but evenly geographically distributed points

- Poland – point data from 3 different areas representing sheltered, exposed and medium exposed areas of the coast. Only absence data. Unevenly geographically distributed data

The biological datasets had different spatial resolutions and were unevenly geographically distributed (Figure 25). To balance the dataset, many data points were removed from overrepresented areas. Data points were also removed to provide a dataset balanced in respect to environmental predictor variables such as depth and wave exposure. The diving transects are laid out perpendicular to depth curves which means that the depth distribution is very well represented in the data set, more than the distribution of the other variables thus influencing the model accordingly. Data points at different depths in the same transect bear less resemblance than data points on the same depth but more distant from each other, which is why data points from the same transects are not considered spatially confounded in a way that would produce errors in habitat modelling. However, in order to balance the dataset in a way that give way for the other variables to influence the model, only a few points per transect were kept as training data. All data were cleared from temporal duplicates (i.e revisits). Subsets of the presence data from the different countries were withheld as test data for validation of the model.

From the above datasets presence records were extracted for training data (i.e to build the model) whereas a subset of absence records were added to the validation dataset. The training data thus consists only of presence data (Figure 26) whereas the validation dataset consists of presence and absence data (Figure 27).

Figure 25. Available biological field data

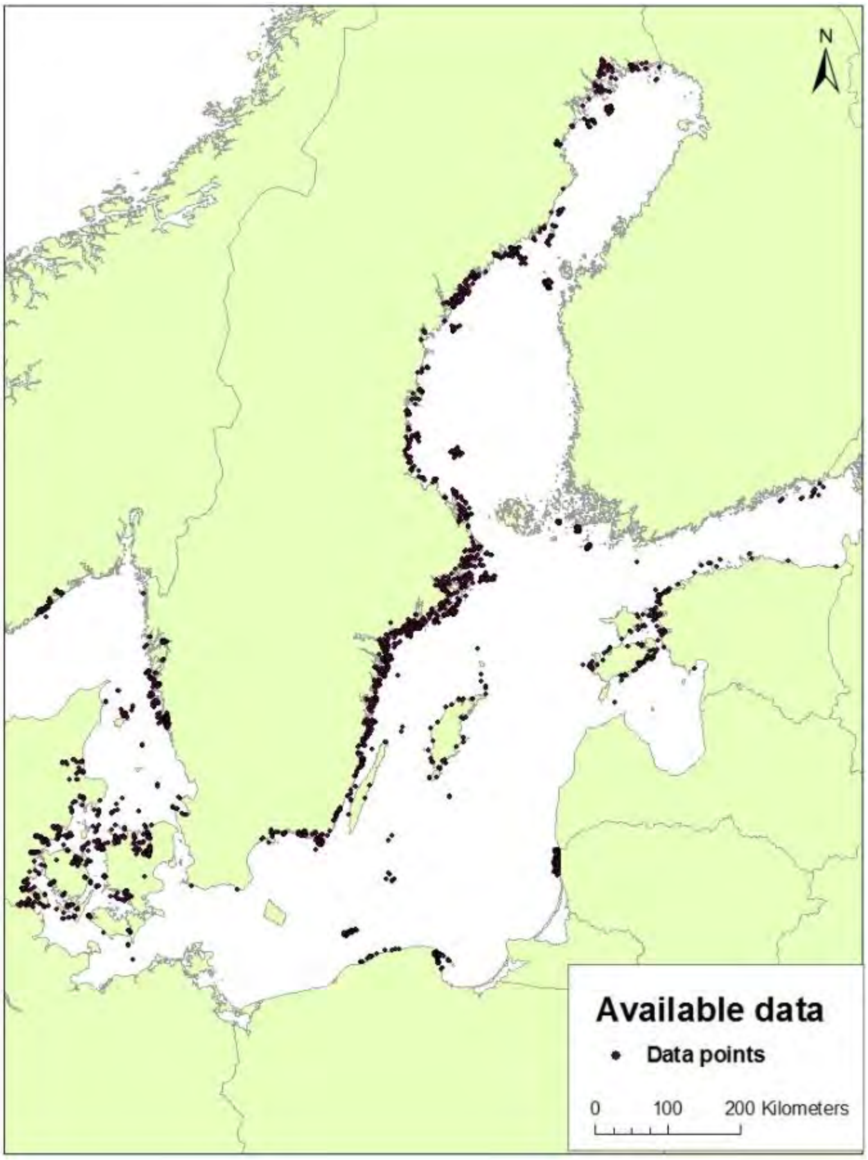


Figure 26. Bladderwrack presence data, used to build the model

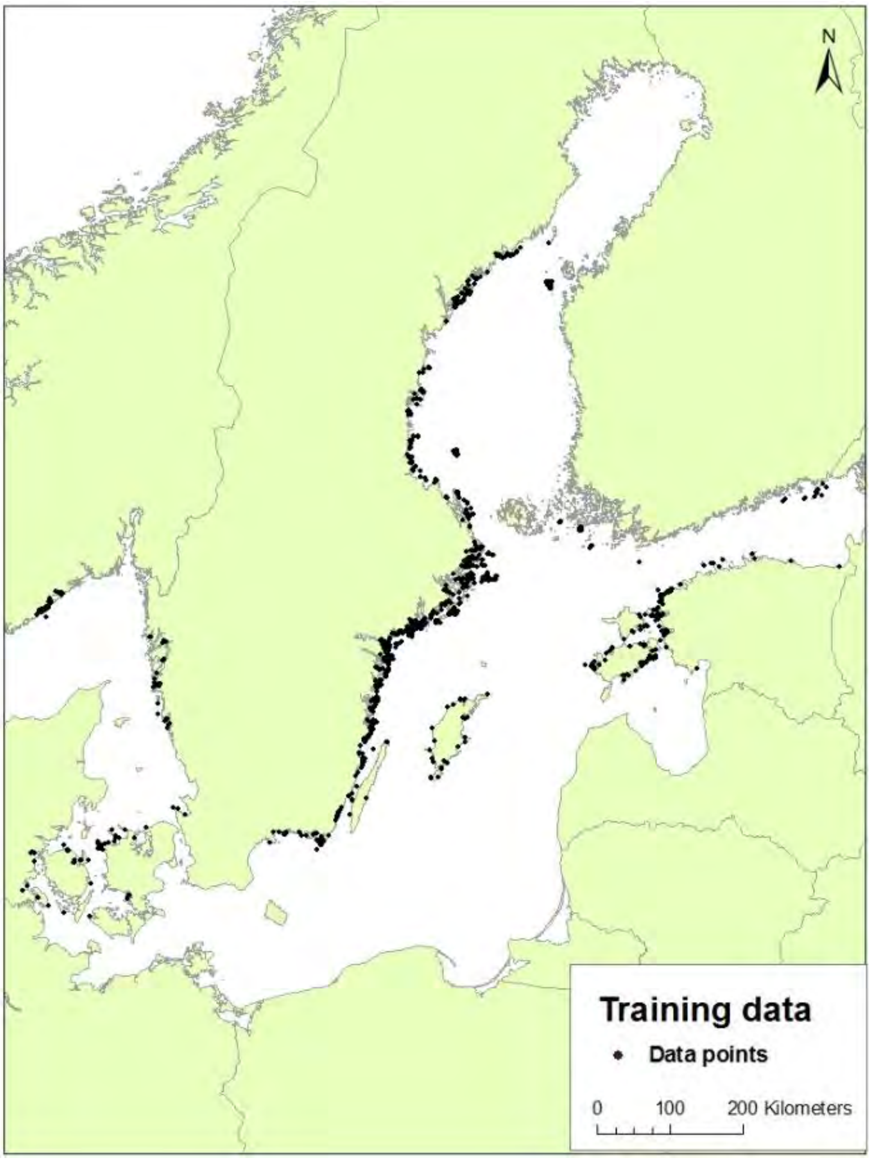
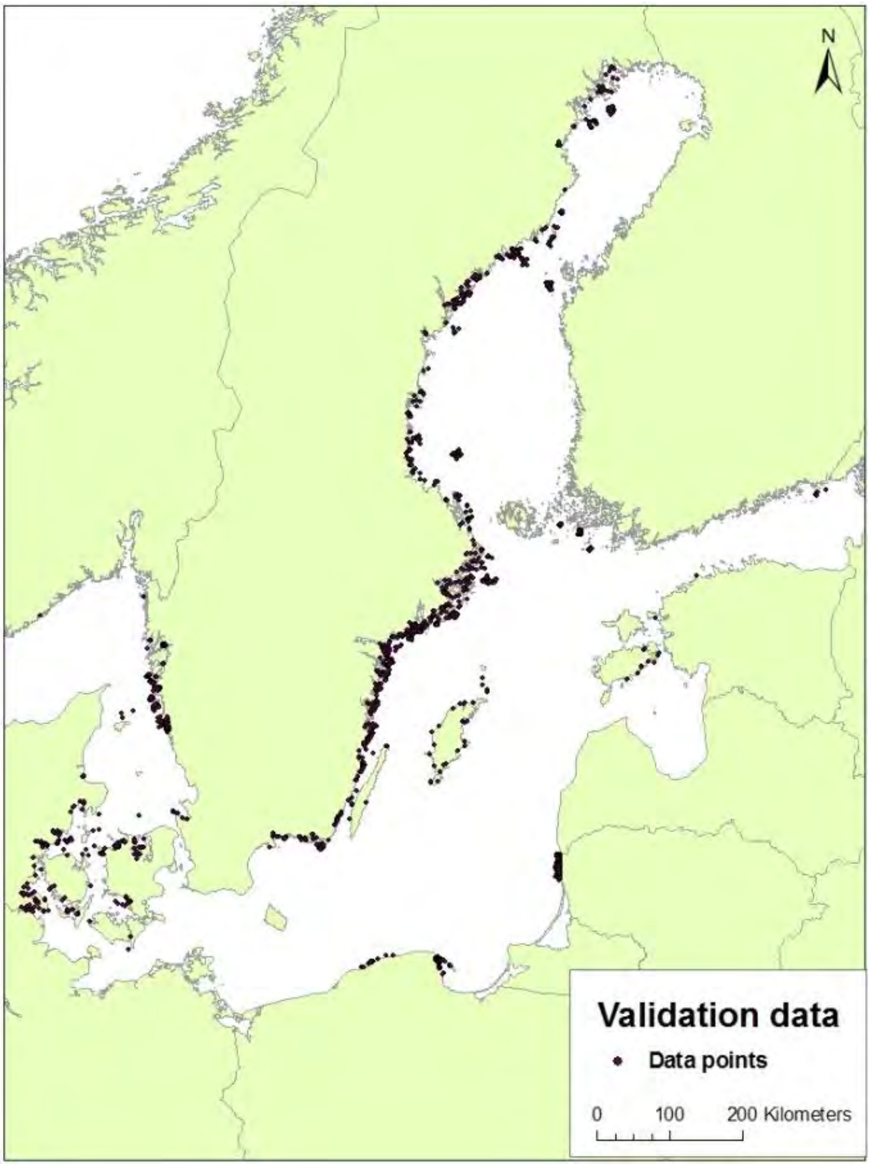


Figure 27. Validation data (presences and absences) for the bladderwrack model



4.3.3 Environmental data

Fetch

Since wave exposure layers were missing for many parts of the Baltic Sea a grid that describes the fetch for the entire MopoDeco area was developed for this project. The fetch-layer is a simplification compared to e.g SWM (Simplified Wave Model) and other exposure models that also incorporates wind regimes in the model.

Depth

The bathymetry grid from the Balance project was used as depth layer (Al-Hamdani and Reker 2007). In addition, for parts of Sweden, Estonia, Finland, Germany and Denmark finer resolution grids were available. These were mosaiced into of the Balance layer to create a grid using the best available depth data.

Salinity

A salinity model was developed by DHI for the Balance project (Al-Hamdani and Reker 2007). This grid was refined at some coastal areas where better field data, representing the effect of freshwater run-off, were available. The layer development is described in more detail below. See Figure 28 for the final grid.

Sand/other

This categorical layer was created through a reclassification of the sediment layer developed in Balance (Al-Hamdani and Reker 2007). Areas with sand were classified as sand and all other areas were classified as no sand/other sediments. The balance sediment map as a whole was considered too coarse for the purposes of bladderwrack habitat mapping and would probably create errors in the model and prediction. The sediment class sand was assessed to possibly provide some valid information on the large sandy areas in the southern Baltic Sea (Figure 29). An attempt to use the sediment class mud as a possible non bladderwrack habitat class was also made. Analysis showed that this sediment class often correlated with areas known to support bladderwrack. In many areas bladderwrack is found in the shallowest part of the coast, attached to bedrock that deeper down is covered in mud. Such areas are classified as mud in the balance sediment layer although a narrow zone closest to the shoreline often consists of bedrock. This is common in more medium exposed to sheltered archipelago areas.

All predictor layers used in the model (depth, salinity, sand and fetch) was cut according to ranges in the training and validation data sets in order to avoid extrapolation and to assure a correct measure of predictive ability (i.e. AUC).

Figure 28. The refined salinity layer that was used in the model and prediction

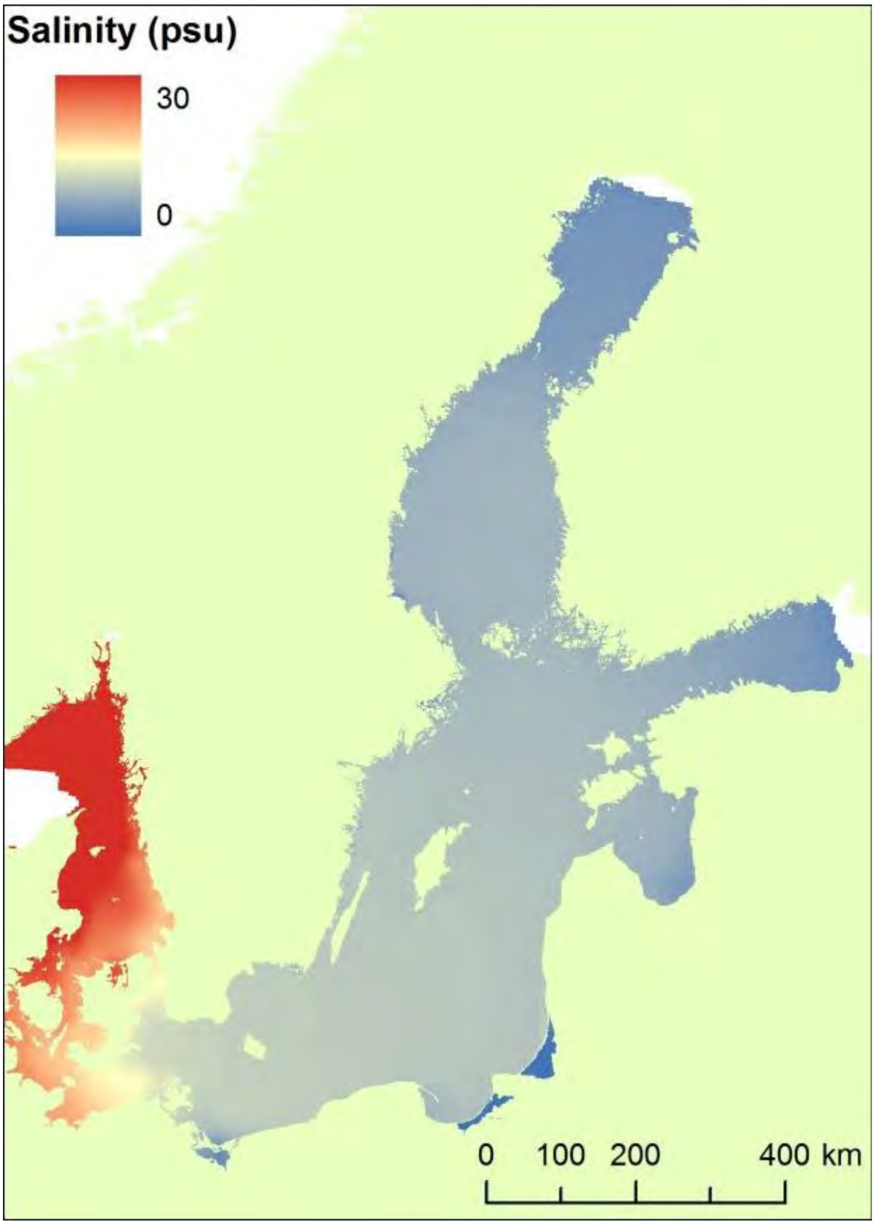
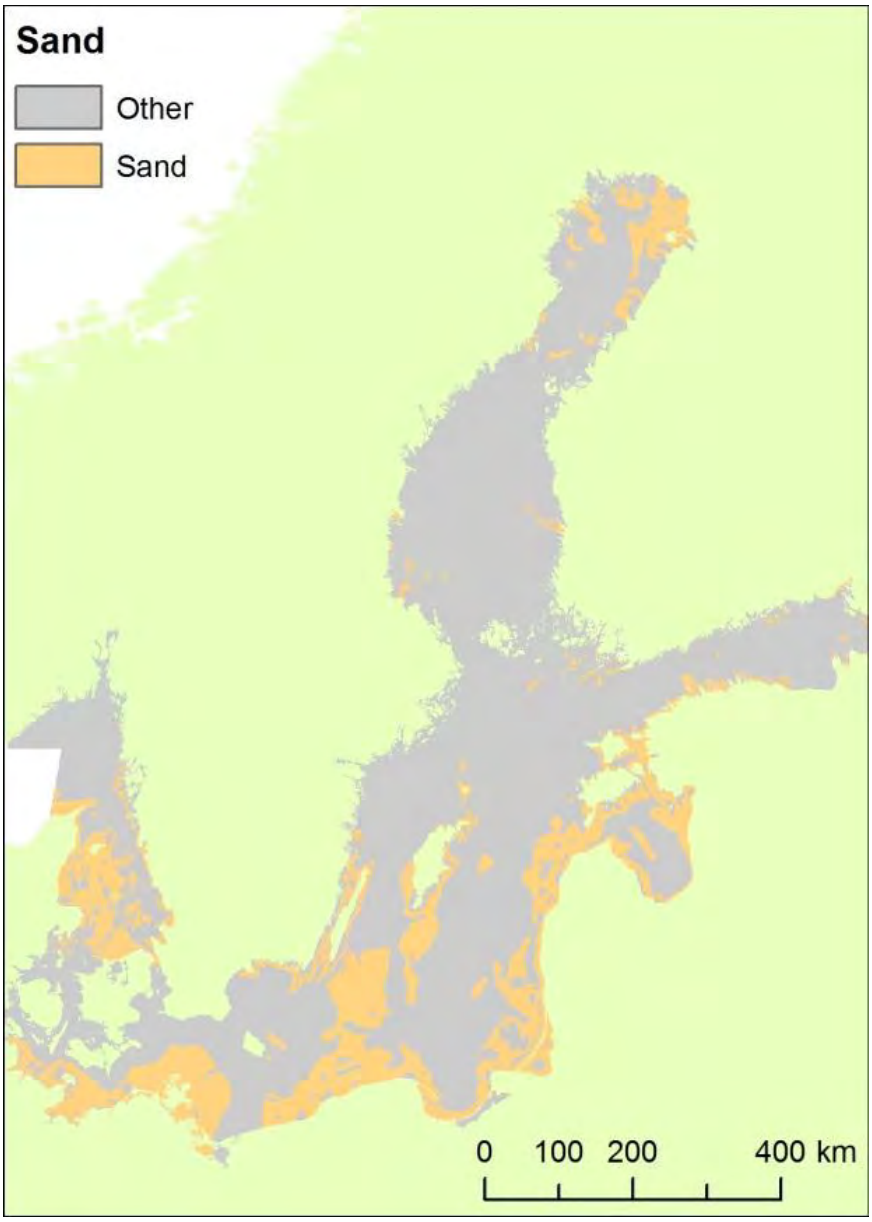


Figure 29. The reclassed sediment layer showing



4.3.4 Salinity layer refinement

The fundament of the salinity information was provided by a mean-salinity layer modelled by DHI within the Balance project for 0–5 m depth over the years 2003–2005 (Al-Hamdani and Reker 2007). The resolution of this mean-salinity grid, 5,000 m, is too coarse to realistically resolve local/small-scale salinity gradients typical for coastal areas.

An effort was therefore made to construct additional local grids describing a refined salinity gradient for several sites along the coast of the Baltic Sea: parts of the Swedish east coast, the Finnish Archipelago Sea, the Gulf of Finland, the Gulf of Riga, and the west coasts of Latvia and Lithuania. These specific regions of the Baltic Sea were selected due to the fact that salinity field data were readily available with respect to the time frame of this project task.

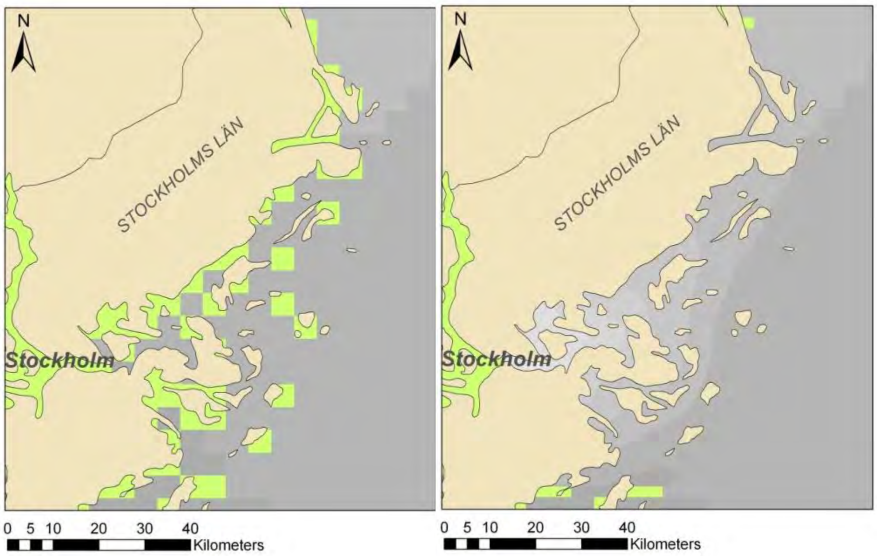
The additional surface salinity data (0–5 m depth) were compiled from different sources, see Table 1. The search of data was extended to the ten-year period 1997–2007 to reassure that a significant amount of data could be collected. For each region the new gridded product was constructed by first converting a sub-region of the DHI-raster to a point shapefile. Next, these point data were merged with the field point-data. After reassuring that the added data were characterized as clustered through Moran's test the merged file was further interpolated into a new raster with 200 m resolution through Kriging interpolation. Due to the relatively sparse and irregular data-coverage the interpolation was carried out without considering land features (islands). By this procedure the salinity information was found to be improved at six of the tested regions (see Table 10). An example is shown for the Stockholm archipelago in Figure 30. For the remaining regions insufficient temporal and/or spatial coverage of the available data led to inconsistent results and/or poor improvement of the salinity gradient as represented by the mean-salinity-raster. For these regions the mean-salinity pattern was therefore bound to be regarded as satisfyingly appropriate. In the final step, all the new constructed grids were overlaid onto the mean salinity raster through a mosaic function, resulting in a final product of a mosaic digital raster with 200x200 m cell-size. The work was conducted in the ESRI software ArcMap 9.2, in the UTM34N map projection.

Table 10. Sources for the additional salinity data used in the efforts to refine the basic mean-salinity layer (ordered from north to south)

Region	Data Source	Time period
Söderhamn coastline	County Administrative Board of Gävleborg	1997–2003
Gävle Bight	County Administrative Board of Gävleborg	1997–2006
Stockholm archipelago	Svealands kustvattenförbund, SKVVF (pers. comm. Walve)	2001–2006
Finnish Archipe-lago Sea	University of Turku (Virtasalo, 2005)	2001
Gulf of Riga	ICES oceanographic database (http://www.ices.dk/ocean/)	1997–2007
Lithuania	ICES oceanographic database (http://www.ices.dk/ocean/)	1997–2007
Kalmar Sound	SHARK/SwedODC (http://www.smhi.se/klimatdata/oceanografi/Havets-fysik-kemi-och-biologi)	1997–2007
Hanö Bight	*SHARK/SwedODC (http://www.smhi.se/klimatdata/oceanografi/Havets-fysik-kemi-och-biologi)*	1997–2007

Italic text font indicates the regions where the salinity gradient was successfully improved and hence incorporated in the final refined salinity layer.

Figure 30. An outcrop of the salinity information for Stockholm archipelago as one of the regions where the mean-salinity raster (left panel) was successfully refined into a interpolated product by addition of measured field data (right panel)



4.3.5 *Results*

The resulting habitat map for bladderwrack is presented below (Figure 31). The most important variable in the model was depth, followed by salinity and sand. Fetch had a very low contribution to the model (see Table 11 for variable contribution to the model). The external validation of the prediction was poor (AUC=0.62) suggesting that the map should be used with great caution and mainly as a large scale illustration of Bladderwrack habitat. As guidance for interpretation of the validity of a model, statisticians (Hosmer and Lemeshow 2000) consider models with AUC values greater than or equal to 0.7 as acceptable. However, on top of such recommendations one must take into consideration the distribution of the data as well as the quality of the environmental layers. Some of the problems and limitations in the use of the grid are presented in the discussion below. One important limitation refers to scale. This prediction is a large scale prediction and does not hold for zooming in to more detail. The layer is delivered in a package together with a layer file that should be used together with the grid. In order to guide potential users of the grid to what areas are more reliable than others a confidence layer was made (Figure 32). This layer should be used together with the habitat layer if it is to be used in planning exercises such as MarXan, in order to weight the information according to spatial differences in quality. The area is divided into different quality classes based on data distribution, the quality of the environmental layers and expert judgment of the final prediction. The interpretation of the different classes in the confidence shapefile is listed in Table 12.

Table 11. Variable contribution in the bladderwrack model

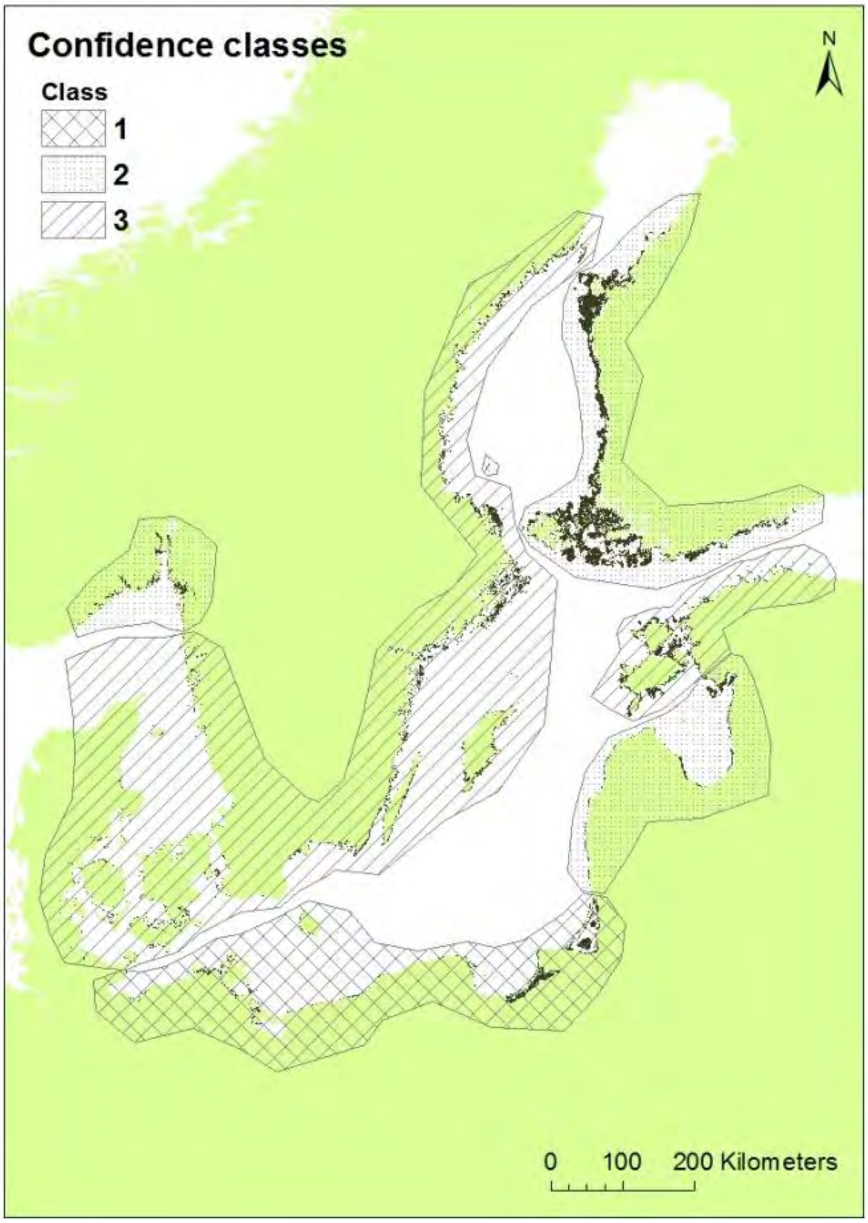
VARIABLE	CONTRIBUTION (%)
Depth	74,7
Salinity	15,9
Sand	7,8
Fetch	1.6

Table 12. Quality classes for the confidence shapefile

CLASS	QUALITY
1	Poor quality- i.e. no or poor data distribution, area with low quality in environmental layers, obvious error in the prediction
2	Intermediate – i.e. somewhere in between the quality of poor and decent based on the same criteria
3	Decent quality – i.e. good data distribution, areas with decent environmental layers and/or no obvious errors

Figure 31. Suitable habitat for bladderwrack (Fucus vesiculosus).The model lack information on substrate in a resolution applicable for species modeling which entails that the prediction must be interpreted as probability of Fucus vesiculosus presence provided there is appropriate hard substrate. External validation was poor (AUC=0.62) which calls for caution when using the map. This is why a warning message has been attached

Figure 32. Confidence layer of the bladderwrack prediction map based on data distribution, the quality of the environmental layers and expert judgment of the final prediction



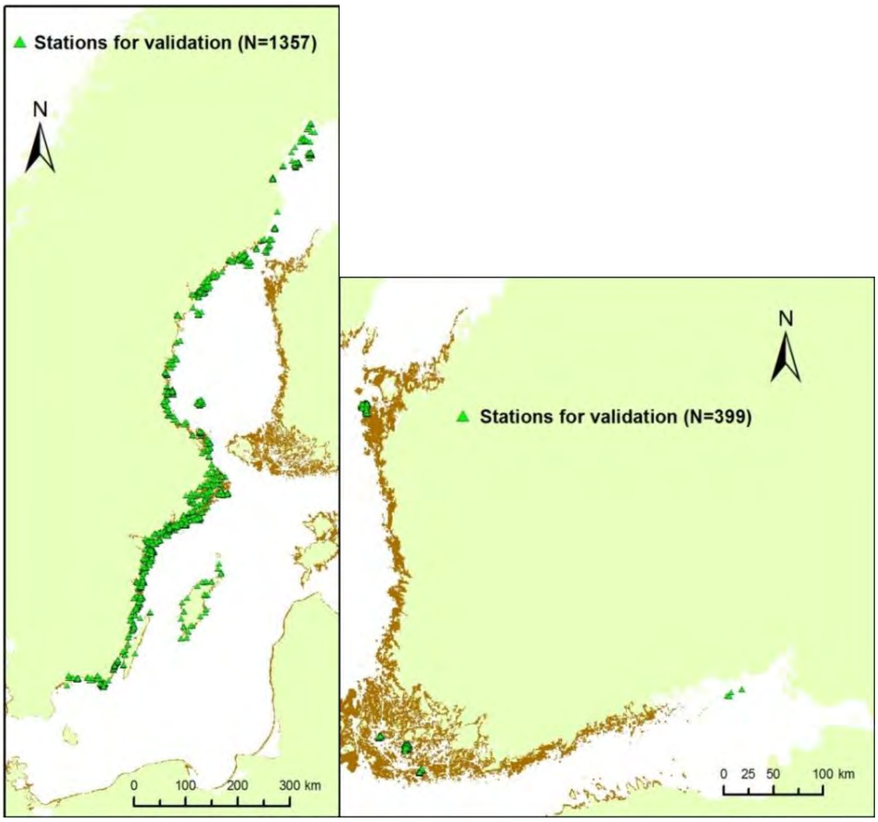
4.3.6 Validation in different areas

The prediction accuracy of the model was also validated in four different areas separately, Finland, Swedish east coast, Swedish west coast and Denmark. The validation results were poor in Sweden (AUC 0.5 for east coast and 0.66 for west coast), good in Finland (AUC 0.81) and excellent in Denmark (AUC 0.9). The results differed between areas for a number of reasons:

- The quality of the prediction
- The geographical distribution of the validation data points
- The prevalence of bladderwrack in the validation data sets
- The number of validation data points

The Swedish east coast was geographically well covered with validation data (Fig. 30). The poor validation result is probably due to the quality of the prediction (see discussion). The validation result in Finland is not reliable since the validation data are limited to a few small areas (Figure 33). The quality of the prediction is good in these particular areas but the result does not tell anything about the rest of the Finnish coast.

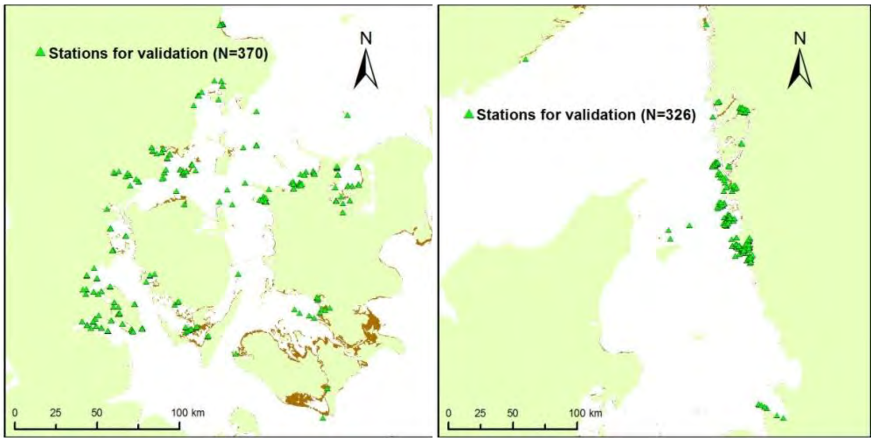
Figure 33. Presence and absence bladderwrack observations used for validation of two separate areas



To the left the Swedish east coast and to the right Finland. The geographical representativeness of the data varied between the two areas.

Even though the Danish area was well covered with validation data, the result is not reliable (Figure 34). In the 370 validation data points there were only 8 presences of bladderwrack. The AUC measure is rather sensitive to low number of presences. Validation points are also absent in areas where the most suitable habits are predicted. The validation results at the Swedish west coast were rather poor probably indicating poor prediction accuracy.

Figure 34. Presence and absence bladderwrack observations used for validation of two separate areas



To the left the Danish east coast and to the right – the Swedish west coast. In Denmark validation data points are missing in the predicted most suitable habitats for bladderwrack. Large parts of the Swedish west coast are missing validation data.

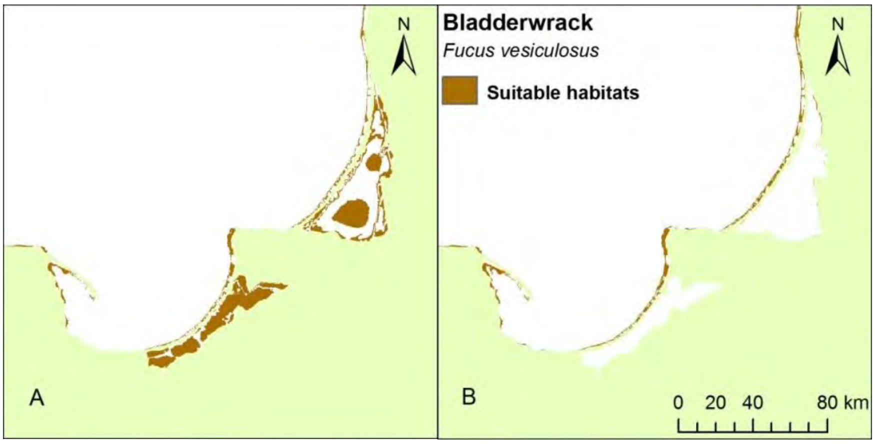
4.3.7 Discussion

Since the external validation was poor the habitat map should be used carefully and with knowledge about the problems in the basic data that built the model. The biological data is unevenly geographically distributed in the project area. It is biased towards some countries which have provided more data. A large part of the dataset is also biased towards areas believed to support high environmental values. In addition there are many errors in the prediction layers and one important structuring factor, i.e substrate, is missing as prediction layer. This entails some limitations in the use of the grid.

All prediction layers suffer from different errors which will be propagated into the prediction. The salinity layer is in a very coarse scale and does not take into account small-scale variability in coastal areas, except in subareas where it has been improved within this project. One example of an area where the salinity layer has been improved is shown in Figure 35. These bays, situated at the coasts of Poland, Russia and Lithuania, are almost entirely enclosed providing fresh water habitats not suitable for bladderwrack. The interpolation behind the original salinity layer did not take into account the narrow strips of land limiting inflow of more saline water into the bays, thus predicting the same salinity level inside as outside the bays. The prediction in 10 A is based on the original salinity layer (from the Balance project) and the prediction in 10

B is based on the improved salinity layer where new salinity data from the bays were available.

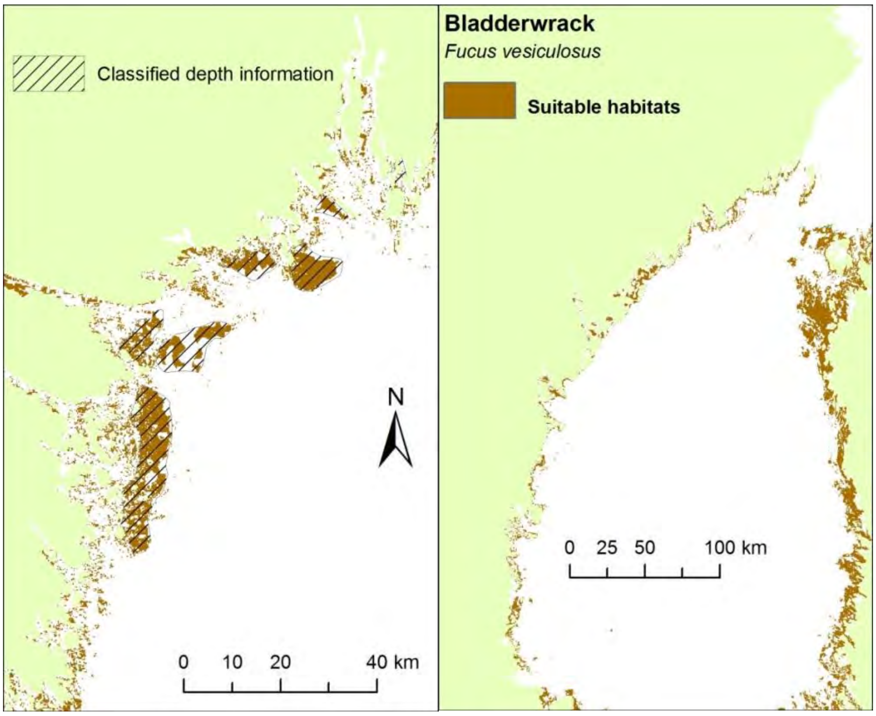
Figure 35



(A) Prediction from the bladderwrack habitat model in two almost enclosed bays at the coast of Poland, Russia and Lithuania based on the original salinity layer. (B) Prediction based on the improved salinity layer where salinity data from inside the bays were available. The fresh water conditions inside the bays are not suitable for bladderwrack. This important information was not available in the original salinity layer.

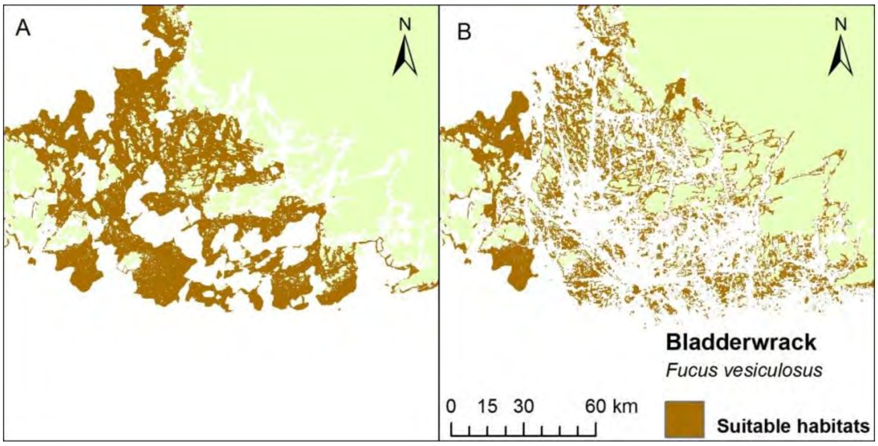
The bathymetry layer suffers from many different errors. The Swedish coast for example has large areas with classified depth information in which the depth is labeled 6–200 meter in the nautical chart and hence comes out as 6 meters depth in the bathymetry (Figure 36). The depth grids are often constructed based on nautical chart information which tends to show depths shallower than what it actually is since this ensures safe navigation. Hence the probability of vegetation occurrence is over-predicted in many areas. Another problem is the coarse scale of the bathymetry from the Balance project. This problem becomes obvious when looking at the prediction the coastal areas of Finland. This coast is more shallow compared to the Swedish side but the variation in depth at a smaller scale are not represented in the layer which makes bladderwrack over-predicted along the Finnish coast (Figure 37A). In the south eastern part of Finland a depth grid in higher resolution was available. This information considerably improved the prediction in this area (Figure 37B). To improve habitat mapping there is a need for more accurate depth data in a higher resolution than provided in the nautical charts.

Figure 36



Left panel: An archipelago area south of Stockholm, Sweden. Brown areas indicate suitable habitat for bladderwrack. Hatched areas are areas with classified depth information. Right panel: A comparison between bladderwrack predictions at the Swedish coast to the left based on higher bathymetry resolution and the Finnish coast to the right with lower bathymetry resolution.

Figure 37



(A) Predicted suitable habitats for bladderwrack in the south eastern part of Finland based on bathymetry from the Balance project. Large areas, too deep for bladderwrack, are predicted as suitable habitats due to the coarse scale of the bathymetry layer. (B) Predicted suitable habitats for bladderwrack in the same area based on a higher resolution bathymetry. The prediction is more accurate and the map could probably be used for management on a more local scale.

The magnitude of errors in the environmental layers is often scale dependent. They can be quite valid on a large scale but does not hold for zooming in to regional scale. One example is the sediment layer from balance that is very erroneous when applied to bladderwrack distribution. However, at a large scale the model is quite accurate even without information on substrate since this variable is most important on a more regional scale where it is possible to distinguish between hard and soft substrate that often occurs in a complex way in the same area. On a smaller scale a better substrate layer would have increased the model´s ability to discriminate between suitable and unsuitable areas, which would have made the map valid also at a regional scale. At a larger scale the variables salinity, depth and sand should be enough to give a broad picture. The predictive ability of the model is very good (AUC=0.946). However, when validating the model with presence/absence external data the model performs poorly (AUC=0.62). This is partly explained by the fact that the model is somewhat over fitted, i.e. fitted too close to the data and not able to generalize to the whole predicted area. The problem of over fitting in this case is mainly due to the model being fitted to data that is unevenly geographically distributed and does not represent the whole area in a correct way. However, the major problem is the poor quality of the predictor layers available for the bladderwrack prediction. This problem can not be solved until more accurate maps for Baltic environmental conditions are available. The most obvious shortcomings are found in substrate and bathymetry maps, but also in the too simplified wave exposure estimate used here, and the uneven accuracy of the salinity grid. The model outcome cannot be better than the input! Biological data as well as the prediction layers suffers from many shortcomings which will be propagated in the model and subsequently in the prediction. This modeling exercise has been most valuable to shear some light on this problem.

One possible solution to handle obvious errors in the final map is different kinds of ad hoc corrections. However, such solutions will only correct what is recognized in the final habitat map as errors but the basic environmental layers will still to be poor. In this way every prediction must be assessed by experts and corrected which is a time consuming and subjective business. A better solution is to put effort on the improvement of the different environmental layers to make them more readily useful for planning both as basic data and as input for more valid species distribution models.

In summary one of the major problems with this prediction is that the model lack information on substrate in a resolution applicable for species modeling. Therefore, the result must be interpreted as probability of bladderwrack presence *provided* there is appropriate hard substrate. The biological data is very unevenly geographically distributed and although some actions has been taken to reduce this unevenness the model will be biased towards the information in the available data and predicts some areas better than others. Furthermore, the different environmental layers are all very spatially uneven in accuracy. This entails that the quality of the prediction is also very spatially uneven, and the quality of every subarea is due to different combinations of errors from environmental layers as well as the data distribution.

The map should only be used as an illustration of bladderwrack distribution in the Baltic Sea and with caution for large scale planning exercises. However, it does not hold for zooming in to local scale planning. For such activities detailed modeling should be used based on more accurate input layers and evenly distributed field data both environmentally and geographically.

During the last few years more accurate depth information has become available in regions of Sweden. This data has been used in smaller scale habitat modeling and the resulting prediction maps are far more accurate than maps based on nautical chart data. Within the 3–4 years higher resolution depth data will be available for the entire Swedish coast if the national program continues as planned.

4.4 Habitat models for blue mussel *Mytilus*

4.4.1 Introduction

The potential distribution of blue mussel *Mytilus edulis/trossulus* in the Baltic Sea was modeled using a modified version of DHI's deterministic filter-feeder model (Møhlenberg & Rasmussen in press.). Thus, the modeling strategy differed from that used for modeling coverage of macroalgae and *bladderwrack* by estimating growth of blue mussels directly on the basis of the available food supply (phytoplankton) modeled by DHI's ecosystem model "BANSAI" for the Baltic Sea. BANSAI has been operational since 2000, and provided yearly estimates of the supply of phytoplankton to mussels during the growth season (March – October) up to 2007 at a resolution of 3 nautical miles.

The *Mytilus* model design was based on three model elements:

- A regional and local hydrodynamic model
- A bio-geochemical model, and
- A deterministic filter-feeder model

4.4.2 Hydrodynamic model

Several numerical 3D flow models have been established within the MIKE modeling framework covering the North Sea and Kattegat. Each of these models has individual strengths. With the purpose of water quality modelling, the so-called BANSAI model (DHI 2006) was chosen as it has been running operationally since 2001. The model provides input data with regard to the flow field and water quality, and consists of two parts:

- A hydrodynamic module for calculating the evolution in water levels, currents, salinity and water temperature
- An ecological module that calculates the spreading of nutrients, the primary
- production, the biomass, and other ecological parameters

Originally the BANSAI model was created in a collaboration between the Swedish Meteorological and Hydrological Institute (SMHI, Sweden), Finnish Institute of Marine Research (FIMR) and DHI.

The model is using DHI's 3-dimensional model system MIKE3 Classic, which is a fully three-dimensional, non-hydrostatic, primitive equation model (Rasmussen, 1991). It is based on the Reynolds-averaged Navier-

Stokes equations and the conservation of mass, salinity and temperature. The prognostic variables are fluid pressure, the three velocity components and the two scalar quantities salt and temperature.

The model represents the water column with a 2 m resolution. The model is operational and based on:

- Meteorology
- Tide, salinity-, temperature and nutrients on the edge of the Atlantic (tide from tidal constituents, salinity and temperature from monthly climatology (ICES), nutrients from climatology supplied with national monitoring data from Denmark and Germany
- Runoff and nutrient loadings from land (runoff from monthly climatology from HELCOM, OSPAR, national monitoring data) and nutrient loadings from climatology supplied with national monitoring data

The model was first calibrated based on measurements from the year 2000 and has been continuously improved since then.

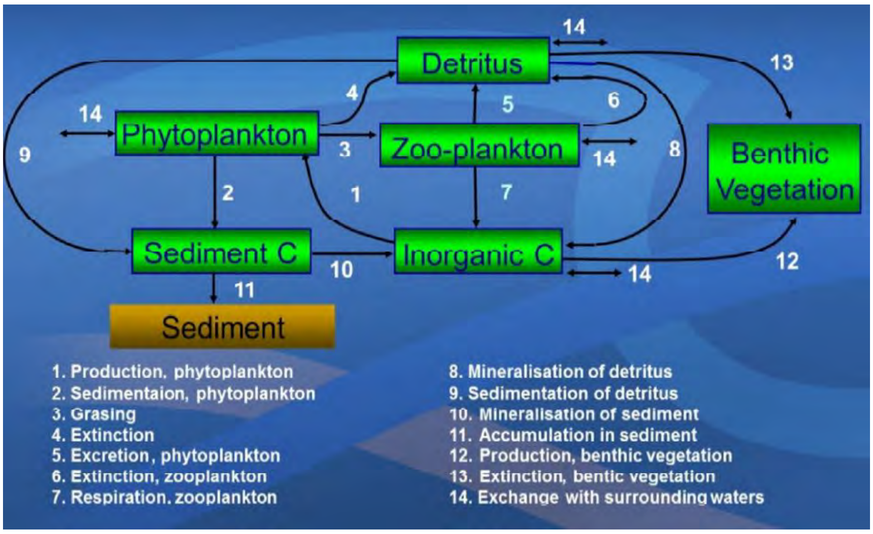
4.4.3 Ecological model

The ecological model consists of an eutrophication model describing the pelagic system with 13 state variables, and seven state variables describing the exchangeable Nitrogen and Phosphorous pools in the sediment (Rasmussen et al., 2009). The pelagic system includes phytoplankton, described in terms of their concentration of carbon (C), nitrogen (N) and phosphorus (P), chlorophyll-a, zooplankton, detritus (C, N & P), inorganic nutrients (dissolved inorganic nitrogen–DIN & PO4–P), total N and P nutrients (including dissolved organic N and P compounds) and dissolved Oxygen (DO). In addition to state variables a large suite of derived variables such as water transparency and Secchi depth is modelled and stored during the modelling process. Benthic organisms are not modelled explicitly, but are included as a forcing in the water quality model. Filter-feeding bivalves constitute on average 93% of the entire biomass of benthic invertebrates in the areas, and their filtering activity can exert a significant grazing loss on phytoplankton. Their effect is included in the model by imposing a filtration loss on phytoplankton and detritus in the near bed model layer according to the filtration capacity calculated from length distribution and total biomass of the different species. Because bivalves are not included as a state-variable they do not participate directly in nutrient cycling and accordingly, 50% of filtered algae (C,N,P) are returned as inorganic solutes to the near-bed layer and

50% are entered into the detritus pool subject to sedimentation and remineralisation. Figure 38 shows the state variables and processes for carbon (C) for the pelagic system.

The ecological model was built using the generic equation solver ECOLab that functions as a module in the MIKE 3 simulation software, and ECOLab is linked to the advection dispersion term of the hydrodynamic flow model, enabling transport mechanisms based on advection-dispersion to be seamlessly integrated into the ECO Lab simulation. Forcings and boundary conditions of the water quality model follows the line of the forcings and boundaries of the hydrodynamic model, but in addition values for all pelagic state variables at boundaries and nutrient concentrations in freshwater loads (monthly basis) in addition to atmospheric loads are included. Boundary values are forced with water quality data extracted from the BANSAI model.

Figure 38. Schematic diagram showing state variables and processes for carbon in the ecological model established to simulate water quality



4.4.4 Model set-up

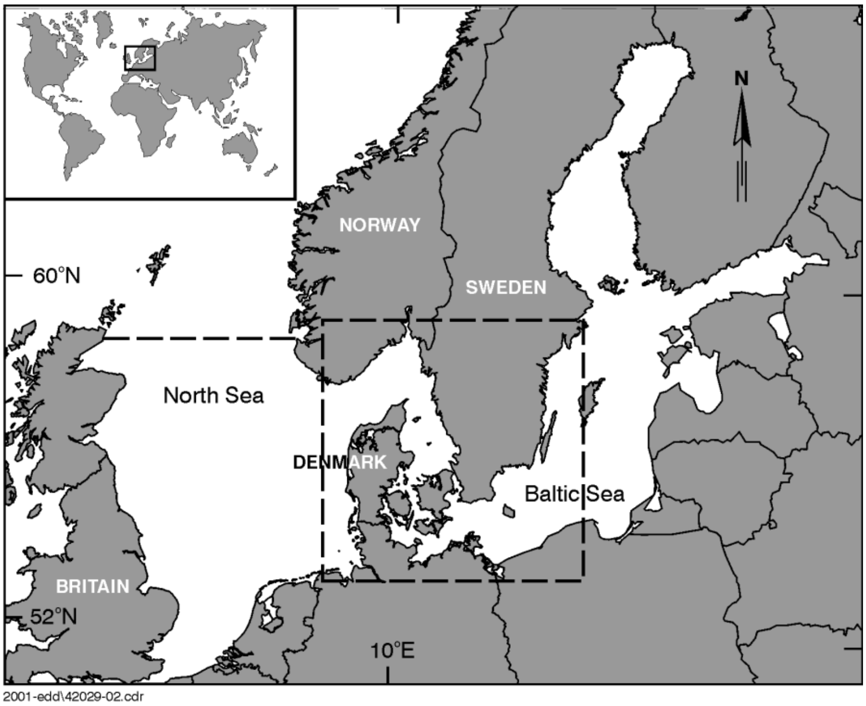
The model area covers the entire Baltic Sea, the transition area and the North Sea, with an open boundary to the north between Stavanger and Scotland (latitude app. 59°) and an open boundary to the south-west in the English Channel (latitude app. 51°). The model area is shown in Figure 39.

The applied model is set up with a horizontal resolution of 1 nm (nautical miles) in the western Baltic and 3 nautical miles in the eastern

Baltic, see Figure 39. The bathymetry of the model domain is based on available surveys. However, some adjustments have been made to ensure an appropriate representation of the deeper trenches, which are very important for the formation of the stratification.

MIKE 3 is a z-layered model, and the vertical resolution in this setup is 2 meters between –6 and –220 m, giving 110 z-layers. Below –220 m a bottom-boundary fitted approximation is applied to the remaining part of the local water column. The surface layer varies with the free surface elevation and extends down to level –6 m (or local depths at shallower depths). The model is run with a time step of 300 sec.

Figure 39. The regional model area



4.4.5 Boundaries, forcings and runoffs

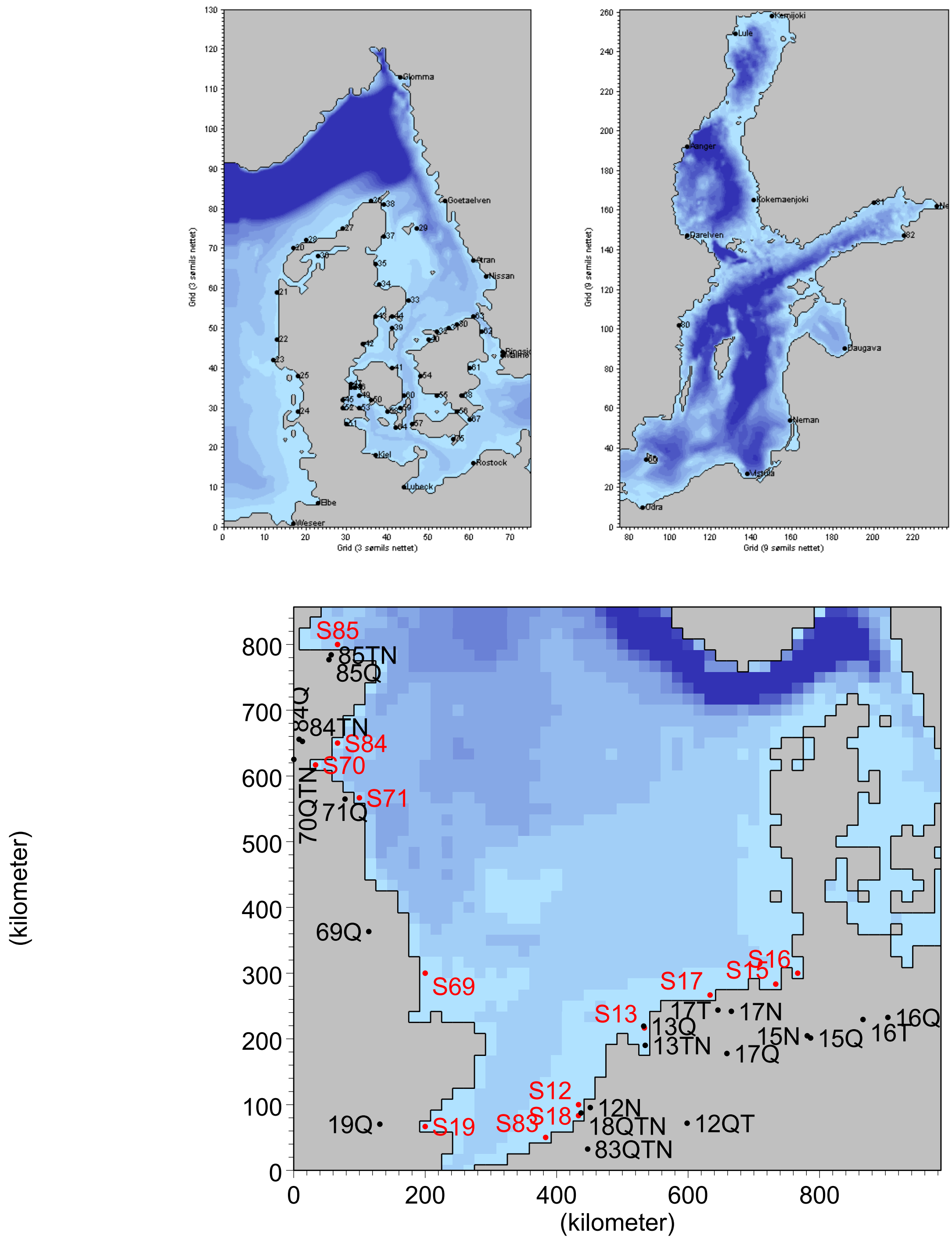
To run the hydrodynamic model, external forcing, boundaries, and initial conditions are required. The required data and their origin are listed in Table 13. The runoff to the model domain is represented by 85 source points (Table 13). The position of the runoff sources are illustrated in Figure 40. Load compilation data from around year 2000 from HELCOM and OSPAR has been applied where actual data has not been available.

Table 13. Data required to run the Mytilus model

HYDRODYNAMIC MODEL	Data origin
Open marine boundaries Astronomical tides (corrected for actual atmospheric pressure)	
Climatological[1] values of temperature and salinity distribution in sections (linearly interpolated to cover the entire sections)	ICES[2] database (on request)
Initial fields	
North Sea and Baltic Sea: Salinity; temperature	ICES database (on request)
Interconnecting Seas: Salinity; temperature	National centre for Environment and Energy (former NERI[3] (MADS database)
Run-off Actual monthly values of flow for rivers to Skagerrak-Kattegat and the Belt Sea	National centre for Environment and Energy (former NERI)[3] SMHI)[4] and IMR)[5] (on request)
Actual daily; weekly or monthly values of flow; water temperature and nutrients (N and P) for German; Dutch and English rivers[7])	HYDABA,[18] NLWKN[9] (on request) Service Desk Data,[10] CEH[11] (on request) Environment Agency[12] (on request) SEPA[13])
Climatological[1]) values for the remaining rivers (Belgium; Germany; Poland; Norway; Russia; Finland; Sweden)	HELCOM/OSPAR
Air-sea exchange	
Climatological[1] values of net precipitation	
Actual 3-hourly 10 m wind and air pressure fields	HIRLAM; DMI)[6]
Actual 3-hourly 2 m air temperature fields	HIRLAM; DMI)[6]
Climatological[1] clearness information	

1) 10 years of monthly mean.
2) International Council for the Exploitation of the Seas, see http://www.ices.dk for further information.
3) National Environmental Research Institute, see http://www.neri.dk for further information.
4) Swedish Meteorological and Hydrological Institute, see http://www.smhi.se for further information.
5) Institute of Marine Research, see http://www.imr.no for further information.
6) Danish Meteorological Institute, see http://www.dmi.dk for further information.
7) These data were updated in this study (Table 4–2).
8) Hydrological database. Federal Institut of Hydrology.see. http://www.bafg.de for further information.
9) Niedersächsischer Landesbetrieb für Wasserwirtschaft, see http://www.nlwkn.de for further information.
10) Rijkswaterstaat Centre for Data and ICT, see http://www.waterbase.nl for further information.
11) National River Flow Archive. Centre for Ecology & Hydrology, see http://www.ceh.ac.uk
12 Environment Agency, http://www.environment-agency.gov.uk
13) Scottish Environment Protection Agency, see http://www.sepa.org.uk for further information.

Figure 40. Runoff positions for the 85 freshwater sources



4.4.6 *Blue mussel CC model*

A carrying capacity (CC) model was established using the output from the hydrodynamic and ecological models. The CC model built on the a concept of combining a physiology-based growth model for a standard individual with an advection term that replenish the food ingested by filter-feeders. On large scale CC depends on the local primary production and on smaller scale current speed plays an increasing role for CC.

The energy balance of a filter-feeding bivalve can be expressed as: $I = P + R_t + F$, where I = ingestion; P = growth, R_t = total respiration (sum of maintenance respiration, R_m, and respiratory cost of growth, R_g), and F = excretion. Rearranging, growth is expressed as $P = I \times AE - (R_m + R_g)$ or $P = (F \times C \times AE) - (R_m + R_g)$, where $AE = (I - F)/I$ = assimilation efficiency, F = filtration rate, and C = algal concentration. In the individual bivalve growth depends on the quantity (C) and quality of suspended food particles including different species of algae, ciliates and zooplankton organisms along with suspended inorganic material (silt). The maintenance food concentration (which just is sufficient for zero growth) and the maximum growth rate for a standard-sized bivalve differs between species and between populations within species as result of adaptation to local composition and concentration of food (Kiørboe at al. 1980, Kiørboe & Møhlenberg 1981a).

Overall the production of benthic suspension-feeders is limited by food availability that in turn can be described by phytoplankton concentration and advection in the near bottom layer (Wildish and Kristmanson, 2005). We used a combination of a quadratic function of modelled current speed (vrms) and a functional response between phytoplankton concentration and individual mussel growth to express an index of carrying capacity (CCI) for benthic suspension feeders:

CCI = a " (PhC – Fm)/(PhC + PhC1/2 – PhCm) (-4 " vrms2 + 4 " vrms)*

where a is a scaling factor, PhC is the concentration of algae (mg C m-3), PhC1/2 (= 180 mg C m-3) and PhCm (= 30 mg C m-3) is the interception of the growth curve on the x-axis (i.e. the concentration that just maintains biomass) (Møhlenberg & Rasmussen, in press). The CCI model takes account of maintenance concentration (30 mg C m-3) and food-satiation and uses a dome-shaped (quadratic) current function (i.e. at current speeds above 0.5 m s-1 feeding process is impaired or mussels are swept away). Daily CCI values from 1 April through October were averaged to yearly means for the years 2000, 2002, 2004 and 2005 for the Baltic.

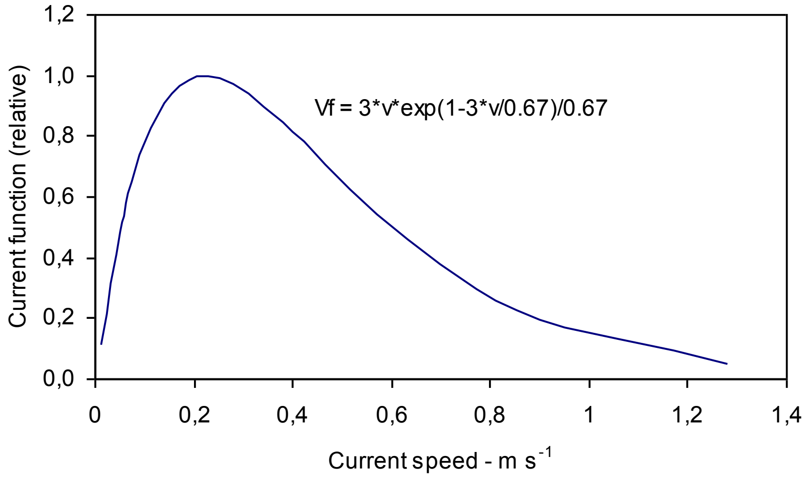
The growth functions described above relate to individual bivalves surrounded by food at constant concentrations. In nature, filter-feeding bivalves aggregate in dense assemblages if current speeds are high, e.g. in tidal areas as the Wadden Sea. In low-current environments plankton algae removed by filtration are only slowly replenished and such environments cannot sustain dense populations. Therefore, the growth functions need to be supplemented by an equation that describes the replenishment of food.

Effect of current speed on growth in individual filter-feeding bivalves has rarely been studied. One example relates to giant scallop (*Pectindae*), where growth rate increased until an optimal speed of 0.15 m s^{-1}, but at larger current speeds the growth decreased as currents interfered with filtration behavior (Wildish et al. 1987). In *Mytilus in situ* growth rate increased with current speed (Riisgård et al. 1994) and wind-induced turbulence (Sand-Jensen et al. 1995). We are not aware of studies where an optimum current speed has been identified, but it is likely that bivalves in benthic environments consisting erodible substrate such as sand cannot maintain their position at current speeds larger than 0.6–1.0 m s^{-1}. To that end we have constructed a bell-shaped current function with an optimum speed at 0.3 m s^{-1} (Figure 41).

The individual growth function can then be combined with the current function to a carrying capacity index reflecting both individual growth conditions and the density of bivalves that can be sustained:

CC index = Gf " Vf

Figure 41. Current function to describe food replenishment and physical stress in filter-feeding bivalves



Controlled experiments of the effects of current speed on growth have only been carried out on oysters, which showed an increase until an optimal current speed of 15 cm s^{-1}, after which the growth started decreasing. Other bivalve species such as blue mussels increase growth in the field with increasing current speed and wind-induced turbulence until a plateau. This is generally interpreted as a consequence of increasing food availability. Mussels which are settled on substrate like cliffs, stones and foundations may survive and grow in even very energy rich environments (e.g. in current speeds > 60–80 cm s^{-1}), while blue mussels on sandy sediments are unable to establish long-living populations at current speeds exceeding 40–50 cm s^{-1}, probably as a result of erosion.

Based on their distribution *Mytilus* seems to prefer full salinity but tolerate reduction in salinity to 4 psu. To account for influence of salinity in CC-indices we have applied functions based on the information available. Specifically, at salinities below 4 psu *Mytilus* is excluded and at salinities above 30 and 35 psu they are not influenced by variations in salinity. Linear function are applied between these two extremes.

The carrying capacity index was included in the geo-biological model as a 3-dimensional variable. Following the model simulations the 3-dimensional coarse (3 nm) index was transformed into 2-dimensional and finer grids (1/3 nm or 617 m) describing the actual bottom index-values in the corresponding depth of the 617 m bathymetry. However, since the surface layer in the geo-biological model is 5 m, no variation was found in shallow areas. The index for depth between 0–5 m was therefore corrected with a function describing increasing food availability with increasing depth:

$$CCindex_{0\text{-}5m} = CCindex_{0\text{-}5m} * 1.75 * (depth+0.15)/(depth+4)$$

Figure 42. Modelled CCIndex for Mytilus in southwestern Baltic 2000 and 2006

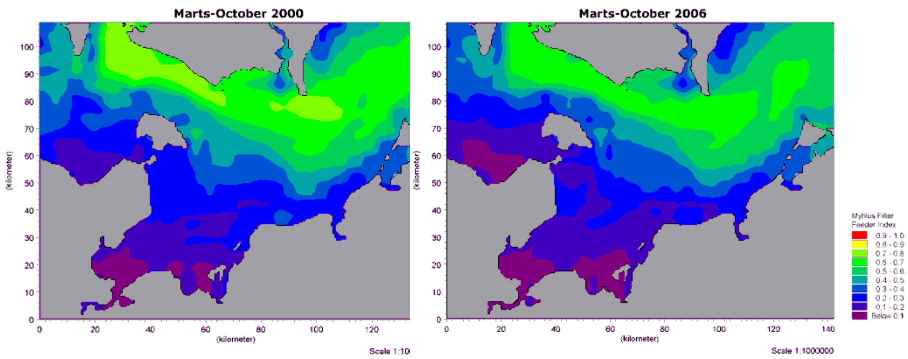


Figure 43. Time series of modelled CCIndex for Mytilus in Great Belt 2000–2006

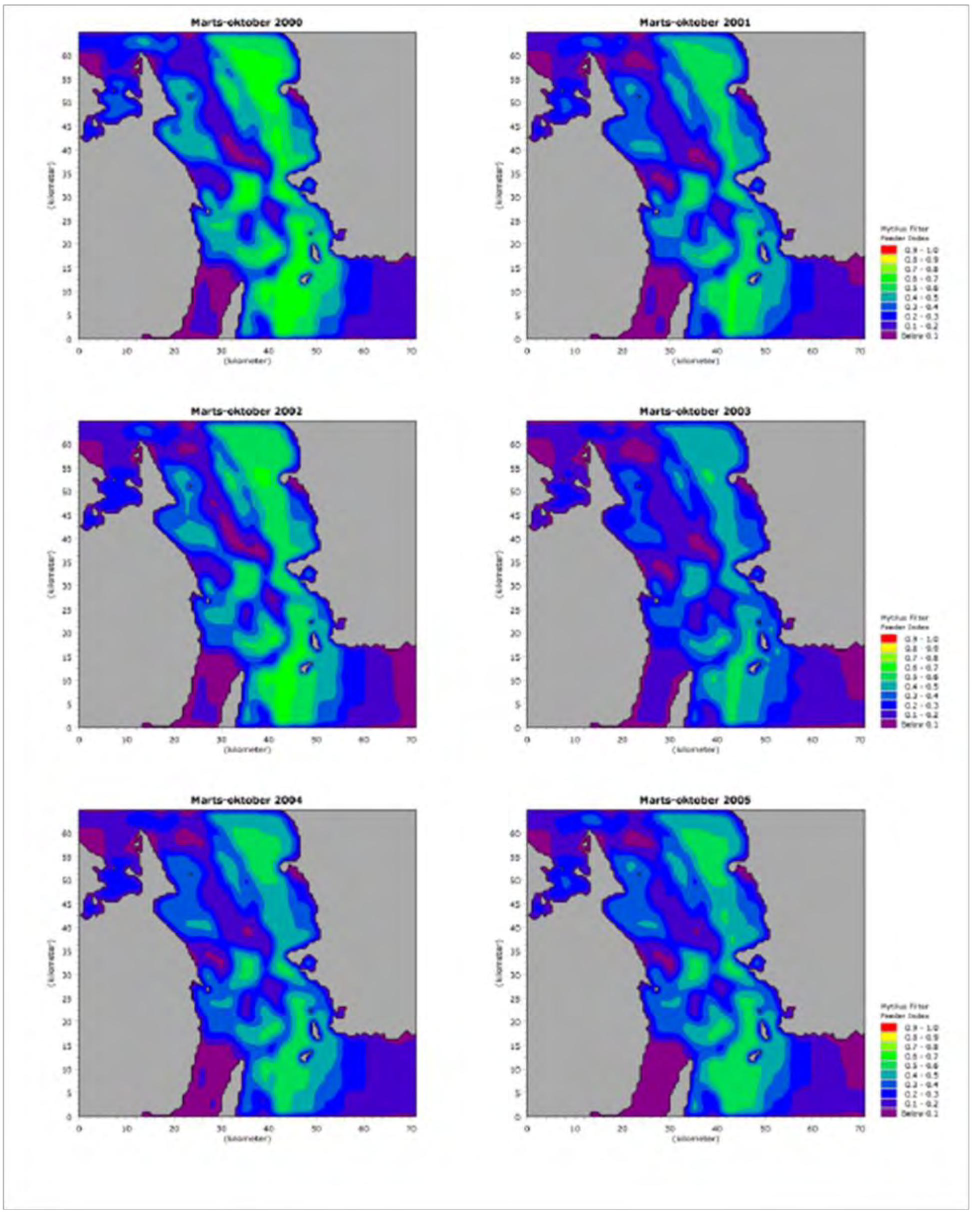
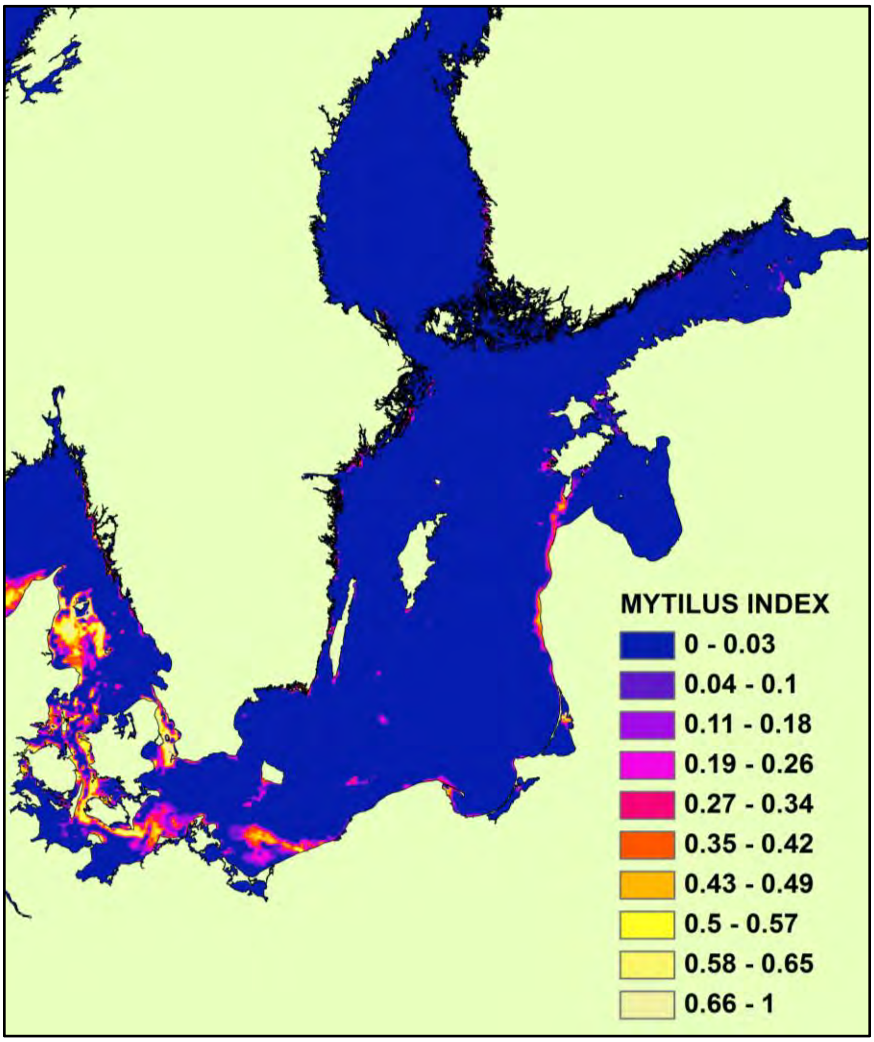


Figure 43. Mean modelled CCIndex for Mytilus in the Baltic Sea 2000–2006



4.4.7 Validation of blue mussel model

The blue mussel model was validated using independent coverage data collected by Hans Kautsky, Stockholm University from a range of locations along the Swedish east coast between 2005 and 2007 and by the National Centre for Environment and Energy (former NERI) in the Danish part of the Kattegat between 1992 and 2009 (Figures 44, 45). We used the ROC (Relative Operating Characteristic) approach to assess the validity of the modelled CC values; i.e. by comparing the modelled CC class (on an arbitrary scale from 0 to 100) with the presence of blue mussels at each location. The coverage threshold indicating presence of mussels was set to 10%.

We first used the ROC technique to explore how well the different categories of modelled carrying capacity matched with the presence of mussels. This was done by calculating AUC values for a range of predictions using lower thresholds of CC from 8 to 50 (after re-scaling to 100, Figure 46). According to Hosmer & Lemeshow (2000), AUC values exceeding 0.8 reflect a good model. Accordingly, CC index values lower than 30 seem to provide the best match with the observed presence of blue mussels (Figure 46).

The ROC plot for the model using external validation data from the eastern Swedish coast and a threshold for presence at a CC value of 30 is shown in Figure 47. The test indicates that the deterministic mussel model performs very well in the Baltic, and accurately predicts the areas with a coverage of blue mussels exceeding 10%.

The ROC plot for the model using external validation data from the Danish part of the Kattegat is shown in Figure 48. The test indicates that the deterministic mussel model performs badly in the Kattegat, where the AUC value for predicted coverage of blue mussels exceeding 10% is as low as 0.49. This result was expected however, as predation by sea stars is known to eliminate extensive coverage of blue mussels from areas nortrh of the Belt Sea. As the deterministic mussel model applied here does not include predation processes the predicted potential growth of blue mussels in areas of high abundance of sea stars is overestimated.

Figure 44. Location of validation data from the Swedish east coast on coverage of blue mussels (Courtesy University of Stockholm)

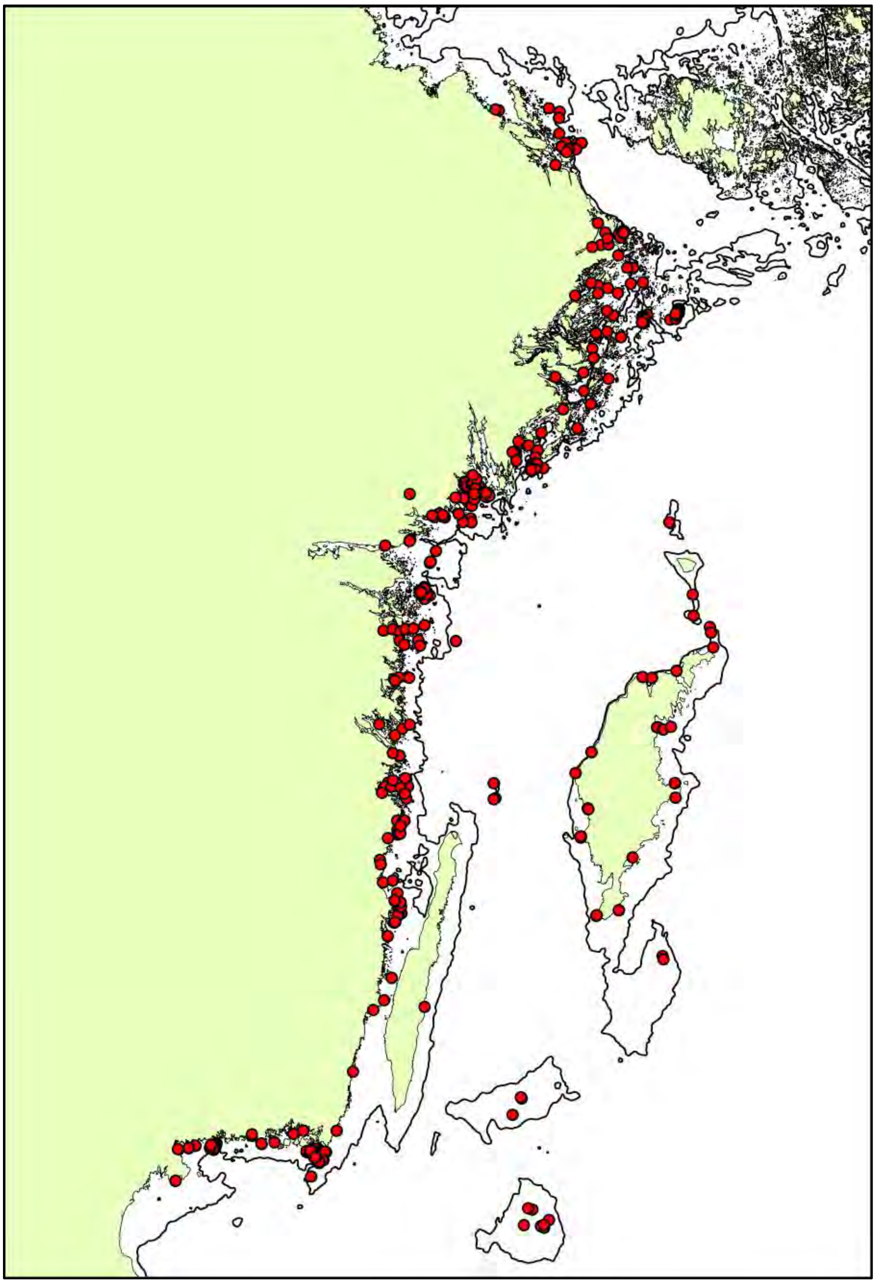


Figure 45. Location of validation data from the Danish part of the Kattegat on coverage of blue mussels (Courtesy National Centre for Environment and Energy)

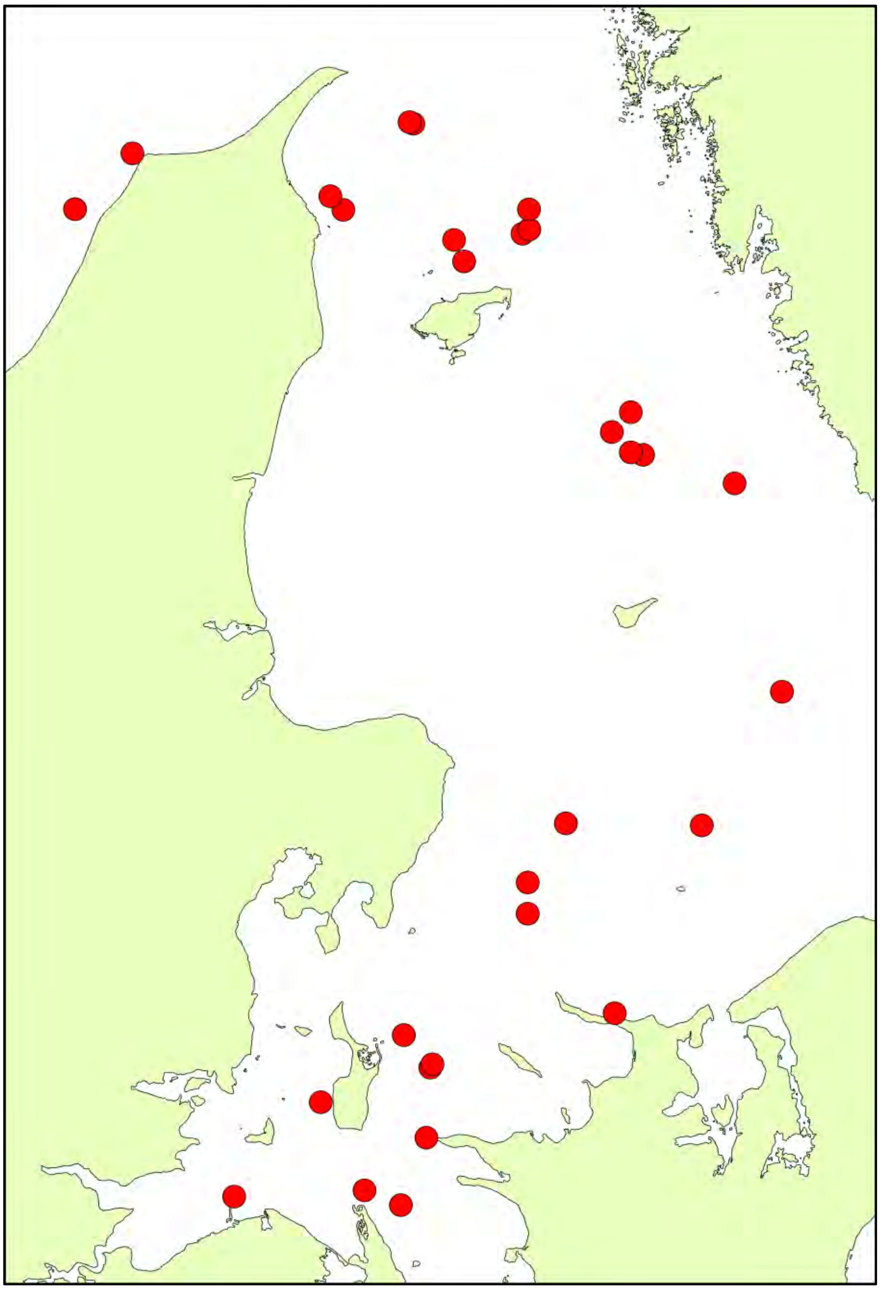


Figure 46. AUC values for ROC tests of recorded mussel coverage vs. modelled CCIndex for Mytilus, undertaken using a range of lower thresholds of the CC Index

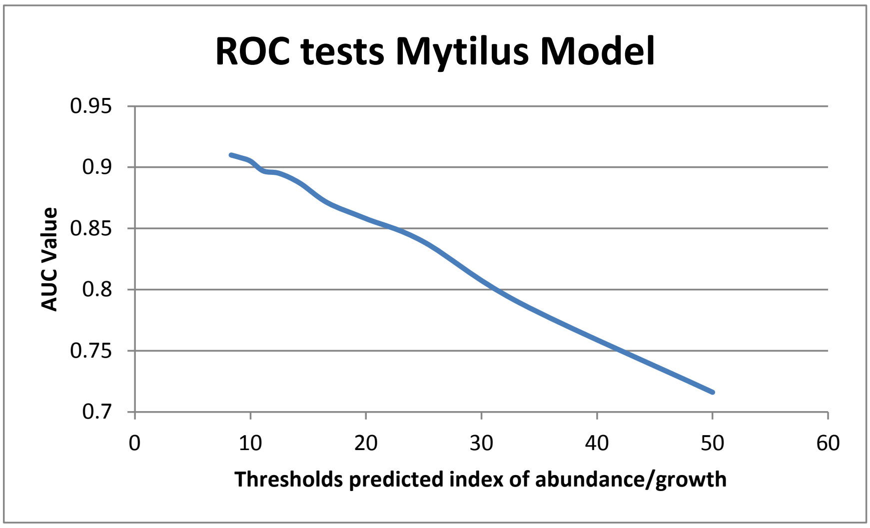


Figure 47. ROC plot of recorded mussel coverage vs. modelled CCIndex for Mytilus along the Swedish east coast, using a lower threshold of the CC Index of 30

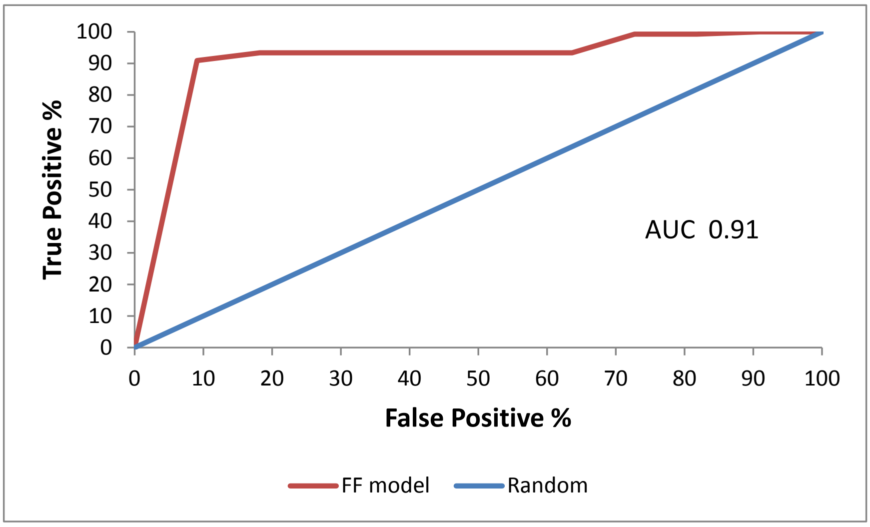
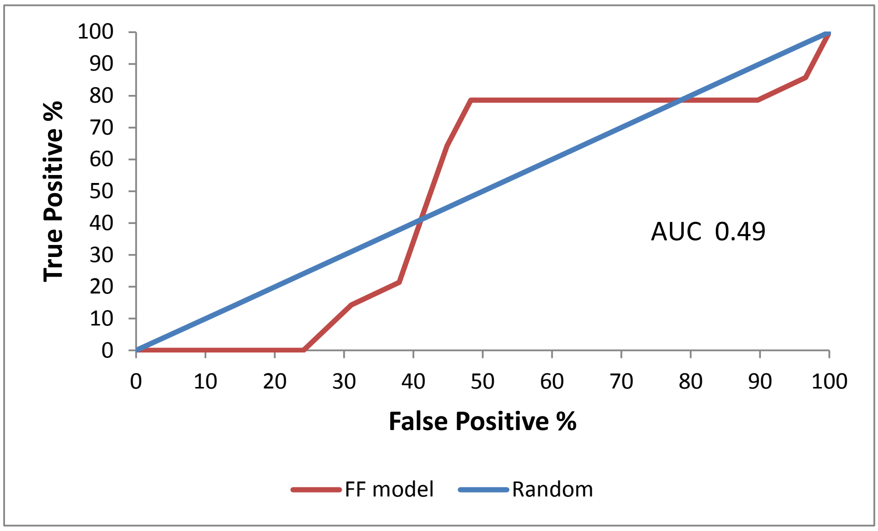


Figure 48. ROC plot of recorded mussel coverage vs. modelled CCIndex for Mytilus in the Kattegat, using a lower threshold of the CC Index of 30



4.5 Potentials and limitations in the regional application of marine spatial models

The development and tests of regional habitat models in Work package 2 has provided useful demonstrations of the limitations of the application of different modelling approaches to predict coverage of habitat forming species over large regions. The applicability of using existing site-specific total and cumulative cover models to produce habitat maps for seaweed forests on reef areas in neighbouring sites highlighted the need for high-resolution data on bathymetry and substrate. Even though data on nutrient concentrations were available for the Kattegat extrapolation from the Danish model sites was only possible to a few sites like Lilla Middelgrund.

The limitations of extrapolating spatial vegetation models to larger regions were further highlighted by the bladderwrack model. Even if this model was designed as presence-only model using MaxEnt which is generally rated among the statistical methods with the highest predictive power (Ellith et al. 2006) the resulting map should only be used as an illustration of bladderwrack distribution in the Baltic Sea and with caution for large scale planning exercises. However, it does not hold for zooming in to local scale planning. For such activities detailed modeling should be used based on more accurate input layers and evenly distributed field data both environmentally and geographically.

The potential for predicting the coverage correctly over large spatial areas was obviously greater in the deterministic blue mussel model. The validation tests indicated that the deterministic mussel model could be used to accurately predict areas with a coverage of blue mussels exceeding 10% at a scale of 615 m. This is even so given the lack of sediment data used in the prediction. However, as the model does not include effects of predation by sea stars, in areas of higher salinity in the Kattegat where sea star predation is significant the model perform less well.

The model work has documented that with calibration data the coverage of blue mussel and other suspension-feeding invertebrates can be predicted confidently using deterministic ecological models over large regions of the less saline part of the Baltic Sea where intensive predation by sea stars is not present. Within the short term it is expected that higher resolution depth and sediment data will be available for the entire distribution range of the bladderwrack. This will allow fine-scale habitat models to be developed and applied to predict the coverage of this and other key species of vegetation in the Baltic Sea.

5. Indicators and tools to assess the favourable conservation status

5.1 Conceptual models of indicators

5.1.1 Introduction

The aim was to develop simple conceptual models which describe the linkages between pressures and structure and functioning of habitat-forming species in different habitats. Moreover, the idea was to give some suggestions for characteristics that could be used as common indicators across the Baltic Sea and Nordic countries. The conceptual models have been made for three Habitats Directive's Annex I habitats: reefs (1170), sandbanks (1110) and lagoons (1150). The three most relevant pressures were selected: eutrophication, physical disturbance and climate change. Pressures and their main effects have been selected keeping in mind the Marine Strategy Framework Directive Annex III table 2.

Suggested indicators are species or species groups that are common in the habitat and easy to identify. Their ecology is quite well known and they are known to respond to enhanced nutrient levels in water. It is important to have historical data about the indicators, but if it is not available the use of modeling is an option. Indicators are suggested which are suitable for assessing the favourable conservation status of the habitat, and which may be regarded as robust despite the salinity differences across the Baltic Sea.

5.1.2 Conceptual models and suggested indicators

The structure of the conceptual models

The figures showing the conceptual models are shown in Appendix I. The models display simple descriptions of the most important effects of the pressures on habitats. In the conceptual models are first described pressures and its habitat effects. In the conceptual models of eutrophication also habitat change levels are described: habitat alteration, habitat fragmentation and habitat loss. In the conceptual models of physical disturbance and climate change the habitat change levels are absent because they are more difficult to identify. In the conceptual models of eutrophication different kinds of indicators can indicate the effects on these three habitat change levels. In other conceptual models (physical disturbance and climate change) are only linked indicators and habitat effects. Thicker arrows from the habitat effects to indicators indicate the most important indicators and their links to the habitat effect.

In every conceptual model is a paragraph describing the model and habitat effects. As the structure of the models is very similar in every habitat the models of reefs being first are described more precisely. In the conceptual models indicators are similar in every habitat as "coverage of habitat-forming species" and "depth limit of habitat-forming species." However, these habitat-forming species include always specific species for the habitat at issue.

An example of habitat change levels

In the conceptual model of eutrophication of reefs *habitat is altered* when the biomass of mussel beds or coverage of the perennial macroalgae changes. Biomass of mussel beds can be increased because of increased food supply. Coverage of perennial macroalgae can be changed because of increased grazing. The abundance of grazers increases along increased filamentous algae which serve more food to grazers. The increased abundance of filamentous algae alters also the habitat.

Habitats are fragmented when coverage of perennial macroalgae and mussel beds is patchy, depth penetration of perennial macroalgae is decreased and fraction of opportunistic algae is increased. Moreover, species diversity of macroalgae is decreased. This can be caused by decreased hard substratum, light attenuation and competition with filamentous algae for space.

Habitats are lost (habitat is not suitable for the typical species of reefs) when species composition of macroalgae has been changed; perennial macroalgae are absent and annual opportunistic algae are present, the diversity is low and only few species are dominant. When habitat is lost only few individuals of perennial macroalgae and mussels will be present.

5.1.3 Reefs

Eutrophication (Appendix 1)

The concentration of nutrients increases with the intensification of eutrophication. As a consequence, the abundance of phytoplankton and filamentous algae increases. Depending on the degree of nutrient limitation of phytoplankton the benthic fauna will benefit from the increased phytoplankton abundance and its abundance increases altering habitat. On the reefs this effect is reflected by the indicator "density/biomass of mussels." The abundance of the grazers increases along the increased amount of opportunistic algae and therefore, also grazing of perennial macroalgae increases. Grazing decreases total/cumulative algal cover – see chapter 4.1.7 for details.

One of the habitat effects could be predation of mussels, as observed in the northern Baltic Sea where predation by roach (*Rutilus rutilus*) may play an important role in controlling the abundance of mussels (Lappalainen et al. 2005) and in the Kattegat where blue mussels are intensively predated by sea stars (pers. comm. Dahl).

Increased amount of phytoplankton causes increased sedimentation which might temporarily overlay hard substrat and thus interfere the settling of perennial macroalgae and mussels. The effects can be seen especially in changes of coverage and lower depth limit of perennial macroalgae and in changes of density/biomass of mussels. Therefore the most suitable indicators could be "coverage of perennial macroalgae" (negative response), "depth limit of perennial macroalgae" (negative response) and "density/biomass of mussels" (positive response). Moreover, when sedimentation is strong the indicator "species composition of habitat-forming species" is essential. Increased sedimentation causes mainly habitat fragmentation, but also habitat loss. The loss of hard substratum diminishes the area where perennial macroalgae and mussels can inhabit and this effect can be measured with the indicator "area of reef."

Filamentous algae can occur as epiphytes on the surface of perennial macroalgae. When they are abundant they shade the macroalgae and hinder the light penetration. Moreover, when more phytoplankton occur in the water column the penetration of light decreases and this effect on the lower depth limit where perennial macroalgae can inhabit. The most important effect of the light attenuation is decreased "depth limit of perennial macroalgae" causing habitat fragmentation and this can be used as an indicator. When light conditions are very poor "species composition of habitat-forming species" is important indicator. The species composition of habitat-forming species will change along the light attenuation because species have different adaptation to low light. Lack of light can cause also habitat alteration and habitat loss affecting “total/cumulative algal cover,” “number of late successional species” and “fraction of opportunistic algae.”

Filamentous algae are fast-growing usually annual macroalgae which inhabit the same substratum type than perennial macroalgae and therefore they compete for the space with slow growing perennial macroalgae. This effect can be seen in all habitat change levels and it can be indicated together with different indicators. Competition for space cause the most important effects on “coverage of perennial macroalgae,” “number of late successional species” and “fraction of opportunistic algae” which could be used as indicators. Other potential indicators are “total/cumulative algal cover” and “species diversity of habitat-forming species.”

Physical disturbance (Appendix 1)

The most important pressure affecting increased sedimentation is dredging. It decreases the area of reef covering hard substratum and thus changes the habitat type. Indicators to indicate the effects of sedimentation could be “coverage of perennial macroalgae,” “density/biomass of mussels,” “depth limit of perennial macroalgae,” “species composition of habitat-forming species” and “area of reef.”

Among others dredging, bottom trawling and boating cause increased resuspension of sediment leading to increased turbidity which have effect on light attenuation. Moreover, constructions such as marinas and docks can shade perennial macroalgae. Indicators to reflect the effects of lack of light could be “coverage of perennial macroalgae,” “number of late successional species,” “depth limit of perennial macroalgae” and “species composition of habitat-forming species.”

Dumping of dredged spoil smothers perennial macroalgae and habitat-forming fauna burying them, hindering their growth and decreasing the diversity. Indicators to indicate the effects of burial of habitat-forming species could be “coverage of perennial macroalgae,” “density/biomas of mussels,” “species composition of habitat-forming species” and “species diversity of habitat-forming species.”

Waves and ice scouring are natural factors in shallow sublittoral areas affecting distribution and abundance of species. However, boating and marine traffic can increase the wave action. Waves and ice scouring affect the upper limit of the perennial macroalgae. In the studies of Rönnberg (1981) increased frequent wave action moved upper limits of annual macroalgae higher and moved the belt of *Fucus vesiculosus* deeper by wave and ice erosion. Heavy waves can also dislodge perennial macroalgae and mussels. As reef species inhabit exposured coasts the wave action can also have positive effects as has been found on the routes of the larger car-ferries where water movements are strong and regular keeping the bottoms free from the sediments and drifting algae (Roos et al. 2004). As increasing turbidity, wave action may have also effects on lower depth limit (Eriksson e al. 2004).

One of the most important indicators to indicate the effects of the wave action is "coverage of perennial macroalgae." Increased wave action can cause also mechanical damage by tearing and removing habitat-forming species or even mortality. Other indicators to indicate increased wave and ice scouring effects could be "number of late successional species," "fraction of opportunistic algae," "depth limit of habitat-forming species," "species composition of habitat-forming species" and "species diversity of habitat-forming species." The reason for using "fraction of opportunistic algae" is that some filamentous algae species tolerate better wave action and ice scraping compared to perennial macroalgae. For example, in the northern Baltic Sea when *Fucus* is eliminated by ice scraping the more scraping-tolerant species, *Cladophora*, will utilize the same area (Kiirikki 1996). Westerbom et al. (2008) found that densities of mussels increased toward increasing exposure while biomasses reached peak values at intermediate sites. Therefore, wave action may have negative effects on large individuals of mussels by dislodging them (Westerbom & Jattu 2006). To indicate wave effects on mussels can the indicator "density/biomass and size of mussels" be used.

Many physical disturbance factors cause direct effects to reefs. For example, bottom trawling, dredging, gravel extraction and algal/mussel harvesting removes species and/or stones altering habitat. Dumping can also cover the hard bottoms with soft sediments changing the habitat type and therefore destroy reefs. Indicators to indicate direct effects could be "coverage of perennial macroalgae," "density/biomass of mussels," "species composition of habitat-forming species," "species diversity of habitat-forming species" and "area of reef."

Climate change (Appendix 1)

Decreased salinity is one of the major changes caused by climate change in the Baltic Sea. As many marine species in the northern Baltic Sea lives already at the edge of their salinity range they are supposed to disappear from the northern parts of the Baltic Sea and move towards southern parts (HELCOM 2009a). This will dramatically change the species composition of reefs because salinity changes will be focused on keyspecies as *Fucus vesiculosus*, *Furcellaria lumbricalis* and *Mytilus edulis* and other mussels. Moreover, species (e.g. *Laminaria saccharina, Fucus serratus, Delessaria sanguinea*) living in the south of Baltic Sea that require high salinity are probably going to disappear in the future from the Baltic Sea. Species that are clearly limited by salinity may become important indicators for assessing of climate change.

Decreased salinity affects the growth rate (e.g. smaller size) and reproduction of macroalgae and mussels, which could be seen in changes of distribution and diversity of species. When salinity is low, asexual reproduction of macroalgae increases leading to loss of genetic diversity, making populations very sensitive to environmental pressures. The indicators to indicate changes in salinity could be "coverage of perennial macroalgae," "fraction of opportunistic algae," "number of late successional species," "species and genetic diversity of habitat-forming species," "density/biomass and size of mussels" and "species composition of habitat-forming species." Many green opportunistic algae cope well in low salinities and their amount have been found to be highest in the most brackish area (Nielsen et al. 1995). Therefore "fraction of opportunistic algae" would be one indicator to indicate decreased salinity. Moreover, measurements of growth and size of species could indicate the changes in salinity range.

A long-term increase in CO_2 results in acidification of the ocean water. Acidification of seawater leads first to decrease of calcification, and in the lower pH regime, to dissolution of calcified structures affecting plankton groups, bivalves and snails. In the Baltic Sea where calcification is already lower owing to low salinity, the effects may be large. Increases in CO_2-concentrations can have also effects on macrophytes and their distribution can change (Short & Neckles 1999). "Coverage of perennial macroalgae" and "density/biomass and size of mussels" could indicate these changes.

As the precipitation and the river run-off are assumed to increase the sea level will arise. Increased storms and stronger wind increases the wave length and exposure making wave action and ice scouring to be stronger and therefore also coastal erosion more intense. Increased exposure effects negatively also on water clarity. When sea level rises also water depth will increase causing reduction in available light to the bottom and having therefore influences on distribution, coverage and depth limits of habitat-forming species (Short & Neckles 1999). However, in the northern Baltic Sea continuous land-uplift may diminish the effects. Besides causing negative effects by drag, removal and mechanical damage, increased exposure can have also positive effects on perennial macroalgae and mussels with hindering the sedimentation. Indicators to indicate increased exposure could be "coverage of perennial macroalgae," "fraction of opportunistic algae," "number of late successional species," "depth limit of perennial macroalgae," "density/biomass and size of mussels" and "species composition of habitat-forming species."

Rising water temperature can stimulate typical warm-water species such as cyanobacteria causing more algal blooms which are harmful to macroalgae and benthic fauna. Along rising water temperature the abundance of cold-water species will decrease. There has been documented that especially benthic fauna suffer from high water temperatures (HELCOM 2009a). Increased abundance of cold-water species can be indicated with the indicators as "species diversity of habitat-forming species" and "species composition of habitat-forming species." Shorter period of ice cover will decrease the ice scraping effect and make, together with warmer sea temperature, the growing season to be longer. Longer growing season can have effect that increases growing rates of fast-growing opportunistic algae in spring giving them a competitive advantage at the expense of slow-growing perennial macroalgae. To indicate the longer growing season could therefore be done with the indicators as "coverage of perennial macroalgae" and "fraction of opportunistic algae."

Warmer sea temperature with increasing marine traffic enables the introduction of non-native species. Depending on the status of the alien species it has different effects on the habitat. According to Olenin & Leppäkoski (1999) alien species can have the status as present, common or dominant. The category "present" is used for species, which never dominate numerically in native benthic communities. "Common" species occur frequently and abundantly in the whole area, but do not exceed the abundance of the native species. The category "dominant" includes species which are numerically dominant (>40%) either by biomass or by density. Non-native species may have negative effects on keyspecies of reef. Moreover, warmer sea temperature enables arrival of new diseases to the Baltic Sea. Diseases may have large effects on macrophytes as has been observed in the past (Frederiksen et al. 2004).

One potential non-native mussel in decreasing salinity for reefs would be fresh-water species *Dreissena polymorpha* which already inhabit in the eastern Gulf of Finland (Orlova et al. 2006) and in some lagoons (Olenin & Leppäkoski 1999, see lagoons). If the status of non-native species is known, indicators for identifying their effects could be "density/biomass of (native) mussels" and "coverage of perennial macroalgae." Other indicators could be "species diversity of habitat-forming species" and "species composition of habitat-forming species."

Increased riverine sediment loads, caused by increased river runoff, will change the hard bottoms of reefs to the soft bottoms causing remarkable changes. Indicators to indicate increased sedimentation could be “coverage of perennial macroalgae,” “depth limit of perennial macroalgae,” density/biomass of mussels” and “area of reef.” Moreover, sediment loads from the rivers have effects on water turbidity which can be indicated with the indicator “depth limit of perennial macroalgae.”

Suggested indicators

Examples for suggested perennial macroalgal species could be *Fucus* spp., *Furcellaria lumbricalis* and *Laminaria* spp., as they are common species on reefs. There are also many other perennial macroalgal species which could be used as indicators. *Mytilus* spp. could be one good indicator species of mussels. *Mytilus* is very common on reefs and is suggested to be an indicator, especially on the deep water where the vegetation is scarce. Both empirical and modeled coverage data on *Mytilus* are available, and therefore *Mytilus* should be included as a potential indicator species of reefs. Other mussels capable to form biogenic reefs in the Baltic Sea are *Modiolus modiolus* and fresh-water species *Dreissena polymorpha*. However, their distribution is quite small and the knowledge of their ecology is not sufficient.

In the northern Baltic where salinity is low the indicator species and metrics are different. There, for example, eutrophication might be only seen in the change of biomass and lower limits of the algal distribution. Because in the northern Baltic perennial species maintaining their biomass during the whole year are hardly found, the annual macroalgae are of major importance as an indicator species (Kautsky 1991).

The indicators are suggested having in mind the goal to assess the favourable conservation status, by suggesting species and indicators which are already in use (e.g. according to Baltic GIG) and commonly studied with long-term data. Many indicators are also suggested under the European Water Framework Directive.

Algal cover/number of late successional species

Water clarity (light penetration), nutrient concentrations and salinity are three of primary growth-regulating factors that have been documented to influence large-scale patterns of distribution and abundance of macroalgae (Nielsen et al. 2002a, 2002b). Macroalgal cover at specific water depths (at deeper depths) is also likely to reflect changes in water quality (Krause-Jensen et al. 2007a). In many studies have been found that both Secchi-depth and nutrient load have a close correlation with total vegetation cover (e.g. Dahl & Carstensen 2005, Krause-Jensen et al. 2007b, Nielsen et al. 2002a).

'Total and cumulative algal cover" has been found to be good indicators of the ecological quality of reefs in open waters in Kattegat (Dahl & Carstensen 2008). However, studies showed that total vegetation cover is only useful as indicator at water depths where the algal cover is less than 100% because physical disturbance affects more in shallow waters. Cumulative cover is less dependent on water depth (Dahl & Carstensen 2008). In MOPODECO the indicators "total and cumulative algal cover" have been developed and intercalibrated to other countries. Carstensen et al. (2008) found "total algal cover" and "number of perennial algal species" together with "fraction of opportunistic algae" in areas of low salinity to be the most promising among the potential algal indicators at least on Danish coastal waters. Many countries have also studied the "coverage of individual species of perennial macroalgae" (Baltic GIG).

Depth limit

The lower depth limit of the macroalgae is controlled by the intensity of light (e.g. Bäck & Ruuskanen 2000). Other factors than light can also regulate the depth limit of macrophytes which are sediment characteristics and oxygen depletion (Solimini et al. 2006). "Depth limit of perennial macroalgae species" is used in many countries along the Baltic Sea and there are data from several species. Depth limit of macroalgae is a common indicator because of its positive relationship with water transparency. Instead of using just one keyspecies, depth limit of algal community and larger algal groups (e.g. brown algae) would also be good indicators (Solimini et al. 2006).

The lower depth limit on hard substratum may be difficult to determine precisely and may be limited by lack of substratum. Therefore, for example the coverage of macroalgae at a specific depth, % coverage depth limit or depth of maximum coverage would be more useful variables than absolute algal depth limit (Krause-Jensen et al. 2007b).

Fraction of opportunistic algae

"The fraction of opportunistic algae" here means relative cover of opportunistic algae compared to perennial macroalgae cover. "Fraction of opportunistic algae" could be a good indicator together with other indicators (Carstensen et al. 2008). However, according to e.g. Krause-Jensen et al. (2007b) "fraction of opportunists" should be estimated in shallow waters where light is not a limiting factor, where physical exposure does not export these algae, and where no strong salinity gradient exist.

Species diversity and composition of macroalgae

"Species diversity and composition of macroalgae" reflect changes in water quality and may be used as an indicator (Wells et al. 2006, Krause-Jensen et al. 2007a). Species composition is an important contributor to the structure of a habitat and therefore the reef as a whole. A measure of species diversity also gives an indication of the quality of a biotope, where any change in diversity may indicate a cyclic change or trend in sediment communities (Davies et al. 2001). However, there is also a natural turnover of emphemeral algae resulting in variable species composition between months, seasons and over several years. Therefore, it would be more useful study changes in general community composition using as a "indicator group" functional groups (e.g. late successional/perennial species and opportunistic/annual species) (Wells et al. 2007).

Density/biomass and size of mussels

Mussels as *Mytilus edulis* is one of the main species on reefs. When occurring as large mussel beds it forms a biogenic reef serving habitat to other species. *Mytilus* reacts positively to eutrophication owing to increased phytoplankton abundance. At the same time, sedimentation affects mussel recruitment by preventing mussel colonization to bare bottoms and post-recruitment survival, and by preventing establishment of algal stands facilitating mussel colonization (Westerbom et al. 2008). Also sedimentation cause siltation and clogging of filter apparatus of mussels.

The size of *Mytilus* could be one parameter when assessing the effects of increased wave action as observations have been made that wave action has negative effects on large individuals by dislodging them (Westerbom et al. 2008, Westerbom & Jattu 2006).

Accordingly, *Mytilus edulis*, except for the Kattegat a promising indicator when assessing the conservation status of reefs.

Area of reef

Indicator "area of reef" indicates the extent of reef. The extent of reef is unlikely to change significantly over time except by the human activity. To measure area of reef, it should be first determined how we define reefs; how much it includes gravel, rocks and sands etc. Moreover, it is important to define, when getting data in the field, if the rocks are really available for macroalgae.

The definition of the reef can be seen in the part of "Harmonisation of the definitions of Annex I habitats." Besides of the physical area, area of reef can be understand also as throughout vegetation which means that when the coverage or density of habitat-forming species decrease the extent of reef also decreases.

Table 14. Recommended indicators (more detailed) for assessing the favourable conservation status of reefs. With the bold text are the most important indicators

	REEFS (1170)
Recommended indicators	Total / cumulative algal cover Coverage of perennial macroalgae Fraction of opportunistic algae Number of late successional species Depth limit of perennial macroalgae Species composition of habitat-forming species Density/biomass and size of mussels Species diversity of habitat-forming species Genetic diversity of habitat-forming species Area of reef

5.1.4 Sandbanks

Eutrophication (Appendix 1)

The structure of the conceptual model of the eutrophication of sandbanks is similar to the conceptual model of eutrophication of reefs.

Benthic fauna benefit from the increased phytoplankton abundance and their abundance increases altering habitat. Especially deposit feeders are gained by organic enrichment (Boström et al. 2001). On the sandbanks this effect could be seen with the indicator as “density/biomass of benthic fauna.” The abundance of grazers increases along the increased amount of opportunistic algae and therefore, also grazing of angiosperms increases altering habitat. Grazing decreases “coverage of habitat-forming species” and this can be used as an indicator.

Sedimentation has different effects on sandbanks than on reefs. On sandbanks sedimentation affects more by hindering anchoring of rooted plants. This cause habitat fragmentation and habitat loss. The indicators which could be used to indicate the effects of sedimentation are “coverage of habitat-forming species” and “species diversity of habitat-forming species.”

Lack of light has many effects on sea-grass and angiosperm communities. Reduction of light availability leads to reduced depth penetration and abundance of sea-grasses (Duarte 1991). The most important effect of light attenuation is decreased “depth limit” of sea-grasses and angiosperms sensitive to poor light conditions, and this could be used as an indictors. Other indicators to indicate the light attenuation on sandbanks could be “coverage of habitat-forming species,” “fraction of opportunistic algae,” and “species composition of habitat-forming species.” Light reduction has influences in all habitat change levels.

Filamentous algae live as an epiphyte on the surface of the rooted plants and shade them. Moreover, filamentous algae can occur as freefloating mats and when covering the seabottom they cause oxygen depletion. On the sandbanks drifting algal mats are serious problem for eelgrass-communities and benthic fauna. The most bottom-water oxygen is consumed by the decomposition of organic material (mainly due to bacteria), resulting in hypoxia and anoxia and initiating the release of toxic hydrogen sulphide from the sediments (HELCOM 2009a). At stages of hypoxia and anoxia, macrozoobenthos is eliminated and important ecosystem services are lost. Filamentous algae do not compete with rooted plants for space because they have different substratum type requirements.

Oxygen depletion can cause habitat alteration, habitat fragmentation and habitat loss depending on its intensity. Indicators for indicating to influences of oxygen depletion could be "coverage of habitat-forming species," "density/biomass of benthic fauna," "species diversity of benthic fauna" and "species composition of benthic fauna."

Physical disturbance (Appendix 1)

Habitat effects and links to indicators are very similar to the conceptual model of physical disturbance of reefs and are congruent with the conceptual model of eutrophication of sandbanks. Habitat effects as sedimentation and turbidity are already explained in context of conceptual model of eutrophication of sandbanks.

Dumping of dredge spoil smothers benthic fauna and flora by burying them. Especially, dumping has effects on benthic fauna which live on the sediment. Mills & Fonseca (2003) found that burial with sediment significantly increased mortality and decreased productivity of *Zostera marina*. The tolerance of benthic fauna to burial depends on their escape capacity, by burrowing and/or by extending their siphons to the new sediment surface (Powilleit et al. 2009), and tolerance to oxygen depletion. Indicators to indicate the effects of burial could be "coverage of habitat-forming species," "species diversity of habitat-forming species and benthic fauna," "species composition of habitat-forming species" and "density/biomass and species composition of benthic fauna."

On sandbanks wave action and ice scouring are also important factors which may have major influence on the vascular plants inhabiting sheltered and shallow areas. One of the keyspecies of sandbanks, *Zostera marina*, occurs mainly in exposed sandy bottoms and exposure has been found to constrain the abundance of it in shallow water (Duarte 1991). Wave action and ice scouring control the upper depth limit of angiosperms (Duarte 1991). When increasing turbidity, wave action may also have influence on lower depth limit by decreasing the light intensity. Moreover, waves may tear off leaves and uproot entire plants. Indicators to indicate increased wave action are "coverage of habitat-forming species," "depth limit of habitat-forming species," "species diversity of habitat-forming species and benthic fauna" and "species composition of habitat-forming species.

Sand extractions are one of the major threats to sandbanks and have large effects on species, substratum and whole habitat when removing parts of it away. Indicators to indicate direct effects on sandbanks are "coverage of habitat-forming species," "species diversity of habitat-forming species and benthic fauna," "species composition of habitat-forming species," "density/biomass and species composition of benthic fauna" and "area of sandbank."

Climate change (Appendix 1)

Habitat effects and links to indicators are very similar to the conceptual model of climate change of reefs and are congruent with the conceptual model of previous conceptual models of sandbanks. Other habitat effects not described here are explained in previous models.

One of the keyspecies of sandbanks, *Zostera marina*, is living in the Baltic Sea, especially in the northern parts, on the edge of its salinity range and when salinity decreases its distribution changes. Sexual reproduction of *Zostera marina* is already rare, at least in the Archipelago of SW Finland, therefore the gene flow is low and large meadows can be formed from one clone. Thereby re-colonization after an extinction event is very small (Boström 2006). With decreasing salinity sexual production will decrease even more and this may have dramatically effects on *Zostera marina*. Decreased salinity have major influence in "coverage of habitat-forming species," "fraction of opportunistic algae," "species diversity of habitat-forming species and benthic fauna," "genetic diversity of habitat-forming species," "species composition of habitat-forming species" which can be used as indicators.

Heavy storms and waves reduce seagrass cover and increases fragmentation of seagrass bed (Fonseca & Bell 1998). Direct impacts of storm activity on seagrass beds include erosion by wave action, shading and smothering by suspended material. Moreover, heavy waves can remove sandbanks altering and fragmentizing habitat. Indicators to indicate increased exposure could be "coverage of habitat-forming species," "depth limit of habitat-forming species," "area of sandbank" and "species composition of habitat-forming species."

Suggested indicators

For sandbanks suggested indicator species of habitat-forming species are *Zostera marina* and other angiosperms that live in *Zostera*-communities as *Potemogeton* spp., *Ruppia* spp. and *Zannichellia* spp. Moreover, when occurring, charophytes can be indicator species as well. *Zostera marina* and other angiosperms are common and typical species on sandbanks along the Baltic Sea. Benthic fauna on sandbanks are not primarily habitat-forming species but they are important species in soft-bottom communities responding to environmental pressures with large fluctuations in diversity, abundance and biomass and, therefore could be regarded as an indicator species in sandbanks. Moreover, when a long-term change in the vegetation is not observed, in many cases marked changes in the associated fauna have been recorded, in general changes in their abundance (Boström et al. 2001). In areas where *Zostera marina*-communities do not occur (for example in the Botnia Sea and Bothnia Bay) other angiosperm species as for example *Potamogeton* spp. and *Myriophyllum* sp., and charophytes can be good indicators in addition to filamentous algae.

The indicators are suggested having in mind the goal, the assessment of favourable conservation status, by suggesting species and indicators which are already in use (Baltic GIG) and commonly studied with long-term data.

Coverage of angiosperms

In many studies has been observed that the abundance of sea-grasses declines with increasing TN load (Short and Burdick 1996; Hauxwell et al. 2003) and probability of sea-grass cover decreases with declining Secchi depth (Krause-Jensen et al. 2003). Abundance of angiosperms as coverage (or biomass, leaf length, leaf width and shoot density) from intermediate water depths towards deeper water tends to decline in parallel to reductions in light availability with depth and increases with increasing transparency (e.g. Duarte 1991). Light penetration has been regarded one of the most important factor affecting distribution of eelgrass (Duarte 1991). Coverage of angiosperms has been used as an indicator for changes in water quality in many countries (Baltic GIG). Moreover, i.a. Selig et al. (2007a) suggested depth distribution limit of macrophytes to be as a valuable indicator of anthropogenic impact.

Depth limit

Z. marina has been regarded as a useful indicator of water quality because water clarity regulates its extension towards deeper waters (Krause-Jensen et al. 2005). The indicator is useful because it is affected by nutrient concentration and water transparency (Nielsen et al. 2002a, 2002b). Boström et al. (2003) described a strong correlation between Secchi depth and the lower depth limit of *Z. marina* in long-term studies of the south coast of Finland. Moreover, Nielsen et al. (2002b) found for the Danish coastal waters a linear relationship between the depth limit of *Z. marina* and the transparency in the coastal waters. Therefore, the depth limit of eelgrass is likely to be a useful indicator also under the European Water Framework Directive (Schories et al. 2009). Instead of using just one species it would be better to measure the whole plant community e.g. depth limit of *Zostera* community or depth limit of other angiosperm-communities.

In general, the lower depth limit of eelgrass is not a useful indicator in very shallow areas where factors other than light play the major regulating role, and the assessment of ecological status of such areas should instead be based on other indicators such as the abundance or species composition of shallow-water flora or fauna (Krause-Jensen et al. 2005). Therefore other indicators should be used too. The indicator "species composition of habitat-forming species" is useful when habitat is changed much and species sensitive to poor light conditions are replaced by species tolerating bad light conditions such as *Potamogeton* spp. and *Myriophyllum* spp.

Fraction of opportunistic algae

Here fraction of opportunistic algae can be measured as relative cover or biomass of opportunistic algae in Zostera beds or other plants. Fast-growing macroalgae gain a competitive advantage over sea-grasses as eutrophication increases (NCM 1998). Eutrophication may cause a shift from sea-grass to communities dominated by fast-growing macroalgae (Isaksson & Pihl, 1992; Valiela *et al.*, 1997). For example, along the Swedish west coast, increased dominance of filamentous algae and heavy overgrowth of epiphytes on *Z. marina* have caused decline of associated faunal communities, e.g. shrimps and crabs (Isaksson & Pihl, 1992). It is clear that increasing amount of filamentous algae is a major threat to the brackish Baltic sea-grass ecosystems.

Species diversity and composition of angiosperms

Species composition is an important contributor to the structure of some biotopes. A measure of species diversity also gives an indication of the quality of a biotope, where any change in diversity may indicate a cyclic change or trend in sediment communities (Davies et al. 2001).

Species composition of angiosperms has been found to change within decreasing water quality and therefore it could be a good indicator (see more information in the chapter of lagoons).

Density/biomass, species diversity and composition of benthic fauna
Zoobenthic indices are commonly used in an evaluating the quality of the sea bottoms. The degree of pollution of the water is not necessarily the same as that of bottom, and the sediment-water interface, where relative long-lived sessile or sedentary organism can be good indicators (Perus et al. 2007). Signs from eutrophication in benthic communities are changes in abundance, biomass and species composition (Perus et al. 2007). Benthic invertebrate status in the central parts of the Baltic Sea, in particular, is more or less entirely controlled by the presence or absence of hypoxia/anoxia (HELCOM 2009a).

The number of species has not significantly changed over time (Bonsdorff et al. 1997) but species composition has changed dramatically with a shift from suspension feeders to deposit feeders, indicating functional disturbances (Bonsdorff et al. 1993). This functional shift, connected to eutrophication, has also been shown in other studies analyzing long-term changes in macrozoobenthos communities along the Finnish coast (e.g. Perus & Bonsdorff 2004). Change in species composition can also been seen in a shift of relative abundance of tolerant and sensitive species. Many different indices have been developed to indicate the changes in zoobenthos. For example, Brackish Water Benthic Index (BBI) developed for Finnish coastal waters consider relative abundance of sensitive and tolerant species, zoobenthic abundance, species richness and Shannon-Wiener index (diversity index). In the conceptual models of sandbanks have been suggested zoobenthic indicators as "species diversity" and "density/biomass" and "species composition" of benthic fauna. Relative abundance of sensitive and tolerant species (e.g. Pearson & Rosenberg 1978) could be included in measures as an indicator. By classifying species on a scale from sensitive to tolerant in relation to mainly organic enrichment, and multiplying them according to their relative importance at a sampling site, we get a good estimate of the benthic community structure and its function (Perus et al. 2007). Indicators can be selected depending on the index which is going to be used.

Area of sandbank
Indicator "area of sandbank" indicates the extent of sandbank. Sandbanks are mobile habitats and change for example by waves and storms. The definition of the sandbank can be seen in the part of "Harmonisation of the definitions of Annex I habitats."

Table 15. Recommended indicators (more detailed) for assessing the favourable conservation status of sandbanks. With the bold text are the most important indicators

	SANDBANKS (1110)
Recommended Indicators	Coverage of angiosperms (+charophytes) Depth limit of angiosperms (+charophytes) Species diversity of angiosperms Fraction of opportunistic algae Species composition of habitat-forming species Species composition of benthic fauna Density/biomass of benthic fauna Species diversity of benthic fauna Genetic diversity of habitat-forming species and benthic fauna Area of sandbank

5.1.5 Lagoons

Eutrophication (Appendix 1)

The conceptual model of eutrophication of lagoons has the same structure as previous models of eutrophication. However, in the lagoons both hard and soft bottoms can occur. Therefore, the conceptual models have elements from both reefs and sandbanks. Hard-bottom species include here mainly macroalgae and mussels. Hard-bottom mussels like *Mytilus* are included in indicator "density/biomass of benthic fauna" as well as soft-bottom fauna. Links between habitat effects and indicators, with some differences, are already described in those previous models of eutrophication.

Physical disturbance (Appendix 1)

Habitat effects and links to indicators are very similar to the conceptual model of physical disturbance of reefs and sandbanks and are congruent with the conceptual model of eutrophication of lagoons. Here are described briefly effects of one of the main pressure (boating) of lagoons. Other pressures are described partly already in previous models.

Dredging, increased turbidity and wave action caused by boating have large effects on vascular plants which occur in sheltered areas, especially in lagoons. Dredging counteracts the natural land-uplift process of the northern Baltic Sea and affects therefore negatively species that dominate in late succession stages in flads (Eriksson et al. 2004). Water turbidity in marinas can be high affecting declines in species richness and depth limits (Eriksson et al. 2004).

Climate change (Appendix 1)

Habitat effects and links to indicators are very similar to the conceptual model of climate change of reefs and sandbanks and are congruent with the conceptual model of previous conceptual models of lagoons. Here are described more detailed known effects of non-native species in lagoons. Other habitat effects not described here are explained in previous models.

Benthic non-native species may have large effects on lagoons. In the Curonian Lagoon alien species *Dreissena polymorpha* is a dominant species. *Dreissena* has altered the habitat and its substratum type because shell deposit of *Dreissena* have changed former soft bottoms into shell gravel and have created patches of hard substratum for sessile species. However, there is not any evidence that *Dreissena* would have replaced other mussels as *Mytilus*. (Olenin & Leppäkoski 1999).

Non-native species *Marenzelleria viridis* is a dominant species in the Vistula Lagoon. *Marenzellaria* has caused remarkable changes in the structure of the soft bottom community by burrowing deeper than most native species and increasing the thickness of the populated surface sediment layer and depth limit of bioturbulation having positive effects on other fauna (Olenin & Leppäkoski 1999). However, when burrowing into the sediment *Marenzellaria* can release nutrients from the sediment increasesing eutrophication (Viitasalo-Frösen et al. 2009). Moreover, *Marenzelleria* may have negative effects on native polychaete as *Nereis diversicolor* which abundance has been declined (Boström et al. 2001).

Suggested indicators

Lagoons are usually shallow complicated habitats because the sea bottom can be sandy, muddy, silty, rocky etc. Moreover, lagoons, especially flads and gloes in the northern Baltic Sea, are dynamic habitats which change in time with land-uplift succession and therefore also the vegetation changes. There are characterizing species in every succession stages (Musterhjelm 1997) which has to consider when using indicators.

As the salinity gradient can change a lot in lagoons being highest near the sea opening, in lagoons occur both fresh-water and marine species. In lagoons occur macroalgae, angiosperms, soft-bottom and hard-bottom fauna. The occurrence of different species depends also on the depth of the lagoon.

Suggested habitat-forming species as indicators for lagoons are charophytes and angiosperms such as *Zostera, Potamogeton* spp. and *Ruppia* spp. Perennial macroalgae, if occurring, can be also habitat-forming indicator species. Moreover, as a habitat-forming fauna *Mytilus* is suggested. However, as lagoons can inhabit also fresh-water species other mussels than *Mytilus* can play an important role in the habitat and can therefore be an indicator species. Moreover, soft-bottom benthic fauna could be used as an indicator.

Coverage of habitat-forming species and depth limit

Charophytes are characteristic species in sheltered coastal lagoons and some endangered species are very depended for expamle on Finnish flads (Raunio et al. 2008). Light is one of the most important environmental factors controlling the development of charophyte populations (Schubert & Blindow 2003). Moreover it has been observerd that the nutrient-enrichment of the waters poses the decline or loss of charophytes (Selig et al. 2007b). Charophytes are very sensitive to poor light conditions because they form mats near the sediment (Schubert & Blindow 2003). When water quality is bad charophytes are expecting to disappear before vascular plats which form mats near the surface water. Therefore, decreased coverage of charophytes could be seen in very early stage of habitat change. *Ruppia maritima* is also very sensitive to poor light conditions and could be good indicator in lagoons.

In recent decades, the number of species, distribution area and biomass of charophytes has declined significantly in the Baltic Sea (Schubert &Blindow 2003). Most of the records on the considerable decline of charophyte populations are from coastal waters of Schleswig-Holstein, the Swedish west coast and the coastal waters of Hanko peninsula in southwestern Finland (Shubert & Blindow 2003). The decline in charophytes correlates in many cases with increased nutrient loading, phytoplankton production, and decreased water transparency (Schubert & Blindow 2003). Due the response to environmental disturbances, especially species *Chara tomentosa* can considered as an important indicator species in soft-bottom environments. However, this species declines also because of natural environmental changes, land uplift and isolation, which has to be considered (Munsterhjelm 2005).

Selig et al. (2007a) have proposed the lower depth limit of vegetation and loss of charophyte-dominated plant communities to be indicator in the inner coastal waters of the southern Baltic Sea. However, in the shallow lagoons, as in archipelago flads in Finland, light does not determine the lower depth limits of species distribution because of the shallowness of flads and therefore depth limit is not valid indicator in finnish flads (Munsterhjelm 1997).

About the coverage and depth limit of angiosperms see chapter of sandbanks.

Fraction of opportunistic algae

Here fraction of opportunistic algae can include relative cover of opportunistic algae compared to perennial macroalgae and relative coverage or biomass of opportunistic algae compared to plants.

See more details in the chapters of reefs and sandbanks.

Species diversity and composition of angiosperms

Changes of plant communities have been found to be a more robust parameter characterizing the quality of waters (Selig et al. 2007a).

In the poor light conditions sensitive species as charophytes dominated communities disappear first. After the loss of *Zostera* and other angiosperms the stands of *Myriophyllum* and *Potamogeton* dominate the vegetation (Selig et al. 2007a, Eriksson et al. 2004). Occurrence of *Potamogeton* and *Myriophyllum* communities indicate eutrophication as they are last to survive when nutrient input is high (Selig et al. 2007b) and could be used as indicators in lagoons.

The vegetation in inlets used as marinas and in inlets adjacent to ferryboat routes declines in species richness and percentage cover with depth. Moreover, change in species composition resulting in increased water movement can be seen in marinas and along ferryboat routes with decreasing abundance of sensitive species. *Chara tomentosa* and *Najas marina* are very sensitive to exposure and they do not thrive in inlets adjacent to ferryboat routes. Instead, *Potamogeton perfoliatus*, which requires circulating water, thrives well adjacent to ferry-boat traffic. (Eriksson et al. 2004).

Submerged vascular plants described here are used as indicators in sandbanks as well.

Density/biomass, species diversity and composition of benthic fauna

About the indicators of benthic fauna see pages chapters of reefs and sandbanks.

Area of reed

One indicator which is not described in the conceptual models could be "area of reed" as an example species *Phragmites australis*. *Phragmites australis* is a very typical species on the shores of the lagoons and because it benefit from the eutrophication its increased area could be one indicator to estimate the condition of lagoons. However, reeds are natural species in lagoons and cutting them too intensively may have large influences on submerged vegetation which require sheltered areas (Munsterhjelm 2005). Cutting of reed has negative effects on e.g. *Chara tomentosa* and other Charas, *Najas marina* (Munsterhjelm 1997). However, some other species also benefit from the cutting of reed and grazing by cattle (Munsterhjelm 1997).

Area of lagoon

Area of lagoon is not suggested in the conceptual model as an indicator but it could still be one indicator. Lagoons are dynamic habitats and its formation can change naturally in the long run. Extent of lagoon can also change by human activity depending on the utilization rate of lagoons. Lagoons are typical areas for recreational boating. Boating, marinas and docks diminish the extent of lagoon. To measure and assess the effects of human activity to extent of lagoon basin is not easy.

Davies et al. (2001) have suggested attributes of physical properties to define favourable condition of lagoons in UK. These attributes are average water depth within the lagoon basin, measurements of isolating barrier and seasonal average salinity.

The definition of the lagoon can be seen in the part of "Harmonisation of the definitions of Annex I habitats."

Table 16. Recommended indicators (more detailed) for assessing the favourable conservation status of lagoons. With the bold text are the most important indicators

	LAGOONS (1150)
Recommended Indicators	Coverage of angiosperms and charophytes (+perennial macrolagae) Depth limit of angiosperms and charophytes (+perennial macroalgae) Loss of charophyte-communities Fraction of opportunistic algae / opportunists in plant-communities Species composition of vegetation Species diversity of habitat-forming species Area of reed Density/biomass of benthic fauna Species diversity of benthic fauna Species composition of benthic fauna Genetic diversity of habitat-forming species and benthic fauna Area of lagoon basin

5.1.6 Conclusions

In the report it has been shown that it is far from easy to find exact indicators for different habitat and pressures. Habitats are complex systems, and many effects of pressures can be indicated with similar indicators. Therefore, by using only single habitat-forming species to assess favourable conservation status of the habitats is insufficient. Instead, a combination of different indicators is required to document the environmental status of the habitats.

5.2 Assessment tools

5.2.1 Assessment of habitat quality on reefs in NATURA 2000 areas in the Kattegat

Assessments of reef quality can be made using the same set of models (Dahl & Carstensen 2008) that were used for mapping vegetation cover of reef habitats (Chapter 4.2). As no "historic" data exists that can be used as a reference and target level for development of algal vegetation we have to use the same models to predict a target given an input of nutrient load assumed as reasonable values for the boundary between favourable and unfavourable conservation status.

5.2.2 Model and observations of vegetation cover

We have chosen a scenario as a reference nitrogen load of 10,000 tons for the period from January–June, more or less in accordance with the level suggested as a reference level in the Water Framework Directive for inner Danish waters (Carstensen et al. 2008). The target level representing the boundary between favourable and unfavourable status is based on a scenario of total nitrogen load on 23,000 tons (January–June), more or less in accordance with the boundary level God–Moderate suggested by Carstensen et al. (2008) for the Water Framework Directive. The load of 23,000 tons total nitrogen reflects the order of load from Danish and Swedish point sources and rivers to Kattegat in the years 1996 and 1997, two years characterized by very low precipitation in the first half of the calendar year. We have also used an average solar radiation level for the period 1993–2005 and a sea-urchin grassing level equal to a sea-urchin cover of 0,1% on hard substrate.

Actual observations on total cover of vegetation of the Danish reefs has been collected at as part of the Danish National Marine Monitoring and Assessment Program (Bijl et al. 2007) and data from Lilla Middelgrund was collected as part of the Balance project and reported in Dahl et al. 2007.

Deviations between actual observations of total vegetation cover from 2008, 2009 and 2010 and the modelled suggested boundary between favourable and unfavourable conservation status have been calculated for 3 Danish reefs in Kattegat and for Lilla Middelgrund in the Swedish part of Kattegat in 2006.

The model describing total cover was found to be the most robust (Dahl & Carstensen 2008) and was chosen as a first priority on deeper reefs where total cover was below 90%. Cumulative cover was used on the shallow part of Lilla Middelgrund in combination with total cover on the deeper part.

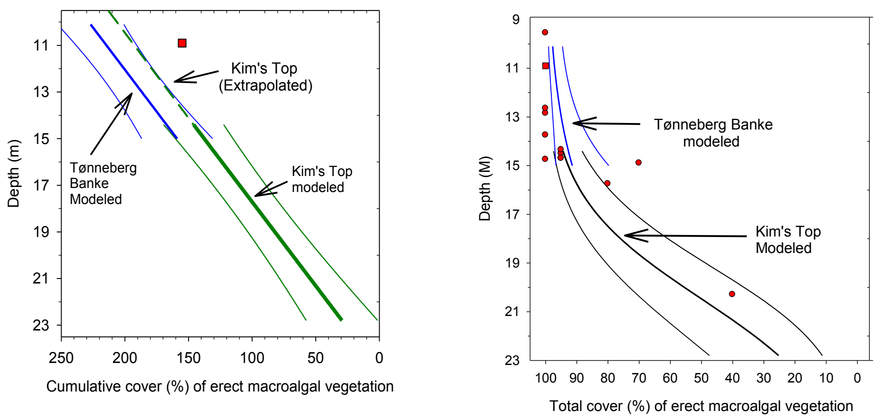
The predicted distribution of cumulative cover on Lilla Middelgrund is shown in Figure 46 together with the single observations in 2006. The observed cumulative cover of 150% were a bit smaller that the predicted on approximately 190%.

The necessary logistic transformation of total cover data makes the prediction unreliable close to 100% and 0%. For this reason observations in the interval between 90–100% and 0–10% have been discarded as unsuited for quality assessment. Three total cover observations from 2006 on the deeper part of Lilla Middelgrund were all below the predicted value and another 4 observations with nearly full cover in the depth range from 14.4–14.7 m were a bit better than the target (Figure 49).

The three reefs in the Danish N-2000 sites show year to year changes (Figure 50). In 2008 only one observation at the deep part of Herthas Flak were better than the suggested target and at lot of the other observations show profound deviation for the modeled target values.

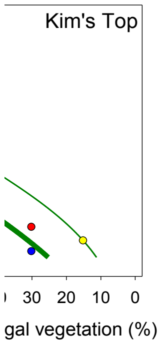
The depth distribution in 2010 was in general better than the other years with several observations of total cover that was better than the predicted target values. But also in this year some observations were below the target value.

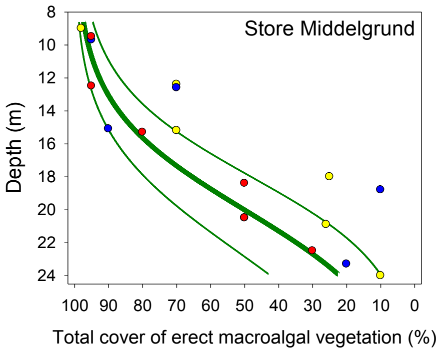
Figure 49



Left: Modelled cumulative cover at different depth at Kim Top and Tønneberg Banke with a total nitrogen load on 23,000 tons to Kattegat from January to June (solid thick lines) with 95% confidence levels (solid thin lines). The green stippled line represents extrapolated cumulative cover at Kim's top to more shallow waters. The red square represents observed cumulative cover at Lille Middelgrund in 2006.
Right: Modelled Total cover at different depth at Kim Top and Tønneberg Banke with a total nitrogen load on 23,000 tons to Kattegat from January to June (solid thick lines) with 95% confidence levels (solid thin lines). The red square represent diver observed total cover at Lille Middelgrund and the red dots represent total cover observed by drop video.

Figure 50





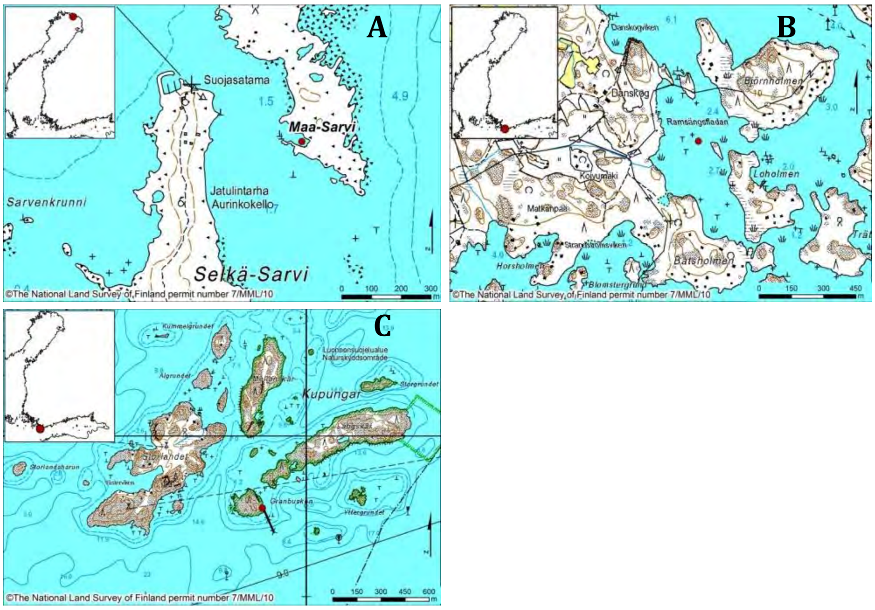
Modelled expected total vegetation cover at different depth representing a suggested boundary between favourable and unfavourable conservation status (solid line) and 95% confidence intervals (thin lines) and observations in 2008 (yellow dots), 2009 (blue dots) and 2010 (red dots) at three reef locations in three different Nature 2000 sites in Kattegat.

5.2.3 Assessment of habitat quality on reefs and in lagoons in NATURA 2000 areas in Finland

Introduction

Three case studies for assessment of habitat quality of NATURA 2000 areas were undertaken in Finland; two in the coastal lagoons of Maa-Sarvi flad in the Bothnian Bay and Danskogfladan flad in Hanko Peninsula on the south coast of Finland, and one in the Granbusken Reef in the Hanko Peninsula (Figure 51).

Figure 51. Location of the three case studies in Finland; Maa-Sarvi flad (A), Danskogfladan flad (B) and Granbusken Reef (C)



5.2.4 MAA-SARVI FLAD

Physiographic limits

Flads are a sub-type of coastal lagoons (European Commission 2007). The Maa-Sarvi flad is located at the Perämeri National Park in the Bothnian Bay. The flad lies at the island of Maa-Sarvi quite far off at the open sea near the border of the Finnish and Swedish territorial waters, in the outer archipelago of the towns of Kemi and Tornio. The distance to the nearest coast is about 15 km.

With an area of about 2500 m^2, the Maa-Sarvi flad is quite small. It has one inlet facing the narrow strait between the islands of Maa-Sarvi and Selkä-Sarvi. The island of Maa-Sarvi is mostly covered with forest (Leinikki & Oulasvirta 1995). At the time being, human activities on the island are minor. In the past the island was inhabited and used for cattle grazing, but the grazing ceased in the 1960's, after which the shores have become shrubby. In 2009, though, Metsähallitus cleared some forest there and will introduce culture landscape management by sheep grazing in 2010.

Physical characteristics

The flad is shallow, with depths of only 0.5–0.9 m. The bottom consists of soft organic sediment with scattered stones of 10–60 cm in diameter. No dredging or construction works have been done in the flad. The nutrient concentrations and salinity in the flad haven't been measured, but the water exchange with surrounding waters is good, the flad's water volume is small and it has no freshwater inflow, so the nutrient concentrations and salinity in the flad would be about the same as in the surrounding waters. However, the boat traffic immediately outside the flad could elevate the nutrient concentrations and the water turbidity in the flad. The surface water salinity in the area is about 1–2.5 ‰ in summer and 0–1.5 ‰ in winter (data from the SYKE database).

The post-glacial rebound (land upheaval) in the Bothnian Bay is 8–9 mm per year (www.itameriportaali.fi). This will gradually make the sill to the flad shallower, eventually isolating the flad from the surrounding waters. Flads in the northern Baltic Sea are constantly going through a series of natural morphological development stages, due to land upheaval accompanied by botanical succession (Munsterhjelm 1997).

Biological characteristics

The soft bottom of the flad is densely covered with vegetation. In 2006 and 2007, 15 vascular plant species and two charophytes were observed in the flad by Essi Keskinen and co-workers at Metsähallitus (Table 17). The only other macroalgal species were the filamentous green algae *Cladophora aegagrophila* and *Cladophora glomerata* that grow very sparsely attached to the rocky surfaces. Benthic diatoms were found to be very abundant on stones, covering all stone surfaces. Also spherical and mat-like cyanobacteria were observed.

Due to patchiness and the difficulty of visual coverage estimation, the percentages in Table 17 don't add up to 100%. Anyway, the soft sediment between the rocks seemed to be maximally covered with macrophyte vegetation. *Potamogeton gramineus* x *perfoliatus* hybrids are common in the area so they probably also occur in Maa-Sarvi as well, though their presence couldn't be ascertained in these studies. Of the species in Table 17, *Alisma wahlenbergii* is classified as vulnerable (Rassi *et al.* 2001) in Finland and *Ranunculus confervoides* as regionally threatened (www.ymparisto.fi).

Large, thallous brown and red algae of marine origin (like the bladder wrack, *Fucus vesiculosus*) don't occur in salinities as low as those of the Bothnian Bay. Instead, a high abundance of epiphytic and epilithic diatoms is characteristic of the Bothnian Bay benthic flora, and diatoms are probably of great significance in the nutrient cycles of the Bothnian Bay (Leinikki and Oulasvirta 1995). In the Bothnian Bay, low salinity reduces faunal diversity (HELCOM 2009b). Zoobenthos occurring in the flad include snails like *Bithynia* sp. and *Lymnea* sp., water mites (Hydracarina), and larvae of caddisflies (Trichoptera). The area has a rich avian fauna, about

60 bird species nesting in the Perämeri National Park (www.outdoors.fi). The Sarvi archipelago is a refuge for many threatened sea-birds.

Functional properties and services

The angiosperm-charophyte biotope type, with alternating macrophyte-covered soft bottom and diatom-clad stones, is generally dominant in shallow, sheltered waters of the Bothnian Bay National Park (Leinikki & Oulasvirta 1995). The Maa-Sarvi flad is a typical example of this kind of biotope. The vegetation in the flad is lush, maximal as for available substrate, not showing signs of external disturbance. The species richness and diversity of the Maa-Sarvi flad is clearly higher than at other sites in the area. The Maa-Sarvi flora and fauna are typical of the Bothnian Bay, but nowhere else in the area are so many species concentrated in one spot, as here. Therefore, the Maa-Sarvi flad can be regarded as a representative site deserving of protection. It may also serve as a refuge area and dispersal centre of characteristic Bothnian Bay biota.

Table 17. Estimated coverage (%) of vascular plants (V) and algae (Ch=Charophytes, MA=other macroscopic algae) in the Maa-Sarvi flad. The species were identified and their coverages were estimated from underwater photographs obtained by scuba diving in the flad 10 times in August of 2006 and 2007. The data was provided by Essi Keskinen, Metsähallitus (unpublished data)

Species		Coverage, %
Callitriche hermaphroditica	V	10
Potamogeton pusillus	V	10
Sagittaria sagittifolia x natans	V	10
Zannichellia major var. repens	V	10
Elatine hydropiper	V	5
Potamogeton gramineus	V	5
Potamogeton perfoliatus	V	5
Subularia aquatica	V	5
Alisma wahlenbergii (VU)	V	1
Eleocharis acicularis	V	1
Isoëtes echinospora	V	1
Myriophyllum sp.	V	1
Potamogeton filiformis	V	1
Potamogeton pectinatus	V	1
Ranunculus confervoides (LC, RT)	V	1
Chara globularis	Ch	5
Nitella flexilis vel opaca	Ch	5
Cladophora aegagrophila	MA	1
Cladophora glomerata	MA	1

At present, the recreational use of the flad is non-existent. The Maa-Sarvi island is also very sparsely used for recreational purposes. Only a few fishermen occasionally visit a couple of fishing huts on the island and the surrounding waters with small motorboats, but not the flad. Most of the recreational pressure is concentrated on the neighbouring Selkä-Sarvi island, with a marina, sauna, camping site etc. Immediately outside the Maa-Sarvi flad, between the islands of Selkä-Sarvi and Maa-Sarvi, there is a boat passage that is quite frequently-used in summertime, but the boats move slowly there due to the shallow and stony waters.

Protection and status

As the island of Maa-Sarvi belongs to the Bothnian Bay National Park, it has restrictions considering human activities. The flad is situated on the southern half of the island, where landing ashore and moving around closer than 100 m of the shoreline is forbidden during the bird nesting period, from May 1st to July 31st. According to observations made by Metsähallitus, the flad is virtually in a natural state and faces only low environmental pressures. Metsähallitus evaluates the conservation status of the flad as good.

The greatest threats to the aquatic environment of the area are contamination and other deterioration of water quality (Leinikki & Oulasvirta 1995). In the future, also maritime construction activities, climate change, and alien species may pose an increasing pressure to the aquatic species and habitats (HELCOM 2009c).

Indicators applied

Potential indicators suggested by Essi Keskinen at Metsähallitus are the numbers and coverages of charophyte, angiosperm, and endangered species, and the coverage of filamentous macroalgae. As the flad is practically in a natural condition, the present values can be used as reference conditions, as well. It should be noted that the charophyte diversity in this area is for natural, biogeographical reasons much lower than for instance in south-western Finland, and therefore can't be compared to the other case study flad, Danskogfladan (see below), in terms of biodiversity status. For a more profound discussion on choosing of biological indicators in flads, see Danskogfladan (below).

The nearest water quality monitoring station is at the open sea about 6.6 km east of the flad. The data from this station have been used as supporting data for the indicators (see BEAT calculations in chapter 5.2.3).

5.2.5 Danskogfladan flad

Physiographic limits

The flad Danskogfladan (also known as Ramsängsfladan) is at the island of Danskog in the archipelago of Tammisaari, in the sheltered inner archipelago zone east of the Hanko peninsula on the south coast of Finland (Figure 48). The flad is roughly elliptical in shape, with an area of 21 hectares. It has three narrow inlets: one in the north, one in the east and one in the south. The shores around the flad are mostly covered with forest. In addition, 4–5 summer cottages are set on the flad shores and they have jetties in the flad. Until the 1980's, there was some cattle grazing round the flad shores, but today the part of the island surrounding the flad is not used for any agricultural purposes.

Physical characteristics

Most parts of the flad are shallow, less than 2 m. The maximum depth in the vegetation survey transects was 2.7 m (Henricson 2008). The water exchange of Danskogfladan happens through the northernmost inlet, which is relatively exposed and has rocky shores. The southern and eastern inlets are overgrown by dense reed beds, and are thus nowadays closed up. In 1978 the eastern inlet was dredged open, temporally enabling water exchange and causing major alterations in the flow patterns and vegetation of the flad. By now, it's overgrown again and no water exchange occurs.

Several dredging activities have been performed in different parts of the flad during the decades (Henricson 2008). The latest was in 2005 in the inner, southernmost part, where a local summer resident illegally dredged a boat passage to their landing site and deposited the spoil in the water nearby. The same spot had been dredged in the early 1980's, too, and other minor dredgings have been done outside cottage jetties. The bottom of the flad consists mostly of soft, organic mud, patchily mixed with sand especially in the southernmost part. Only the northern inlet has enough exposure to prevent accumulation of organic matter, and the bottom there is rocky, stony and sandy.

As the flad has regular water exchange with the surrounding waters and only minor freshwater inflow, the salinity in the flad would likely be the same as in the waters in the area around. In the summers of 2003 and 2005 (14 measurements) the surface water salinity in the flad was about 4–6 ‰ (R. Munsterhjelm, unpublished), which is the same as the regular summer salinity in the area (data from the SYKE database). The flad's summer nutrient concentrations, instead, may differ from the surrounding waters. The lush vegetation of undisturbed, sufficiently isolated flads can lower the nutrient concentrations compared to surrounding waters. Macrophytes also stabilise the loose sediment surface, improving water transparency. On the other hand, dredging and recreational activities like boating may elevate the nutrient concentrations and decrease the water transparency in flads.

Biological characteristics

The macrophyte data and much of the background information about Danskogfladan were kindly provided by C. Munsterhjelm, also including data from R. Munsterhjelm. Both of them have carried out several surveys of the aquatic vegetation in the flad in the years 1978–2008. They have also had access to results of surveys made there by other researchers. Part of the data has been published by R. Munsterhjelm (1997, 2005) and reported in an assessment of dredging impacts on the flad's vegetation (Henricson 2008). In the two surveys (1978 and 2008) compared by Henricson (2008) and presented here (Table 18), the vegetation was assessed from transects made with boat, studied visually with a water field glass and sampled with a Luther rake. The abundance of macrophyte species was assessed on a seven-graded (1978) or four-graded (2008) scale from the individual study spots along the transects. Therefore, no species-specific coverage estimates can be given for the whole flad.

Most of the flad seabed consists of soft, partly sandy sediment. It is covered with fresh- and brackish-water vegetation characteristic of flads and other sheltered soft bottoms. In the latest vegetation survey of 2008, Henricson observed 23 species of aquatic vascular plants, macroalgae and mosses, while Munsterhjelm had found 26 species in 1978 (Table 18). Of these, 4 in 1978 and 5 in 2008 were charophytes. In the central, open part of the flad, the dominating macrophyte taxa in 2008 were *Myriophyllum* spp., *Najas marina*, *Ceratophyllum demersum* and *Potamogeton pectinatus*. The filamentous, mat-forming alga *Vaucheria* cf. *dichotoma*, which in the 80's and 90's dominated the central, open part of the flad, had now declined. The marginal, shallower parts of the flad are characterised besides the above-mentioned angiosperms (especially *Najas marina*), by *Ruppia maritima* and *Zannichellia palustris*, charophytes (the most abundant being *Chara aspera*), and in places, the moss *Drepanocladus aduncus*. The hard substrates of the northern inlet support some attached macroalgae like *Fucus vesiculosus*.

The innermost (southernmost) part of the flad is lined with a reed belt, *Phragmites australis*. The reed stands are here and there abundantly accompanied with the club-rush *Schoenoplectus tabernaemontani*. The reeds are expanding along with the flad becoming shallower and shore grazing being absent.

Table 18. Vascular plants (V), algae (Ch=Charophytes, MA=other macroscopic algae) and mosses (m) observed in Danskogfladan in the aquatic vegetation surveys of 1978 and 2008 (Henricson 2008)

Species		1978	2008
Bolboschoenus maritimus	V	x	
Callitriche hermaphroditica	V	x	x
Ceratophyllum demersum	V	x	x
Chara aspera	Ch	x	x
Chara baltica	Ch		x
Chara canescens	Ch	x	
Chara globularis	Ch		x
Chara tomentosa	Ch	x	x
Chorda filum	MA	x	x
Drepanocladus aduncus	m		x
Eleocharis acicularis	V	x	
Eleocharis parvula	V	x	
Fucus vesiculosus	MA	x	x
Lemna trisulca	V	x	x
Myriophyllun sibiricum	V	x	x
Myriophyllum spicatum	V	x	x
Najas marina	V	x	x
Phragmites australis	V	x	x
Potamogeton filiformis	V	x	
Potamogeton pectinatus	V	x	x
Potamogeton perfoliatus	V	x	x
Potamogeton pusillus	V	x	x
Ranunculus circinatus	V	x	x
Ruppia maritima	V	x	x
Schoenoplectus tabernaemontani	V	x	x
Tolypella nidifica	Ch	x	x
Typha angustifolia	V	x	
Vaucheria cf. dichotoma	MA	x	x
Zannichellia palustris	V	x	x
Number of species		26	23

Of the species in Table 18, *Chara baltica* is classified as near threatened in Finland (Rassi et al. 2001). A new evaluation of threatened species in Finland is in preparation and the status of some charophytes may change consequently, but the results are not yet available (Aino Juslén, SYKE, pers. comm.). According to HELCOM (2007), *Chara tomentosa* and *Fucus vesiculosus* are under threat or declining in other parts of the Gulf of Finland, but not in the Finnish waters. However, in the region around this case study, *C. tomentosa* is considered rare or sparse and declining, and is mainly known from flads (R. Munsterhjelm *et al.*, unpublished).

The water macrovegetation in the flad has changed significantly over the years, through both natural and human-induced causes (Henricson 2008). These include e.g. land uplift, the opening and eventual re-overgrowing of the eastern inlet, and ceasing of cattle grazing on the shores. Munsterhjelm (1997) states that compared to the early data of Häyrén from 1935, no major changes in the vegetation had occurred by 1978. After 1978 notable changes were discernible, though, due to dredgings in the late 1970's and early 1980's. Especially the opening of the eastern inlet in 1978 changed the vegetation drastically by altering the water circulation, destroying the homogenous *Chara tomentosa* beds and favouring *Ceratophyllum demersum* and *Myriophyllum* spp. (Henricson 2008). In addition, the ceasing of cattle grazing along the flad shores in the 1980's has led to major vegetation changes (Munsterhjelm 1997) that won't revert unless the grazing is restarted. For example, the five angiosperm species found in the survey of 1978 but not in 2008 (Table 2), probably disappeared because of the ending of grazing (Henricson 2008). Other important factors affecting the aquatic flora in flads include eutrophication, as well as physical disturbance and increasing water turbidity caused by dredging and recreational boating, fishing, and bathing (Munsterhjelm 2005). The small dredgings outside summer cottages are considered to have had only minor effects on the flad's state (C. Munsterhjelm, pers. comm.). No drastic changes in the aquatic vegetation have occurred in the 2000's (Henricson 2008).

The bottom fauna in Danskogfladan is dominated by insect larvae, both in biomass and abundance (C. Munsterhjelm, pers. comm.). Also Oligochaetes, *Hydrobia* snails, Ostracods, and water mites (Hydracarina) are abundant. In general, flads maintain a rich benthic fauna. The young-of-the-year fish species composition of the flad was surveyed in July-August of 2003 (Lappalainen & Urho 2006). The samples were taken by 6 hauls with a beach seine. A total number of eight species was found, three of them cyprinids (Table 19). The most abundant (as catch per unit effort) species was bleak, *Alburnus alburnus*. Gobies, *Pomatoschistus* spp., were numerous as well. No endangered species or alien species were found among the observed fish. As the macrophyte vegetation has remained roughly similar throughout the past ten years, the same can be surmised for the fish community. Therefore, the 2003 fish results (Table 19) can be considered to represent the present state. No other studies have been made on the fish of Danskogfladan.

Table 19. Young-of-the-year fish species in Danskogsfladan, sampled by beach seine in July–August 2003 (Lappalainen and Urho 2006). Origin: M = marine species, F = freshwater species. CPUE = catch per unit effort

Species		Origin	CPUE
Cyprinids:			
Roach	Rutilus rutilus	F	4
Tench	Tinca tinca	F	<0.5
Bleak	Alburnus alburnus	F	394
Other species:			
Perch	Perca fluviatilis	F	1
Pike	Esox lucius	F	1
Three-spined stickleback	Gasterosteus aculeatus	M	<0.5
Common/sand goby	Pomatoschistus spp.	M	40
Other species	(not specified)	(?)	<0.5

Functional properties and services

In southern Finland, flads often harbour an abundant charophyte flora, and serve as refuges for charophytes threatened in more open waters (Schubert & Blindow 2003). Charophyte species are numerous and abundant in the Danskogfladan, and sensitive species like *Chara tomentosa* and *C. baltica* are present. Therefore, it is a site of special importance for charophytes (Catherine Munsterhjelm, pers. comm.).

Being relatively sheltered, flads and resembling habitats are often important as valuable spawning and nursery areas for many fish species (Lappalainen & Urho 2006). Especially the aquatic macrophyte vegetation provides protective surroundings for the fish young, and harbours a rich invertebrate fauna (Henricson 2008). The recreational pressure on the flad is rather light, consisting mainly of the summer cottage habitation and the small-boat traffic to them. “Outsiders” seldom roam the flad.

Protection and status

Danskogfladan lies within the Natura 2000 site FI0100005, the Hanko and Tammisaari archipelago and Pojo Bay marine protected area. This Natura site is a part of the Baltic Sea Protected Area network recommended by HELCOM. In the valid local shore plan, the shore of the flad is appointed a nature protection area and should according to the plan dictations be maintained in a natural state. Building on the shore is restricted and construction of new jetties is prohibited.

In the assessment of threatened habitat types in Finland (Raunio et al. 2008), flads are defined as a habitat type of internationally especially great responsibility for Finland. After recurrent human disturbance in the past, the flad is now recovering, and evolving towards a more natural state (Henricson 2008). For example the charophyte vegetation has become more diverse and abundant in recent years. The improvement is much due to the overgrowth and subsequent re-clogging of the eastern inlet (which was dredged open in 1978), that has enabled the flow pattern to return to a more natural one (Henricson 2008). Today, the flad has returned to its original, pre-dredging state and is in a good condition (C. Munsterhjelm, pers. comm.). The flad is also approaching the following stage in the natural isolation succession of flads (Henricson 2008).

Indicators applied

The community composition of the aquatic vegetation is a good indicator of water quality and habitat condition (Munsterhjelm 2005). Therefore, it has been used in Danskogfladan, too, as a tool for assessing the impact of dredging and comparable operations on the aquatic environment (Henricson 2008). As the macrophyte flora of 1978 can be regarded as reflecting the situation in the early 20th century (see above), the results of the vegetation survey of 1978 can be used as a reference condition for a natural state of the flad. Coverages of macrophytes haven't been estimated, hence only numbers of taxa are available for indicator use.

Charophytes are particularly sensitive to water turbidity and mechanical disturbance (Munsterhjelm 2005). They might thus be useful as bioindicators (Schubert and Blindow 2003). Especially *Chara tomentosa* is an important indicator for human-induced environmental changes (Munsterhjelm 2005). That species was observed in Danskogfladan both in 1978 and 2008 (Table 18), but had actually first declined and then increased again between those years (Henricson 2008). *Chara baltica* (near threatened in the evaluation of Rassi *et al.* 2001) and *C. globularis* were observed in 2008 but not in 1978, which is another sign of improvement in the status of the flad. *Chara tomentosa*, *C. baltica* and *C.canescens* are sensitive species indicating a natural-like state (C. Munsterhjelm, pers. comm.), and they were found in Danskogfladan. During the 20th century, a general decline of charophytes has occurred in several countries bordering the Baltic Sea, mainly because of eutrophication and habitat destruction (Schubert and Blindow 2003). In this specific archipelago area, the decline concerns *Chara aspera*, *C. baltica* and *C.canescens*, and especially *Chara tomentosa* (Schubert and Blindow 2003). Yet, both species number and coverage of charophytes have increased in Danskogfladan (Table 18, Henricson 2008).

It should be acknowledged, though, that the macrophyte vegetation of a flad is greatly dependent on natural environmental factors like the flad's developmental stage in the succession of flad isolation by land upheaval, the salinity and temperature regimes, seabed type, and biogeographical region. For example, the diversity of the vegetation decreases during flad development (Munsterhjelm 2005). Consequently, the flad flora is very site- and time-specific, and it might prove truly challenging to find universal indicators for the biodiversity status of flads in general. Also, especially when using higher taxa in the indicators it must be remembered that e.g. within a plant or algal genus or family, the individual species may be influenced differently by particular environmental changes. On the other hand, certain species may have wide environmental tolerances and should thereby be avoided in inference about the status. For instance, *Ceratophyllum demersum* and *Myriophyllum* spp. are tolerant of elevated nutrient concentrations and turbidity and may thus indicate eutrophication, but are also found in waters of a natural state. Therefore, it is crucial to note the identities, not only the number of the species. On the whole, the adequacy of indicators, including their reference conditions and acceptable deviations, is greatly case-specific.

Valid, generally accepted indicators for the assessment of coastal fish communities in the Baltic Sea are so far lacking. Such indicators should be selected according to the properties of local fish populations, and the assessment should always include indicators involving the most dominant species (HELCOM 2006). Like for macrophytes, the fish species composition in flads is very site-specific, also naturally. It depends on e.g. physical environmental factors like salinity, temperature, exposure and bottom quality, and also on the sampling method (HELCOM 2006, Lappalainen & Urho 2006). For example, the number of fish species usually decreases with decreasing salinity (www.helcom.fi). Furthermore, the fish community becoming cyprinid-dominated may indicate eutrophication if the biomass increases at the same time, but also other factors may modify the community structure (HELCOM 2006). Therefore, defining suitable status indicators and their reference conditions and acceptable deviations requires profound knowledge of the local environmental characteristics. Potential indicators suggested by HELCOM to describe the status of the coastal fish fauna in the Baltic Sea are species richness, relative abundance (catch per unit effort) of species, the ratio between functional groups, and the trophic level of fish communities (HELCOM 2006). These indicators are however based on all age classes and many also on biomass. As the available fish data for Danskogfladan only comprises young-of-the-year fish abundances, it cannot be used for indicator construction. The fish data may therefore be regarded as biological background data only, reflecting the fish species that typically spawn in flads of the area.

The nearest water quality monitoring station with both winter and summer sampling is in the inner archipelago zone about 4 km northwest of the flad. The data from this station may be used as supporting data for the indicators (see BEAT calculation in chapter 5.2.3). For the winter nutrients this should be acceptable, but as mentioned above, in summer the water transparency may be better and nutrient concentrations lower inside the vegetated flad than in surrounding waters. Water quality variables have not been regularly monitored inside the flad, though (only occasional measurements exist), and are therefore not available as supporting data.

5.2.6 Granbusken reef

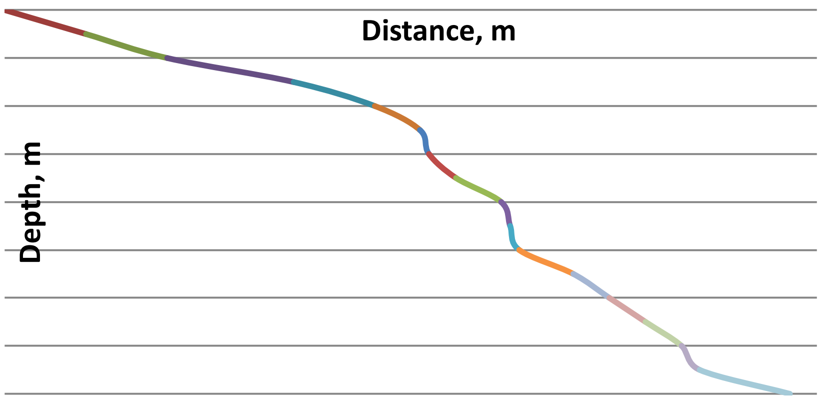
Physiographic limits

The reef is at the small (2.3 ha) island of Granbusken in the outer archipelago zone of the Hanko peninsula on the south coast of Finland (Figure 52). The reef studied here is on the eastern, exposed shore of the island. The island of Granbusken is rocky and relatively high, with coniferous vegetation on the top. The island is uninhabited and strictly protected (see below), and has thus no on-site human impacts.

Physical characteristics

Reefs are hard-substratum habitats (European Commission 2007). Submerged rocky shores with algal zonation are included in this habitat type, and they are common in the Finnish outer archipelagos (Airaksinen & Karttunen 2001). All shores of the Granbusken Island are mostly rock down to a water depth of about 20 m. In the deepest parts, small boulders, stones, gravel, and sand co-occur with the rock. No soft sediments are present. The uppermost slope at the studied part of the shore reef is fairly gentle, deepening to 7 m on a 50 m distance from the water's edge (Figure 49). Tides being absent in the Baltic Sea, the reef is permanently submerged.

Figure 52. The shore profile at the studied site of the Granbusken reef



As the reef is exposed and the water exchange is very good, salinity and nutrient concentrations at the reef are the same as in the surrounding waters. The surface water salinity at the nearest water quality monitoring station 1,5 km east of the reef is about 5–6,5 ‰ (data from the SYKE database). Upwelling events may occasionally elevate the water nutrient concentrations at the reef.

Biological characteristics

On wave-exposed rocky shores (reefs) of south-western Finland, typically three algal belts can be distinguished: an upper zone of annual filamentous algae, the bladder wrack (*Fucus vesiculosus*) belt, and deepest, the red algal zone (Airaksinen & Karttunen 2001). Within the algal community there is prominent successional development during the growth season. Ice scraping prevents the growth of perennial algae in the upper eulittoral (Schubert & Schories 2008). The maximum growth depth of *Fucus vesiculosus* on exposed shores of this coastal area is 5–6 m, with an optimum at 2–3 m (Bäck & Ruuskanen 2000).

The Granbusken reef consists of a gradually deepening rocky shore slope with characteristic zonated macroalgal vegetation. On such a rocky substrate, no other macrophytes occur. On this site, several vegetation transects have been made since 1998, but the study details and researchers have varied. In the last decade, the maximum growth depth of the bladder wrack was 3.9 m (average of four years). In 2009, its optimum growth depth was 2–3 m, where the coverage was 90% (Table 20). The macrophyte data was obtained from the SYKE database of aquatic macrophytes. The example data of 2009 (Table 20) was collected by A. Ruuskanen in 2009 from a vegetation transect by scuba diving. The coverage of algal species was estimated from 1 m^2 squares at 1 m depth intervals. The total length of the transect was 87 m and depth 16 m.

The benthic fauna at the reef is typical of exposed northern Baltic Sea rocky shores (A. Ruuskanen, pers. comm.). For instance, a healthy blue mussel (*Mytilus trossulus*) band occurs, and isopods (*Idotea* spp.) and snails graze on the algae.

Functional properties and services

The bladder wrack (*Fucus vesiculosus*) is one of the most important Baltic phytobenthic species, having high biomass and productivity, and serving as a habitat for species-rich communities of associated organisms (www.helcom.fi). The community supported by *Fucus* in one of the most diverse in the Baltic Sea (Airaksinen & Karttunen 2001). Therefore, the state of the bladder wrack belt is likely to markedly influence the whole coastal ecosystem. In general, reefs are of Baltic-wide importance and are in many cases hot spots for the biodiversity (www.helcom.fi). The strongest threat to both *Fucus* and red algae is eutrophication. At the Granbusken reef, the *Fucus* belt is well developed and healthy.

The Granbusken reef also has exceptionally good growth conditions for the red algae *Furcellaria lumbricalis* and *Phyllophora pseudoceranoides*, which are able to grow quite deep at this site. The reef and the island where it lies have no recreational, economical or land use pressures because of the strict protection (see below). This ensures the habitat remaining unaffected by on-site human disturbance for the time being.

Table 20. Coverage (%) of macroalgae in depths 0–13 m at the Granbusken reef on 14.7.2009. At 14–16 m depths, the rocky bottom was replaced with the finer substrates and only Phyllophora pseudoceranoides was present at a coverage of 1%. Bottom substrate (main substrate in bold): ro = rock, bo = small boulder, st = stone, gr = gravel. Group: G = green, B = brown, R = red algae. Type: a = annual, p = perennial

Depth, m			**0**	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	**13**
Distance, m			0	9	18	32	41	46	47	50	55	56	57	63	67	71
Bottom substrate			ro	ro	ro	ro	ro	ro	ro	ro	ro, st	bo, st, gr	ro, st, gr	ro, st, gr	ro, bo, st, gr	ro
Species	**Group**	**Type**					**Coverage of algal species,%**									
Cladophora glomerata	G	a	100	60												
Pilayella littoralis	B	a		20												
Fucus vesiculosus	B	p			90	*)	*)									
Cladophora rupestris	G	p			30	10	40									
Ectocarpus siliculosus	B	a				80	90			60	40	10				
Ceramium tenui-corne	R	a		10		5	5	80	40	5	10		5	5	5	
Coccotylus truncatus	R	p				1										
Furcellaria lumbi-calis	R	p							10	20	10	10	5	5	5	10
Phyllophora pseudocera-noides	R	p									5	5	30	30	20	5

*) *Fucus* had been grazed away

Protection and status

The island of Granbusken and some of the surrounding islands have been a nature protection area since 1957, when they were set under protection to be reserved for research and educational purposes by the Tvärminne biological station. In practice, the islands have a protection status of a strict nature reserve, and even landing ashore on the islands is allowed only with permission by the biological station. In addition, the reef lies within the Hanko and Tammisaari archipelago and Pojo Bay marine protected area, Natura 2000 site FI0100005 (the same as for Danskogsfladan above), which is a part of the Baltic Sea Protected Area network recommended by HELCOM.

The reef faces relatively little anthropogenic pressure, and its state can be considered as close to natural as possible (A. Ruuskanen, pers. comm.). The water exchange is good and the macroalgal biodiversity is high. The ongoing eutrophication of the Baltic Sea may affect the reef in general, but no specific effects e.g. from industry on the coast or ship traffic at the open sea can be observed on the Granbusken reef. The macroalgal vegetation of the Granbusken reef has been monitored by the regional environmental authorities for more than ten years. No drastic change in its condition has been observed, though year-to-year variation exists in growing depths and coverages of the macroalgal species. Occasional, presumably weather-inflicted outbursts of filamentous algae may temporarily regress the bladder wrack, but it will soon recover due to good resilience induced by the favourable growth environment.

Indicators applied

The best indicator to be gathered from the macrophyte data available from the Granbusken reef, is the lower depth limit of the *Fucus vesiculosus* belt. Similar depth limits for the red algae *Furcellaria lumbricalis* and *Phyllophora pseudoceranoides* have successfully been used for indicator purposes elsewhere, but have not been tested in Finland and shouldn't thus be used here (A. Ruuskanen, pers. comm.). Other variables like coverages and sizes of certain species, or ratios of different taxa or groups at certain depths are also possible, but are not tested in Finland, either. Also, because the study method has varied some from year to year and consequently the results vary (besides the standard monitoring, new study methods have been tested at the site), choosing of the most representative and truthful study year may prove difficult. However, for the sake of the example, another indicator chosen for this work is coverage of *Fucus vesiculosus* on its optimal growth depth (in 2009).

The nearest water quality monitoring station is at the open sea about 1.5 km east of the reef. The data from this station can be used as supporting data for the indicators (see BEAT calculations chapter 5.3).

5.3 Application of BEAT, the HELCOM Biodiversity Assessment Tool

Assessing conservation status of a specific area, e.g. a Natura 2000 area, is hindered by the lack of widely accepted multi-metric indicator-based assessment tools. Currently, only a few tools have been developed and applied in regard to integrated assessment of biodiversity status, e.g. the HELCOM Biodiversity Assessment Tool BEAT and the Norwegian Nature Index (HELCOM 2009, 2010 and Certain 2011).

Here, we demonstrate how the indicators including site-specific target values for total cover of submerged aquatic vegetation developed by MOPODECO are useful in regard to assessment of conservation status within four Natura 2000 areas, all being stone reefs, in the Kattegat.

The Kattegat forms a transition between the Baltic Sea, located south of the Kattegat, and the North Sea / Skagerrak system, located north of the Kattegat. Detailed descriptions of the study area can be found in Andersen et al. 2010, whilst comprehensive descriptions and assessments of the Baltic Sea and the North Sea / Skagerrak system can be found in HELCOM (2010) and OSPAR (2010), respectively.

The stone reefs included in the study are: Hertha's Flak, Kim's Top, and Store Middelgrund. Reference conditions for total cover of erect macroalgal vegetation for each reef have been estimated based on an empirical model with the use of estimated nitrogen load to Kattegat in pristine drainage areas from Denmark and Sweden as described in Dahl & Carstensen (2008). Acceptable deviation is for demonstration purposes estimated as the vegetation cover equal to the load to Kattegat that was observed in 1997. This year was characterized by very low precipitation over Denmark and southern Sweden resulting in a considerable lower nutrient load to Kattegat (Dahl & Carstensen 2008)

The assessment of conservation status at four stone reefs in the Kattegat is based on information on the total cover of macroalgae at different depths. The data used cover data sampled in the years 2008, 2009 and 2010. For this indicator, synoptic information is required in regard to reference conditions (RefCon), acceptable deviation for reference conditions (AcDev) and the present state of biological diversity (State). The methodology for estimation of this information is found Krause-Jensen et al. (2008).

The assessment tool used, BEAT, was originally developed for assessment of biodiversity status of the Baltic Sea region (HELCOM 2009, 2010). The tool is a clone of the HELCOM Eutrophication Assessment Tool HEAT (Andersen et al. 2010, 2011), which has been developed and used for assessment of eutrophication status in the Baltic Sea and North Sea regions. The key differences between BEAT and HEAT are: BEAT makes use of biodiversity indicators, while HEAT only makes used of indicators related to nutrient enrichment and eutrophication effects; and 2) in BEAT, the indicators are grouped under the following four themes: i) broad-scale habitats/landscapes, ii) communities, iii) species and iv) supporting indicators.

As a first step, a Biodiversity Quality Objective (BQO), or target, is calculated per indicator:

Equation 1: BQO = RefCon ± AcDev

The above approach used here is based largely on the WFD, especially regarding the emphasis on defining reference conditions and acceptable deviations. This approach is largely compatible with the Habitats Directive.

Step 2 is to calculate a Biodiversity Quality Ratio (BQR), which is directly comparable with the Ecological Quality Ratio principle sensu the WFD (Andersen et al. 2011). The BQR approach used in this assessment marks the ratio (0 to 1) between BioState and the RefCon, i.e. RefCon/State if degradation increases indicator value, otherwise the inverse. For indicators that have a numerically positive response to a given pressure factor, for example, primary production, the border between "Unaffected by human activities" and "Moderately affected by human activities" is calculated as:

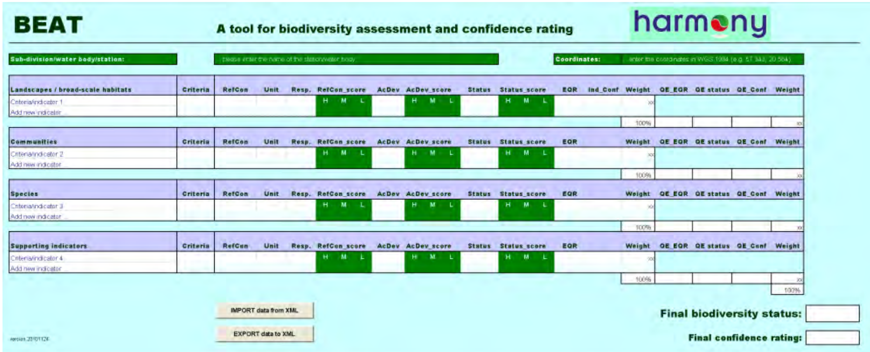
Equation 2: If State ≤ RefCon x (1+AcDev in decimal form), i.e. if BQR > 1/(1+AcDev in decimal form), then BQO is fulfilled for the indicator in question.

For indicators that have a numerically negative response to degradation (e.g., population sizes of a fish species or depth distribution of submerged aquatic vegetation), the status is calculated as:

Equation 3: If State ≥ RefCon x (1 – AcDev in decimal form), i.e. if BQR> (1 – AcDev in decimal form), then BQO is fulfilled.

Step 3 is about combining indicators within four categories: 1) broad-scale habitats/marine landscapes, 2) communities, 3) species, and 4) supporting indicators. The classifications are, as a first step, based on a weighted average of the BQO and BQR values within a category. Weights are established via expert judgement. If not specified otherwise, the weighting is kept neutral by giving each of the indicators equal weights. Further, on the basis of the BQR and AcDev values, each category was given a quantitative assessment according to the principles described above for a single indicator. It should be noted that an indicator has a BQO and thus only to two "classes," whilst a category has five classes (high, good, moderate, poor and bad). More information on how classes are defined can be found in Dahl & Carstensen (2008).

Figure 53. A clean BEAT sheet. Please note the structure with four categories of indicators: i) broad-scale habitats/landscapes, ii) communities, iii) species and iv) supporting indicators



In step 4, the results of the four categories are combined by applying the so-called "One out – All out" principle to the Categories I–IV. This implies that the category most sensitive to human activities defines the overall status of biodiversity within an assessment unit. However, in this demonstration, we have decided on not to include any other data / indicators that those developed by MOPODECO. The consequence is that the assessments are made for "Communities" only.

The assessment of conservation status at the stone reefs in the Kattegat is based on information on the total cover of long-lived macroalgae at different depths. The assessment reveals that the status generally tends to be unfavourable. The results are summarised in Table 21.

The Biological Quality Ratio was lowest in 2008 with an average of 0.544 and for all three reefs the conservation status was assessed to be bad. In 2009, the average BQR increased to 0.700 indicating a slightly better conservation status. At Kim's Top, the conservation status improved to poor. The trend continued in 2010, where the average BQR was 0.855. At Hertha's Flak, the conservation status was assessed to be high in 2010, while Kim's Top and Store Middelgrund were both classified as having a moderate conservation status.

Table 21. Biological Quality Ratios (BQR) and classifications of conservation status for "Communities" at Hertha's Flak (2008–2010), Kim's Top (2008–2010), Store Middelgrund (2008–2010) og Lilla Middelgrund (2006)

Reef	2008	2009	2010			
	BQR	Status	BQR	Status	BQR	Status
Hertha's Flak	0,640	Bad	0,803	Bad	0,985	High
Kim's Top	0,491	Bad	0,655	Poor	0,776	Moderate
Store Middelgrund	0,502	Bad	0,643	Bad	0,805	Moderate

Although the above assessments are based on a limited set on indicators, the results are in accordance with other more or less similar assessments made in the Kattegat, e.g. HELCOM (2010). However, assessing stone reefs without including indicators being proxies for benthic invertebrates and non-commercially exploited fish species might potentially bias the final classification of conservation status. The results of this demonstration should thus be regarded as preliminary. On the other hand, acknowledging that long-lived macroalgae are characteristic for light exposed stone reefs, it would be prudent to conclude that any future assessments of conservation status for communities living on stone reefs ought to include macroalgae.

The use of multi-metric indicator-based assessment tools is currently growing. The tools as such should be regarded as a framework relying on good indicators. A first step toward better assessments of conservation status would be to develop more and better biodiversity indicators. A future next step would then be to optimize the tools. This should be done once better "input" data exist.

6. Conclusions

MOPODECO has accomplished the main aims of the project, thus supporting the Environmental Action Programme 2009–2012 of the Nordic Council of Ministers in several ways, including assisting the common strategy for sustainable development and implementation of the EC Marine Strategy in the Nordic environment. The harmonization of environmental protection work in the Nordic countries has been strengthened both through the establishment of a Pressure Evaluation Matrix and suggestions for harmonisation of definitions of Annex I habitats in the Baltic Sea. The Pressure Evaluation Matrix will further support the sustainable use of the resources in the Baltic Sea, and the basis for ecosystem approach to management in this regional sea. The deliverables from the conceptual indicator models, the macroalgal cover indicator model, bladderwrack and blue mussel models have all greatly assisted priorities related to the development of both habitat mapping tools and EcoQOs for the Baltic Sea. Albeit limitations, the potential for developing reliable regional statistical models of the coverage of habitat forming species for the description of the biological features of Annex I habitats has been demonstrated, and provided availability of high-resolution data on bathymetry and surface sediments is secured successful applications could be a reality within all Nordic waters in the medium term (5–10 years). The application of deterministic, ecological models for prediction of the potential regional coverage of blue mussels and other communities of suspension-feeding animals could be reality in the short term (0–5 years). The transnational analysis of vegetation data from Norwegian waters to the Gulf of Finland has indicated that total vegetation cover might both represent one of the best functional descriptors of reefs, and provide excellent input data on reference and target conditions for assessing the conservation status of reef habitats in Skagerrak and the Baltic Sea. The great advantage of total macro-algae cover is that it can be used over a large depth range, whereas cumulative cover can be used as a substitute in more shallow waters while controlling for physical disturbance. Once better input data become available, the habitat model and assessment tool applications constitute case studies which can easily be taken up and further refined by HELCOM and national institutes involved in developing assessment tools for biodiversity and conservation status of the species and habitats in the Baltic Sea.

7. References

Airaksinen, O. & Karttunen, K. 2001: Natura 2000 -luontotyyppiopas. Oy Edita Ab, Helsinki.

Airaksinen, O. and Karttunen, K. (2001): Natura 2000 –luontotyyppiopas. 2. korjattu painos. Ympäristöopas 46. Suomen ympäristökeskus, Helsinki. 194 pp. (Natura 2000 habitat manual; in Finnish).

Al-Hamdani, Z. and Reker, J. (eds.), 2007. Towards marine landscapes in the Baltic Sea. BALANCE interim report #10. Available at http://balance-eu.org/.ISBN: 978-87-7871-203–5.

Andersen, J.H., C. Murray, H. Kaartokallio, P. Axe & J. Molvær. 2010. A simple method for confidence rating of eutrophication status classifications. Marine Pollution Bulletin 60:919–924.

Andersen, J.H., P. Axe, H. Backer, J. Carstensen, U. Claussen, V. Fleming-Lehtinen, M. Järvinen, H. Kaartokallio, S. Knuuttila, S. Korpinen, M. Laamanen, E. Lysiak-Pastuszak, G. Martin, F. Møhlenberg, C. Murray, G. Nausch, A. Norkko& A. Villnäs. 2011. Getting the measure of eutrophication in the Baltic Sea: towards improved assessment principles and methods. Biogeochemistry 106:137–156.

Andrulewicz E, Kruk-Dowgiallo L, Osowiecki A (2004) Phytobenthos and macrozoobenthos of the Slupsk Bank stony reefs, Baltic Sea. In: Kautsky H, Snoeijs P (eds), Biology of the Baltic Sea. Kluwer Academic Publishers, Netherlands. Hydrobiologia 514: 163–170.

Andrulewicz E, Wielgat M (1999) Selection of southern Baltic banks – future marine protection areas. In: Andrulewicz, E. 1997: An overview on lagoons in the Polish coastal area of the Baltic Sea. International Journal of Salt Lake Research 6:121–135.

Anonymous 2000. Environmental Quality Criteria, Coasts and Sea. 5052. Swedish Environmental Protection Agency. 138 pp.

Anonymous 2006. Baltic Sea Millestone 6 report. Coastal GIGs. European Comission. 39 pp.

Anonymous 2007: Website of German Federal Agency of Nature Conservation. Available at http://www.bfn.de/0316_typ1150+M52087573ab0.html

Aunins A., Bambe B., Engele L., Ikauniece S., Kabucis I., Laime B., Larmanis V., Reriha I., Rove I., Rusina S., Salmina L., Sniedze R. 2010. Eiropas savienibas aizsargajamie biotopi Latvija. Noteiksanas rokasgramat. A. Aunins (red.), Latvijas Dabas fonds, 319 p.

Baden S, Boström C, Tobiasson S, Arponen H, Moksnes PO. 2010. Relative importance of trophic interactions and nutrient enrichment in seagrass ecosystems: A broad-scale field experiment in the Baltic–Skagerrak area. Limnology and Oceanography 55: 1435–1448.

Bäck, S. and Ruuskanen, A. 2000. Distribution and maximum growth depth of *Fucus vesiculosus* along the Gulf of Finland. Marine Biology 136: 303–306.

Berger, R., Bergström, L., Granéli, E. & Kautsky, L. 2004. How does eutrophication affect different life style stages of Fucus vesiculosus in the Baltic Sea? – a consept model. – Hydrobiologia 514: 243–248.

Bijl, L. van der, Boutrup, S. & Jensen, P.N. (red) 2007: NOVANA. Det nationale program for overvågning af vandmiljøet og naturen. Programbeskrivelse 2007–2009 – del 2. Danmarks Miljøundersøgelser. 120 s. – Faglig rapport fra DMU nr. 615

Blæsbjerg, M., Pawlak, J.F., Sørensen, T.K. and Vestergaard, O. 2009. Marine Spatial Planning in the Nordic region – Principles, Perspectives and Opportunities. Nordic Council of Ministers. TemaNord 2009: 528.

Blindow, I., 2000. Distribution of charophytes along the Swedish coast in relation to salinity and eutrophication. Int. Rev. Hydrobiol. 85:707–717.

Blomqvist EM, Bonsdorff E, Essink K (eds) Biological, Physical and Geochemical Features of Enclosed and Semi-enclosed Marine Systems. Kluwer Academic Publishers, Netherlands. Hydrobiologia 393: 271–277.

Boedeker D. 2001: Identification and demarcation of marine habitats in Germany. In. Application of NATURA 2000 in the Marine Environment. Workshop at the International Academy for Nature Conservation (INA) on the Isle of Vilm (Germany) from 27 June to 1 July 2001. German Federal Agency for Nature Conservation (BfN), p. 19.

Boedeker D. Krause J.., von Nordheim H. (2006). Interpretation, identification and ecological assessment of the NATURA 2000 habitats “sandbank” and “reef.” In: von Nordheim H., Boedeker D. and Krause J.C. (eds.) Progress in marine conservation in Europe. Natura 2000 sites in German offshore waters. Springer Berlinn Heidelberg New York, 47–64.

Bonsdorff, E. and Blomqvist, E.M. 1993. Biotic couplings on shallow water soft bottoms—examples from the northern Baltic Sea. Oceanogr. Mar. Biol. Annu. Rev. 31: 153–176.

Bonsdorff, E., Blomqvist, E.M., Mattila, J. and Norkko, A. 1997. Long-term changes and coastal eutrophication. Examples from the Åland Islands and the Archipelago Sea, northern Baltic Sea. Oceanologica Acta 20: 319–329.

Boström, C. 2001. Ecology of seagrass meadows in the Baltic Sea. Åbo Akademi, Finland. PhD-thesis. 47 pp.

Boström, C. 2006.Distribution and structure of angiosperm communities in the Archipelago Sea, SW Finland. Research report for the VELMU pilot project “BIOGEO” (Linking Marine Key Biotopes and Geological Features: A Pilot Survey of Macrophyte Communitites on Sublittoral Moraines). Finland’s Ministry of the Environment. 20 pp.

Bostroöm, C., Baden, S.P., Krause-Jensen, D., 2003. The seagrassesof Scandinavia and the Baltic Sea. In: Green, E.P., Short, F.T.

Bučas, M., Daunys, D., Olenin, S., 2007. Overgrowth patterns of the red algae Furcellaria lumbricalis at an e<xposed Baltic Sea coast: The results of a remote underwater video data analysis. Estuarine and Coastal and Shelf Science 75(3): 308–316.

Carstensen, J., Krause-Jensen, D., Dahl, K. & Henriksen, P. 2008. Macroalgae and phytoplankton as indicators of ecological status of Danish coastal waters. National Environmental Research Institute of Denmark, NERI Technical Report No. 683. 90 pp.

Certain, G., O. Skarpaas, J.-W. Bjerke, E. Framstad, M. Lindholm, J.-E. Nilsen, A. Norderhaug, E. Oug, H.-C. Pedersen, A.-K. Schartau, G. van der Meeren, I. Aslaksen, S. Engen, P.-A. Garnåsjordet, P. Kvaløy, M. Lillegård, N.G.Yoccoz & S. Nybø. 2011. The Nature Index: A General Framework for Synthesizing Knowledge on the State of Biodiversity. PLoS ONE 6(4):1–14.

Cloern JE. 2001. Our evolving conceptual model of the coastal eutrophication problem. Marine Ecology Progress Series 210: 223–253.

Conley DJ, Paerl HW, Howarth RW, *et al.* 2009. Controlling eutrophication: nitrogen and phosphorus. Science 323: 1014–15.

Dahl K, Leth JO, Al-Hamdani Z, Carstensen J, Lundsteen S, Anthony D. 2007. Mapping marine habitats in Kattegat. BALANCE Interim Report No.21.

Dahl, K & Carstensen, J. 2005. Hårdbundsvegetation som indikator for naturkvalitet og bevaringsstatus på stenrev. In: Dahl, C., Andersen, J.H., Riemann, B. (eds), Carstensen, J., Christiansen, T., Krause-Jensen, D., Josefson, A.B., Larsen, M.M., Petersen, J.K., Rasmussen, M.B., Strand, J. Redskaber til vurdering af miljø- og naturkvalitet i de danske farvande. Typeinddeling, udvalgte indikatorer og eksempler på klassifikation. Technical report from NERI no. 535 (In Danish with English summary), pp 26–53

Dahl, K. & Carstensen, J. 2008. Tools to assess conservation status on open water reefs in Nature-2000 areas. National Environmental Research Institute of Denmark, NERI Technical Report, No. 663. 25 pp.

Dahl, K. et al. 2004. Tools to assess the conservation status of marine Annex 1 habitats in Special Areas of Conservation. Phase 1: Identification of potential indicators and available data. National Environmental Research Institute of Denmark, NERI Technical Report, No. 488. 96 pp.

Dahl, K., Larsen, M.M., Andersen, J.H., Rasmussen, M.B., Petersen, J.K., Josefson, A.B., Lundsteen, S., Dahllöf, I., Christiansen, T., Krause-Jensen, D., Hansen, J.L.S., Ærtebjerg, G., Henriksen, P., Helmig, S.A. & Reker, J. 2004: Tools to assess the conservation status of marine Annex 1 habitats in Special Areas of Conservation. Phase 1: Identification of potential indicators and available data. National Environmental Research Institute, Denmark. 96 pp. – NERI Technical Report No. 488.

Dahl, K. & Carstensen, J. 2008. Tools to assess conservation status on open water reefs in Natura-2000 areas.National Environmental Research Institute, University of Aarhus.25 pp. – NERI Technical Report No. 663. http://www.dmu.dk/Pub/FR663.pdf

Daunys D., Bucas M., Zemlys P. (2008). Spatial prediction of Furcellaria lumbricalis in Lithuanian coastal waters. In: Dienesen (ed.) Mapping and modelling of marine habitat in the Baltic Sea. Balance Interim Report 27 (Draft): 178–181.

Daunys, D., Rinne, H., Downie, A.-L. and Gullström, M. (2008): Harmonisation of the definitions of Annex I habitats. WP1 Deliverable Report, MOPODECO project. Draft version (15 December 2008).

Davies, J., Baxter, J., Bradley, M., Conner, D., Khan, J., Murray, E., Sanderson, W., Turnbull, C. & Vincent, M. (eds.) 2001. Marine Monitoring Handbook. Joint Nature Conservation Committee, 405 pp.

Duarte, C. M., 1991. Seagrass depth limits, Aquat. Bot. 40: 363–377.

Eggert, A., Ihnken, S., Selig, U., Karsten, U., Schubert, H. 2006: Distribution of three submersed macrophytes in coastal lagoons of the German Baltic Sea: comparison of laboratory and field data. Botanica Marina 49:386–395.

Ehler, C. and Douvere, F. (2009). Marine Spatial Planning: a step-by-step approach toward ecosystem-based management. Intergovernmental Oceanographic Commission and Man and the Biosphere Programme. IOC Manual and Guides No. 53, ICAM Dossier No. 6. Paris: UNESCO.

Ekebom, J., Jäänheimo, J. and Reker, J. (editors; 2008). Towards Marine Spatial Planning in the Baltic Sea. BALANCE Technical Summary Report 4/4. Available at: http://balance-eu.org

Elith J, Graham C H, Anderson R P, Dudı´k M, Ferrier S, Guisan A, Hijmans R J, Huettmann F, Leathwick J R, Lehmann A, Li J, Lohmann L G, Loiselle B A, Manion G, Moritz C, Nakamura M, Nakazawa Y, McC. Overton J, Townsend Peterson A, Phillips S J, Richardson K, Scachetti-Pereira R, Schapire R E, Sobero´n J, Williams S, Wisz M S and Zimmermann N. E., 2006. Novel methods improve prediction of species" distributions from occurrence data. Ecography 29: 129–151.

Eriksson, B. K. & Johansson, G. 2003: Sedimentation reduces recruitment success of Fucus vesiculosus (Phaeophyceae) in the Baltic Sea. – Eur. J. Phycol. 38: 217–222.

Eriksson, B.K. & Bergstrom, L. 2005. Local distribution patterns of macroalgae in relation to environmental variables in the northern Baltic Proper. Estuarine Coastal and Shelf Science 62, 109–117."

Eriksson, B.K., Sandström, A., Isæus, M, Schreiber, H. & Karås, P. 2004. Effects of boating activities on aquatic vegetation in the Stockholm archipelago, Baltic Sea. Esturiane, Coastal and Shelf Science 61:339–349.

European Commission (2007a). Guidelines for the establishment of the Natura 2000 network in the marine environment. Application of the Habitats and Birds Directives. (EU interpretation manual). European Commission (2007b). Interpretation Manual of European Union Habitats – EUR27.

European commission DG environment (2007): Interpretation manual of European Union habitats – EUR27.

Federal Agency for Nature Conservation, 2010. http://www.bfn.de/0316_typ 1110+M52087573ab0.html accessed on 2010 06 20.

Fonseca, M.S., Bell, S.S., 1998. Influence of physical setting on seagrass landscapes near Beaufort, North Carolina. Mar. Ecol. Prog. Ser. 171, 109–121.

Frederiksen, M., Krause-Jensen D., Holmer, M. & Laursen, J. 2006. Long-term changes in area distribution of eelgrass (*Zostera marina*) in Danish coastal waters. Aquatic Botany 78: 167–181.

Gasiūnaitė Z., Daunys D., Olenin S., Razinkovas A. (2008). The Curonian Lagoon. In: Ecology of Baltic Coastal Waters (ed. U. Schiewer). Ecological Studies 197: 197–215.

Gilek M., Littorin B., Saetre P. (2001). Spatial patterns of abundance and growth of Mytilus edulis on boulders in the northern Baltic Sea proper. Hydrobiologia 452: 59–68.

Gotland Municipality (2005): Conservation plan for Natura 2000 sites: Hoburg Bank.

Guidelines for the establishment of the Natura 2000 network in the marine environment (2007): Application of the Habitats and Birds Directives.

Habitats and species covered by the EEC Habitats Directive. A preliminary assessment of distribution and conservation status in Denmark. – National Environmental Research Institute, Denmark. 121 pp. – NERI Technical Report No 365.

Hagerhal B, Miljo A, Skov H (1998) Proposal for offshore Baltic Sea Protected Areas (BSPAs), Expert report for HELCOM EC – NATURE. Pp 103.

Hauxwell J, Cebrián J-. & Valiela I. 2003. Eelgrass (*Zostera marina*) loss in temperata estuaries: relationship to land-derived nitrogen loads and effect of light limitation imposed by algae. Mar Ecol Prog Ser 247:59–73.

HELCOM. 1998. Red list of marine and coastal biotopes and biotope complexes of the Baltic Sea, Belt Sea and Kattegat. Balt. Sea Environ. Proc. No. 75.

HELCOM 2006. Assessment of coastal fish in the Baltic Sea. Baltic Sea Environmental Proceedings No. 103A. Helsinki Commission, Helsinki. 28 pp.

HELCOM 2007. HELCOM lists of threatened and/or declining species and biotopes/habitats in the Baltic Sea area. Baltic Sea Environmental Proceedings, No. 113.

HELCOM. 2009. Biodiversity in the Baltic Sea – An integrated thematic assessment on biodiversity and nature conservation in the Baltic Sea. Balt. Sea Environ. Proc. No. 116B.

HELCOM 2009. Eutrophication in the Baltic Sea – An integrated thematic assessment of the effects of nutrient enrichment and eutrophication in the Baltic Sea region. Balt. Sea Environ. Proc. No. 115B.

HELCOM. 2009. Fact sheets on threatened and/or declining species and biotopes/habitats. Available at: HELCOM 1998: Red list of marine and coastal biotopes and biotope complexes of the Baltic Sea, Belt Sea and Kattegat. Baltic Sea Environment Proceedings No. 75.

HELCOM. 2010. Ecosystem health of the Baltic Sea 2003–2007. HELCOM Initial Holistic Assessment. Baltic Sea Environment Proceedings 120. Available at: www.helcom.fi/publications.

HELCOM. 2010. Towards a tool for quantifying the anthropogenic pressures and potential impacts in the Baltic Sea marine environment. A background document on the method, data preparation and testing of the Baltic Sea Pressure and Impact Indices. Baltic Sea Environment Proceedings, *in press*. To be available in: www.helcom.fi/publications.

HELCOM Habitat WorkingGroup, 2003. "WWF's proposal for designation of the Baltic sea as a particularly sensitive sea area (PSSA)"– input to the meeting of HELCOM HABITAT 4/2003, Document 3.1/5. pp 1–7.

Henricson, C. 2008. Vattenvegetationen i Danskogfladan. En bedömning av muddringsarbetens inverkan på fladan och på Natura 2000-områdets naturvärden. 27 pp. (The aquatic vegetation of Danskogfladan. An assessment of the impact of dredgings on the flad and on the nature values of the Natura 2000 reserve; in Swedish).

Henricson, C., Sandberg-Kilpi, E. & Munsterhjelm, R. 2006. Experimental studies on the impact of turbulence, turbidity and sedimentation on Chara tomentosa L. Cryptogamie, Algologie 27 (4): 419–434.

Holt T.J., Rees E.I., Hawkins S.J., Seed R. 1998. Biogenic Reefs (volume IX). An overview of dynamic and sensitivity characteristics for conservation management of marine SACs. Scottish Association for Marine Science (UK Marine SACs Project). 170 pages.

Hosmer, D.W. & Lemeshow, S. 2000. Applied logistic regression. Wiley, New York.

Howarth R, Chan F, Conley D, Garnier J, Doney SC, Marino R, Billen G. 2011. Coupled biogeochemical cycles: eutrophication and hypoxia in temperate estuaries and coastal marine ecosystems. Frontiers in Ecology and the Environment 9: 18 – 26

http://biodiversity.eionet.europa.eu

http://cdr.eionet.europa.eu/

http://eunis.eea.europa.eu/habitats-names.jsp

http://linnaeus.nrm.se/flora

http://swenviro.naturvardsverket.se/dokument /epi/basinventering.basinvent.htm

http://www.helcom.fi/ (viewed in January 2010).

http://www.helcom.fi/environment2/biodiv/endangered/Biotopes/en_GB/Sandbanks/

http://www.naturvardsverket.se/sv/Arbete-med-naturvard/Skydd-och-skotsel-av-vardefull-natur/Natura-2000/Vagledning/Art--och-naturtypsvisa-vagledningar-for-Natura-2000-/

http://www2.dmu.dk/1_viden/2_Publikationer/3_fagrapporter/rapporter/FR535.PDF

Isaksson, I. & Pihl, L. 1992 Structural changes in benthic macrovegetation and associated epibenthic faunal communities. Netherlands Journal of Sea Research 30: 131–140.

Jansson B.O., Aneer G., Nellbring S., 1985. Spatial and temporal distribution of the demersal fish fauna in a Baltic archipelago as estimated by scuba census. Mar. Ecol. Prog. Ser. 23, 31–43.

Jones S. (2001) Submerged sand banks and reefs inventory. In: Application of Natura 2000 in the marine environment. Workshop at the International Academy for Nature Conservation (INA) on the Isle of Vilm (Germany from 27 June to 1 July 2001. Annex 10, p. 69–71

Kautsky, H. 1989. Quantitative distribution of plant and animal communities of the phytobenthic zone in the Baltic Sea. Contributions from the Askoe Laboratory, University of Stockholm, Sweden. No. 35, 80 pp.

Kautsky, H. 1991. Influence of eutrpohication on the distribution of phytobenthic plant and animal communities. Int. Revue ges. Hydrobiol. 76:423–432.

Kautsky, L. and Kautsky, N. 2000: The Baltic Sea, including Bothnian Sea and Bothnian Bay. In: Sheppard, C. (ed.). Chapter 8: Seas at the millennium, an environmental evaluation. Elsevier, pp. 1–14.

Kemp, W.M., W.R. Boynton, J.E. Adolf, D.F. Boesch, W.C. Boicourt, G. Brush, J.C. Cornwell, T.R. Fisher, P.M. Glibert, J.D. Hagy, L.W. Harding, E.D. Houde, D.G. Kimmel, W.D. Miller, R.I.E. Newell, M.R. Roman, E.M. Smith, and J.C. Stevenson. 2005. Eutrophication of Chesapeake Bay: historical trends and ecological interactions. Marine Ecology Progress Series 303: 1–29.

Kiirikki, M. 1996: Mechanisms affecting macroalgal zonation in the northern Baltic Sea. Eur. J. Phycol. 31:225–232.

Krause-Jensen, D., Carstensen, J. & Dahl, K. 2007a. Total and opportunistic algal cover in relation to environmental variables. – Marine Pollution Bulletin 55:114–125.

Krause-Jensen, D., Carstensen, J., Dahl, K., Bäck, S. & Neuvonen, S. 2009. Testing relationships between macroalgal cover and Secchi depth in the Baltic Sea. Ecological Indicators 9:1284–1287.

Krause-Jensen, D., Middelboe, A.L., Carstensen, J. & Dahl, K. 2007b. Spatial patterns of macroalgal abundance in relation to eutrophication. – Marine Biology 152(1):25–36.

Krause-Jensen, D., Pedersen, M.F., Jensen, C. 2003. Regulation of eelgrass (*Zostera marina*) cover along depth gradients in Danish coastal waters. Estuaries 26:866–877

Krause-Jensen, D., Sagert, S., Schubert, H. & Boström, C. 2008. Empirical relationships linking distribution and abundance of marine vegetation to eutrophication. – Ecological Indicators 8:515–529.

Langangen A (1974): Ecology and distribution of Norweigan charophytes. Norweigan Journal of Botany 21: 31–52.

Lappalainen, A. and Urho, L. 2006. Young-of-the-year fish species composition in small coastal bays in the northern Baltic Sea, surveyed with beach seine and small underwater detonations. Boreal Environment Research 11: 431–440.

Lappalainen, A., Westerbom, M. & Heikinheimo, O. 2005. Roach (*Rutilus rutilus*) as an important predator on blue mussel (*Mytilus edulis*) populations in a brackish water environment, the northern Baltic Sea. – Marine Biology 147:323–330.

Leinikki, J. and Oulasvirta, P. 1995. Perämeren kansallispuiston vedenalainen luonto. Metsähallituksen luonnonsuojelujulkaisuja, Sarja A No 49. 86 pp. (The underwater nature of the Bothnian Bay National Park; In Finnish).

Lundsteen S, Dahl K, Tendal O.S. (2008) Biodiversity on boulder reefs in central Kattegat. BALANCE Interim Report No. 15.

Magorrian, B.H. 1996. The impact of commercial trawling on the benthos of Strangford Lough, Northern Ireland. PhD Thesis, Queens University of Belfast, 218 pp.

Martin G., Möller T., Kotta J., Daunys D., Jermakovs V., Bucas M., Siaulys A., Saskov A., Aigars J. (in prep.). Benthic marine habitats in the eastern Baltic Sea. 121 p.

Mills, K.E. & Fonseca, M.S. 2003. Mortality and productivity of ***eelgrass*** Zostera marina under conditions of experimental burial with two sediment types. Mar. Ecol. Prog. Ser. 255:127–134.

Møhlenberg, F. & E.K. Rasmussen, in press. Size structured model of mussel (Mytilus edulis L.) growth. Ecol. Model.

Muller-Karulis B., Jermkovs V., Aigars J. (2007). The modeling of Furcellaria lumbricalis habitats along the Latvian coast. Balance Report 23, 25 pages.

Munsterhjelm R. 1997: Aquatic macrophyte vegetation of flads and gloes. Acta Bot. Fennica 157:1–68.

Munsterhjelm, R. 2005. Natural succession and human-induced changes in the soft-bottom macrovegetation of shallow brackish bay on the southern coast of Finland. University of Helsinki, Finland. PhD-thesis. 53 pp.

Naturvårdsverket 2010. Undersökning av Utsjöbankar. Inventering, modellering och naturvärdesbedömning. Rapport 6385. 201 pp.

NCM (Nordic Council Of Ministers) 1998: Operative methods for mapping and monitoring phytobenthic zone biodiversity in the Baltic Sea. – Tema Nord 1998:568. 72 pp.

Nielsen, R., Kristiansen, A., Mathiesen, L., Mathiesen, H., 1995. Distributional index of the benthic macroalgae of the Baltic Sea area. Acta Botanica Fennica 155:1–51.

Nielsen, S. L., Sand-Jensen, K., Borum, J. and Geertz-Hansen, O. 2002a. Depth colonisation of eelgrass (*Zostera marina*) and macroalgae as determined by water transparency in Danish coastal waters. Estuaries 5: 1025–1032.

Nielsen, S. L., Sand-Jensen, K., Borum, J. and Geertz-Hansen, O. 2002b. Phytoplankton, nutrients, and transparency in Danish coastal waters. Estuaries *2*5: 930–937.

Nordic Council of Ministers (Nordiska ministerrådet) (1998): Vegetationstyper i Norden. TemaNord 1998: 510.

Numminen, S. 1999: Fladat ja kluuvijärvet Saaristomerellä. Lounais-Suomen ympäristökeskus, Suomen ympäristö 339. Astro Oy, Turku.

Olenin, S. & Leppäkoski, E. 1999: Non-native animals in the Baltic Sea: alteration of benthic habitats in coastal inlets and lagoons. Hydrobiologia 393:233–243.

Olenin, S. 1994. Lithuanian Stony Bottom Localities: a Link in a Chain of the Baltic Sea Marine Protected Areas. Marine Protected Areas and Sustainable Fisheries. In: N.L.Shackell & J.H.M.

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

Phillips S J, Anderson R P and Schapire R E., 2006. Maximum entropy modeling of species geographic distributions. Ecological Modelling, Vol 190/3–4: 231–259.

Willison (Editors). Proc. 2'nd International Conference on Science and the Management of Protected Areas. Dalhouse University, Halifax, Nova Scotia, Canada: 145–148.

Oliver, G. 2007: Conservation status assessment report – Coastal lagoons (1150) ??

Olsen S (1944) Dansih Charophyta. Konglige Danske Videnskabssellskapet. Biologiska tidskrifter 3.

Orlova, M.I., Telesh, I.V., Berezina, N.A., Antsulevich, A.E., Maximov, A.A. & Litvinchuk, L.F 2006. Effects of nonindigenous species on diversity and community functioning in the eastern Gulf of Finland (Baltic Sea). Helgol Mar Res 60: 98–105

OSPRA 2008. Descriptions of habitats on the OSPAR list of threatened and/or declining species and habitats. Supersedes 2004–7 including amendments agreed by BDC 2006 (BDC 2006 Summary Record (BDC 06/10/1) § 3.32) and BDC 2008 (BDC 2008 Summary Record (BDC 08/12/1) § 4.22). Reference Number: 2008–07, p. 8.

Pawlaczyk, P., Kepel, A., Jaros, R., Dzięciołowski, R., Wylegała, P., Szubert, A., Sidło, P.O. 2004: Natura 2000 shadow list in Poland.

Pearson, T.H. and Rosenberg, R. 1978. Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. Oceanogr. Mar. Biol. Annu. Rev. 16, 229–311.

Pearson, T.H. and Rosenberg, R. 1978. Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. Oceanogr. Mar. Biol. Annu. Rev. 16, 229–311.

Pedersén, M. and P. Snoeijs. 2001. Patterns of macroalgal diversity, community composition and long-term changes along the Swedish west coast. Hydrobiologica 459: 83–102.

Perus, J., Bonsdorff, E., Bäck, S., Lax, H-G., Villnäs, A. & Westberg, V. 2007. Zoobenthos as indicators of ecological status in coastal brackish waters: a comparative study from the Baltic Sea. Ambio 36:250–256.

Pihl, S., Ejrnæs, R., Søgaard, B., Aude, E., Nielsen, K.E., Dahl, K., Laursen, J.S. 2001:

Powilleit, M., Graf, G., Kleine, J., Riethmüller, R., Stockmann, K., Wtzel, M.A. & Koop, J.H.E. 2009. Experiments on the survival of six brackish macro-invertebrates from the Baltic Sea after dredged spoil coverage and its implications for the field. Journal of Marine Systems 75:441–451.

Rassi, P., Alanen, A., Kanerva, T., and Mannerkoski, I. (editors) 2001. Suomen lajien uhanalaisuus 2000. – Ympäristöministeriö & Suomen ympäristökeskus, Helsinki. 432 p. (Evaluation of Threatened Species in Finland 2000; In Finnish).

Raunio, A, Schulman, A. & Kontula, T. (Eds.) 2008. Assessment of threatened habitat types in Finland – Part 2: Habitat type descriptions. The Finnish Environment 8/2008. 572 pp.

Raunio, A., Schulman, A. and Kontula, T. (editors; 2008). SY 8/2008 Suomen luontotyyppien uhanalaisuus – Osa 2, 2 Itämeri ja rannikko.pdf. Suomen ympäristökeskus, Helsinki. Available at: http://www.ymparisto.fi/ (viewed in January 2010). (Threatened habitats in Finland – Part 2, 2 The Baltic Sea and coast; in Finnish)

Rinkineva, L. & Molander, L-L. 1997: Merenkurkun fladat ja kluuvijärvet. Merenkurkun neuvoston julkaisut 3:1–42.

Rönnberg, O., 1981. Traffic effects on rocky-shore algae in the Archipelago Sea, SW Finland. Acta Academiae Aboensis, Series B 41, 1–86.

Roos, C., Rönnberg, O., Berglund, J., Alm, A. 2004 Long-term changes in macroalgal communities along ferry routes in a northern Baltic archipelago. Nordic Journal of Botany 23 (2): 247–259.

Salomonson, A. 2003. Natura 2000 Merenkurkussa – fladojen ja primäärisukkession metsien seuranta. Interreg IIIA ohjelman Kvarken Miljö-projektin raportti. s. 5–16.

Schiewer, U. (ed.) 2008: Ecology of Baltic coastal waters. Berlin, Heidelberg, Springer-Verlag. Ecological studies 197. pp.419.

Schories, D., Pehlke, C & Selig, U. 2009. Depth distributions of *Fucus vesiculosus* L. and Z*ostera marina* L. as classification parameters for implementing the European Water Framework Directive on the German Baltic coast. Ecological indicators 9:670–680.

Schubert, H. & Blindow, I. 2003 (eds.). Charophytes of the Baltic Sea. – The Baltic Marine Biologists publication 19:1–326, 16 pl. A.R.G. Gantner Verlag K.-G., Ruggell.

Schubert H. 2002: (Development of ecological quality standards for submersed macrophytes of coastal lagoons of the German Baltic Sea, German). "Entwicklung von leitbildorientierten

Bewertungsgrundlagen für Übergangsgewässer entsprechend EU-Wasserrahmenrichtlinie.

Oral presentation at the Annual Meeting of the Federal Agency for Coastal Monitoring Programs (BLMP), Güstrow, Germany, May 2002. Del. 20/32.

Schubert, H. and Schories, D. 2008. Macrophytobenthos. In: Feistel R., Nausch G., Wasmund N. (eds). State and evolution of the Baltic Sea, 1952–2005 John Wiley & Sons, Inc., Hoboken, pp. 483–516.

Selig, U., Eggert, A., Schories, D., Schubert, M., Blümel, C. & Schubert, H. 2007a: Ecological classification of macroalgae and angiosperm communities of inner coastal waters in the southern Baltic Sea. Ecological Indicators 7:665–678.

Selig, U., Eggert, A., Schories, D., Schubert, M., Blümel, C. & Schubert, H. 2007a: Ecological classification of macroalgae and angiosperm communities of inner coastal waters in the southern Baltic Sea. Ecological Indicators 7:665–678.

Selig, U., Schubert, M., Eggert, A., Steinhardt, T. Sagert, S. & Schubert, H. 2007b. The influence of sediments on soft bottom vegetation in inner coastal waters of Mecklenburg-Vorpommern (Germany). Estuarine, Coastal and Shelf Science 71:241–249.

Short F.T. & Burdick D.M. 1996. Quantifying eelgrass habitat loss in relation to housing development and nitrogen loading in Waquoit Bay, Massachusetts. Estuaries 19:730–739

Short, F. & Neckles, H. 1999. The effects of global climate change on seagrasses. Aquatic Botany 63: 169–196.

Sjöberg B (ed.) (1992) Ocean and coasts, National Atlas of Sweden. Sveriges Nationalatlas, 128 pp.

Skov, H. & Dinesen, G. (Eds.) 2007. Mapping and modelling of marine habitats in the Baltic

Sea region. BALANCE Interim Report No. 10. The Danish Forest and Nature Agency. 183 pp.

Sohlman A (ed.) (2008): Arter och naturtyper i Habitatdirektivet – tillståndet i Sverige 2007. ArtDatabanken, SLU, Uppsala.

Solimini, A.G., Cardosa, A.C. & Heiskanen, A-S. (Eds.) 2006. Indicators and methods for the ecological status assessment under the Water Framework Directive. Linkages between chemical and biological quality of surface waters. European Comission – Joint Research Center. 252 pp.

Spehn EM, Joshi J, Diemer M, Ko¨rner C (2000) Above-ground resource use increases with plant species richness in experimental grassland ecosystems. Funct Ecol 14:326–337

Steneck, R.S., Graham, M.H., Bourque, B.J., Corbett, D., Erlandson, J.M., Estes, J.A. & Tegner, M.J., 2002. Kelp forest ecosystems: biodiversity, stability, resilience and future. – Environmental Conservation 29:436–459.

Swedish Environmental Protection Agency (Naturvårdsverket) (2000): Kust- och skärgårdsområden i Sverige. Report 5116.

Swedish Environmental Protection Agency (Naturvårdsverket) (2003): Natura 2000 i Sverige – handbok med allmänna råd. Handbook 2003:9.

Swedish Environmental Protection Agency (Naturvårdsverket) (2004a): Handbok för vegetationsklädda bottnar, ostkust. Version 1: 2004-04-27.

Swedish Environmental Protection Agency (Naturvårdsverket) (2004b): Uppföljning av Natura 2000 i Sverige. Report 5434.

Swedish Environmental Protection Agency (Naturvårdsverket) (2006): Inventering av marina naturtyper på utsjöbankar. Report 5576.

Sydänoja, A. 2008: Saaristomeren ja Selkämeren fladat. Lounais-Suomen ympäristökeskuksen raportti 1/2008. Edita Prima Oy, Helsinki. In Finnish.

Szymczak, E. & Piekarek-Jankowska, H. 2007: The transport and distribution of the river load from the Reda River into the Puck Lagoon (Poland). Oceanological and Hydrobiological Studies Vol. 36, no. 1, pp. 103–124.

Timm, T., Kumari, M., Kübar, K., Sohar, K. & Traunspurger, W. 2007. Meiobenthos of some Estonian coastal lakes. Proc. Estonian Acad. Sci. Biol. Ecol. 56: 179–195.

Tolstoy A, Österlund K (2003) Alger vid Sveriges Östersjökust: en fotoflora. ArtDatabanken, SLU, 282 pp.

Tyler-Walters, H. and Hiscock, K. (2005). Impact of human activities on benthic biotopes and species. Report to Department for Environment, Food and Rural Affairs from the Marine Life Information Network (MarLIN). Plymouth: Marine Biological Association of the UK. [Contract no. CDEP 84/5/244]

Valiela, I., McClelland, J., Hauxwell, J., Behr, P. J., Hersh, D. & Foreman, K. 1997 Macroalgal blooms in shallow estuaries: Controls and ecophysiological and ecosystem consequences. Limnology and Oceanography 42, 1105–1118.

Viitasalo-Frösen, S., Laine, A. & Lehtiniemi, M. 2009. Habitat modification by motile surface stirrers vs.semi-motile burrowers: potential for a positive feedback in a eutrophied ecosystem. *Marine Ecology Progress Series* 376: 21–32.

Virtasalo, J. J., Kohonen, T., Vuorinen, I. & Huttula, T., 2005: Sea bottom anoxia in the Archipelago Sea, northern Baltic Sea – implications for phosphorus remineralization at the sediment surface. Marine Geology 224 (1–4), 103–122.

Vuorinen, I., Antsulevich, A.E., & Maximovich, N.V. 2002 Spatial distribution and growth of the common mussel Mytilus edulis L. in the Archipelago Sea, Northern Baltic. Boreal Environment Research 7:41–52.

Walker, D.I. & Kendrick, G.A. 1998. Threats to macroalgal diversity: Marine habitat destruction and fragmentation, pollution and introduced species. Botanica Marina 41:105–112.

Walve, J., 2008 (pers. comm.): Dept. of Systems Ecology at Stockholm University and Svealands Kustvattenvårdsförbund, http://www.kustdata.su.se/.

Wells, E., Wilkinson, M., Wood, P.& Scanlan, C. 2007. The use of macroalgal species richness and composition on intertidal rocky seashores in the assessment of ecological quality under the European Water Framework Directive. Marine Pollution Bulletin 55 : 151–161.

Wennberg, S., Nöjd, A., Lindblad, C. 2008: Mapping of marine Natura 2000 habitats in GIS – a case study from the Stockholm archipelago and Archipelago Sea. BALANCE interrim report No. 30. Available at www.balance-eu.org/publications.

Westerbom, M. & Jattu, S. 2006. Effects of wave exposure on the sublittoral distribution of blue mussels (Mytilus edulis) in a heterogeneous archipelago – Marine Ecology Progress Series 306: 191–200.

Westerbom, M. 2006. Population dynamics of blue mussels in a variable environment at the edge of their range. University of Helsinki. PhD-thesis. 62 pp.

Westerbom, M., Kilpi, M. and Mustonen, O. 2002. Blue mussels, Mytilus edulis, at the edge of the range: population structure, growth and biomass along a salinity gradient in the north-eastern Baltic Sea. Mar. Biol. 140: 991–999.

Westerbom, M., Mustonen, O. & Kilpi, M. 2008. Distribution of a marginal population of *Mytilus edulis*: responses to biotic and abiotic processes at different spatial scales. – Mar. Biol. 153:1153–1164.

Wijkmark N., Isaeus M. (2010). Wave exposure calculations for the Baltic Sea. Report for EUS SeaMap project, 37 pages.

Zettler ML, Röhner M, Frankowski J, Bochert R, Glockzin M, Glockzin I (2006) Benthologische arbeiten zur ökologischen bewertung von windenergie-anlagen-eignungsgebieten in der Ostsee: Zusammenfassung der Untersuchungen zum makrozoobenthos in potentiellen NATURA 2000-Gebieten. Institut für Ostseeforschung Warnemünde, 222 pp.

Zettler, M.L., Gosselck, F. (2006). Benthic assessment of marine areas of particular ecological importance within the German Baltic Sea EEZ. In: von Nordheim H., Boedeker D. and Krause J.C. (eds.) Progress in marine conservation in Europe. Natura 2000 sites in German offshore waters. Springer Berlinn Heidelberg New York, 141–156.

8. Appendix

8.1 Conceptual models (Figures)

Figure 54. Conceptual model of reefs describing the linkages between eutrophication effects and suggested indicators on three habitat change levels; habitat alteration, habitat fragmentation and habitat loss. Thicker arrows indicate the most important links between habitat effects and indicators

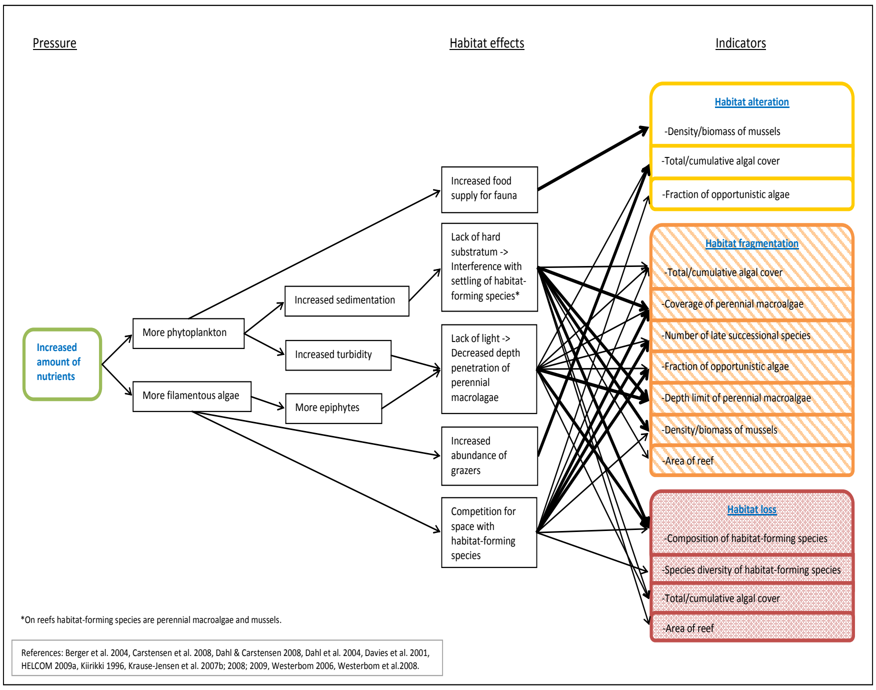


Figure 55. Conceptual model of reefs describing the linkages between the effects of physical disturbance and suggested indicators. Thicker arrows indicate the most important links between habitat effect and indicators

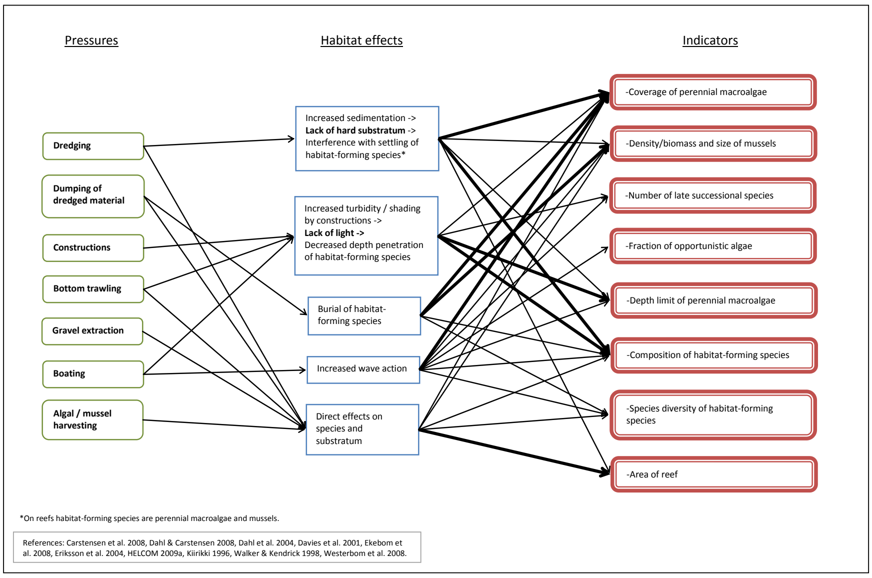


Figure 56. Conceptual model of reefs describing the linkages between the effects of climate change and suggested indicators. Thicker arrows indicate the most important links between habitat effect and indicators

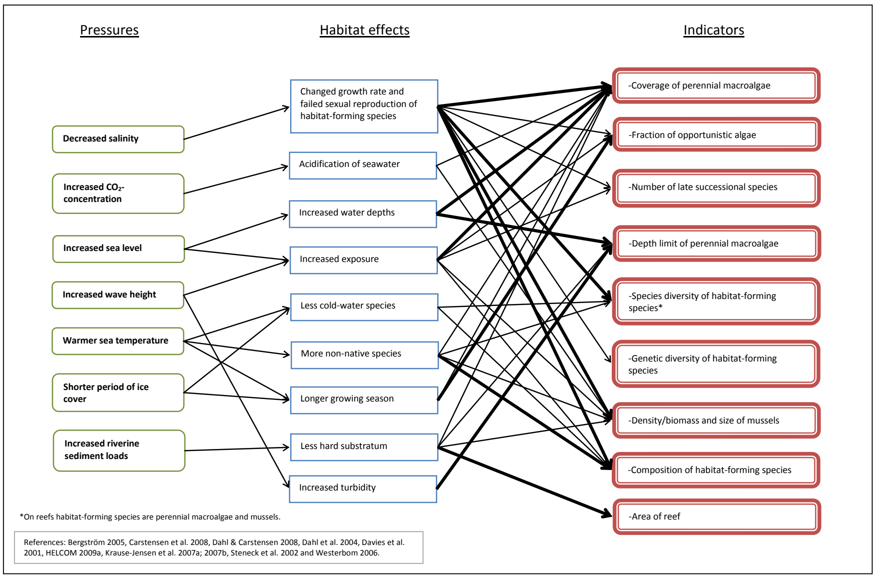


Figure 57. Conceptual model of sandbanks describing the linkages between the eutrophication effects and suggested indicators on three habitat change levels; habitat alteration, habitat fragmentation and habitat loss. Thicker arrows indicate the most important links between habitat effect and indicators

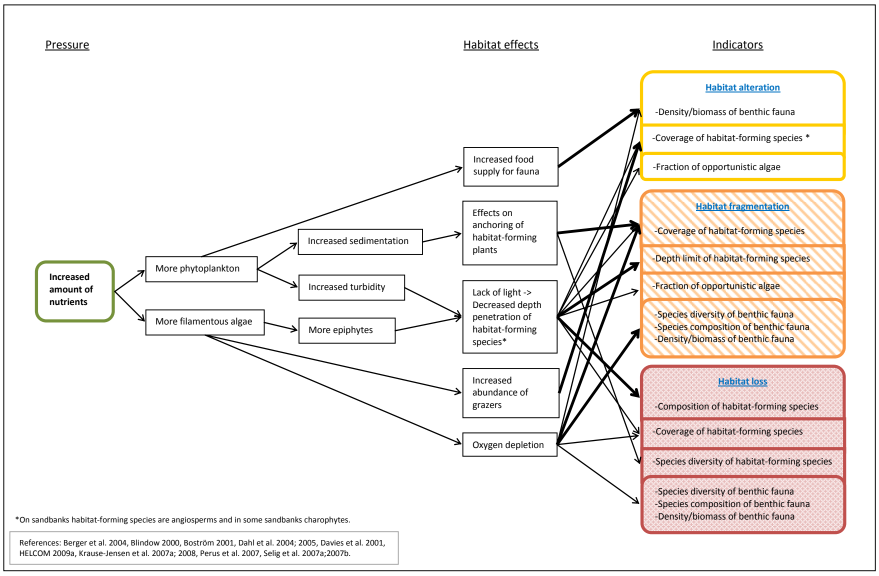


Figure 58. Conceptual model of sandbanks describing the linkages between the effects of physical disturbance and suggested indicators. Thicker arrows indicate the most important links between habitat effect and indicators

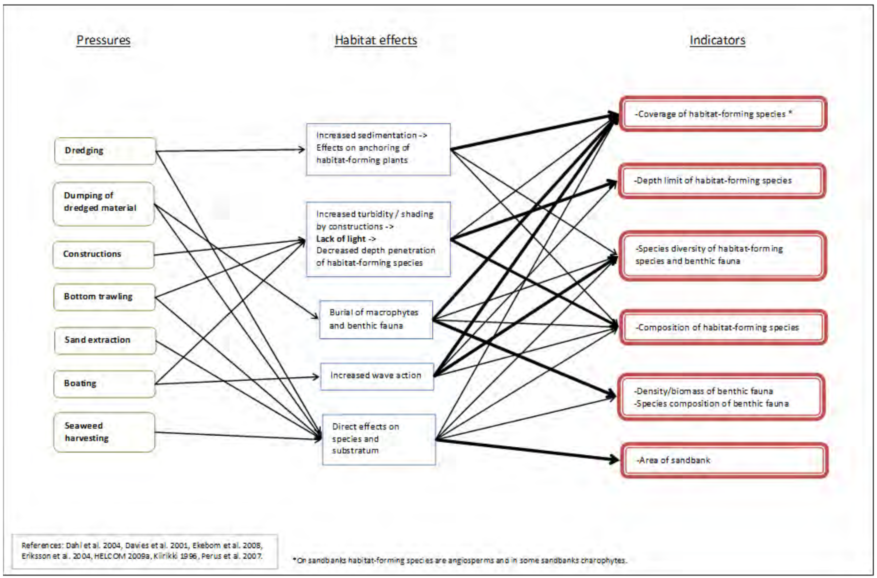


Figure 59. Conceptual model of sandbanks describing the linkages between the effects of climate change and suggested indicators. Thicker arrows indicate the most important links between habitat effect and indicators

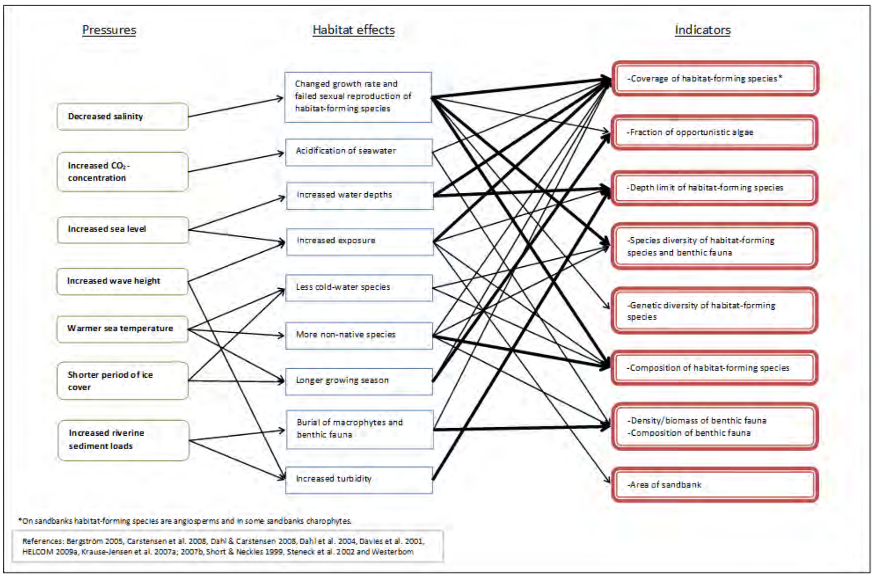


Figure 60. Conceptual model of lagoons describing the linkages between the effects of eutrophication and suggested indicators on three habitat change levels; habitat alteration, habitat fragmentation and habitat loss. Thicker arrows indicate the most important links between habitat effect and indicators

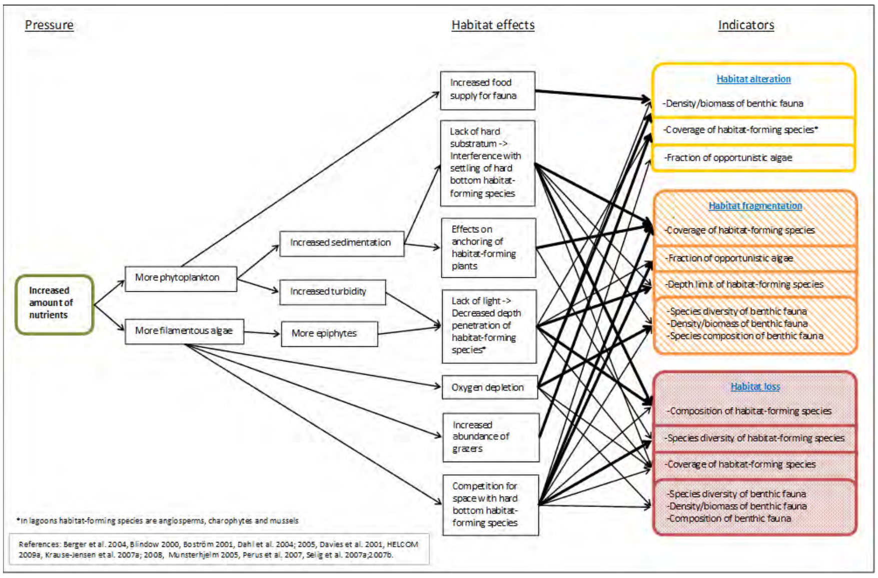


Figure 61. Conceptual model of lagoons describing the linkages between the effects of physical disturbance and suggested. Thicker arrows indicate the most important links between habitat effect and indicators

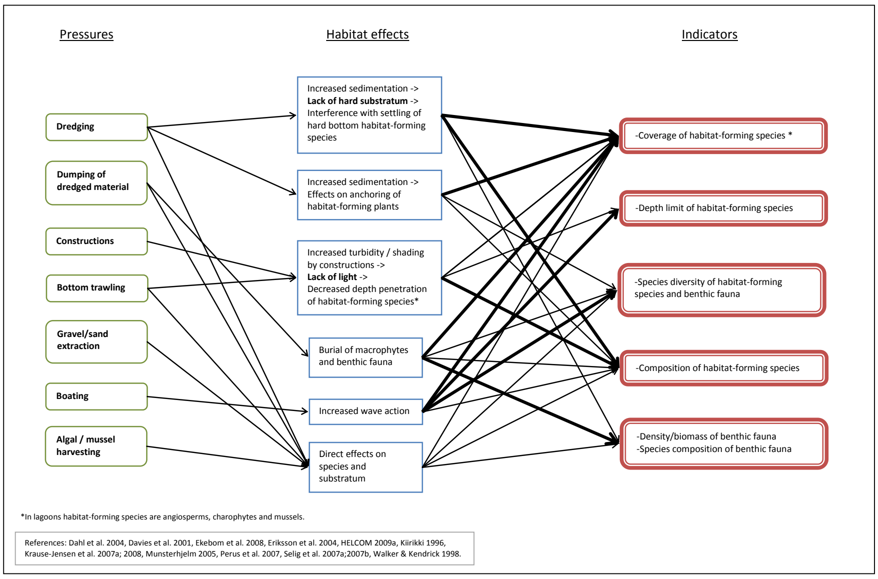
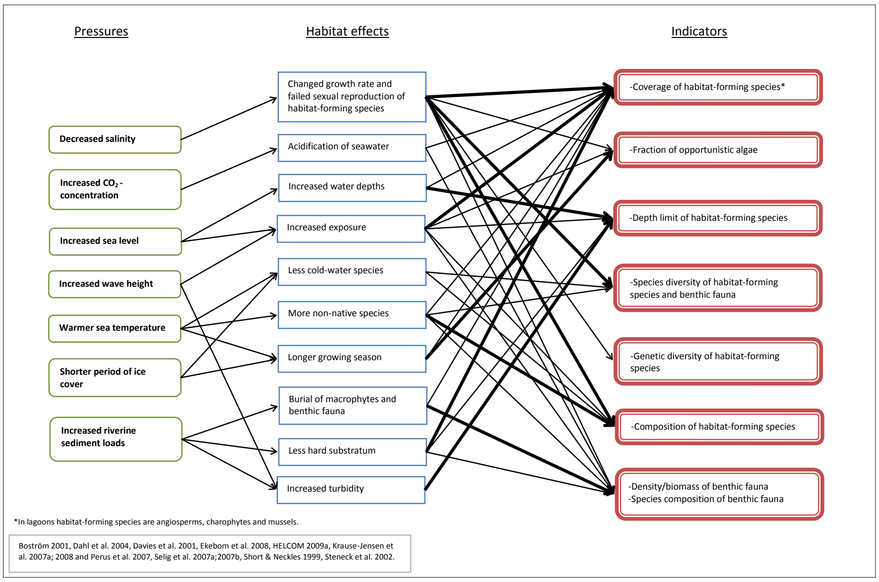


Figure 62. Conceptual model of lagoons describing the linkages between the effects of climate change and suggested indicators. Thicker arrows indicate the most important links between habitat effect and indicators



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MOPODECO

MOPODECO has aimed to fill the gaps in harmonisation of the definition of the EU Habitat Directive Annex I habitats and the view on the main pressures and threats to these habitats between the Nordic countries and countries surrounding the Baltic Sea. The project has also aimed to develop standards for describing the functional characteristics of the Annex I habitats, including standards for application of modelling tools for quantifying the coverage of habitat-forming species. In MOPODECO, the harmonised definitions, pressure evaluation matrices, functional descriptors and habitat model application standards are integrated into proposals for unified indicators of favourable conservation status for Annex I habitats in Nordic waters. Common standards for assessing the favourable conservation status of these habitats are suggested by applying these indicators in case studies from the northern and southern parts of the Baltic Sea.

TemaNord 2012:532
ISBN 978-92-893-2368-0
http://dx.doi.org/10.6027/TN2012-532