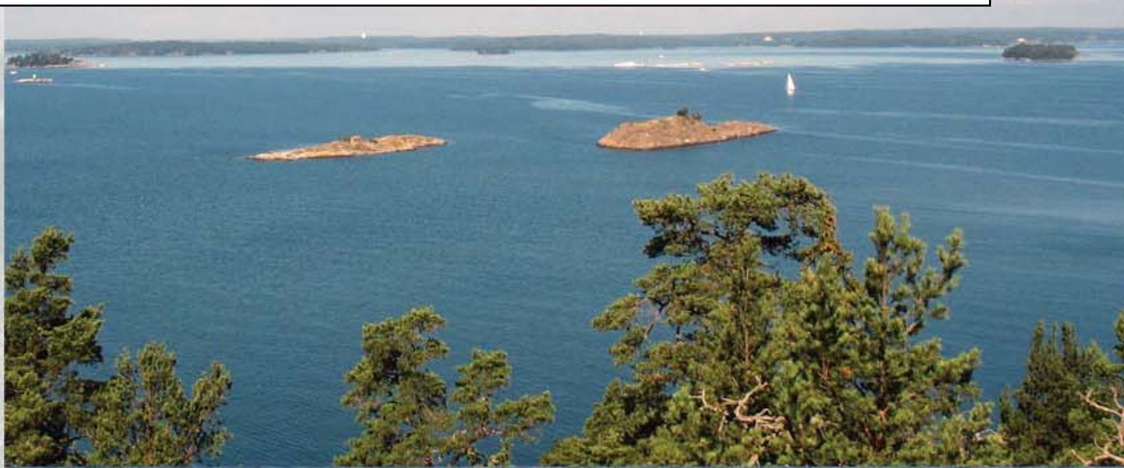
BALANCE Interim Report No. 27

Mapping and modelling of marine habitat in the Baltic Sea

DRAFT



Baltic Sea Region
INTERREG III B

BALANCE

Title	BALANCE Interim Report No.
Mapping and modelling of marine habitats in the Baltic Sea region	27

Authors

Grete E. Dinesen (ed.), The Danish Forest and Nature Agency

In association with:

Jesper H. Andersen, DHI Water, Environment & Health, Denmark
Johnny B. Reker, The Danish Forest and Nature Agency

With report contribution from (in alphabetic order):

Trine Bekkby, Norwegian Institute for Water Research
Ulf Bergström, The National Board of Fisheries, Sweden
Martynas Bučas, Klaipeda University, Lithuania
Ida Carlén, AquaBiota Water Research, Sweden
Karsten Dahl, The National Environmental Institute, Denmark
Darius Daunys, Klaipeda University, Lithuania
Jørgen Hansen, The National Environmental Institute
Kristjan Herkül, University of Tartu, Estonia
Martin Isæus, AquaBiota Water Research, Sweden
Gerd Kraus, The Danish Institute of Fisheries Research
Jonne Kotta, University of Tartu, Estonia
Jørgen O. Leth, The Geological Survey of Greenland and Denmark
Cecilia Lindblad, The Swedish Environmental Protection Agency
Georg Martin, University of Tartu, Estonia
Frithjof Moy, Norwegian Institute for Water Research
Hans Nilsson, Norwegian Institute for Water Research
Anna Nójd, Finnish Environment Institute
Kjell Magnus Norderhaug, Norwegian Institute for Water Research
Helen Orav-Kotta, University of Tartu, Estonia
Are Pedersen, Norwegian Institute for Water Research
Alfred Sandstróm, The National Board of Fisheries, Sweden
Mart Simm, University of Tartu, Estonia
Matias Sköld, The National Board of Fisheries, Sweden
Claus Sparrevohn, The Danish Institute for Fisheries Research
Göran Sundblad, The National Board of Fisheries, Sweden
Sandra Wennberg, Metria, Sweden
Petras Zemblys, Klaipeda University, Lithuania

Date

May 2008

Approved by

Revision	Description		Checked	Approved	Date
	Front page illustration:				
1	Final report				
0	Draft report				

Key words	Classification
BALANCE; Baltic Sea; Kattegat; Skagerrak, marine	☒ Open

habitats; predictive modelling; GIS mapping;	☐ Internal ☐ Proprietary

Distribution	No of copies
BALANCE Secretariat	** + pdf
BALANCE partnership	** + pdf
BSR INTERREG IIB Joint Secretariat	1
Archive	1

CONTENTS

0 EXECUTIVE SUMMARY

This report summarises the methodologies applied for mapping and modelling of selected broad-scale marine habitats in four pilot areas in the Kattegat-Skagerrak area and the Baltic Sea. The report also summarises the potential applications and limitations of applying the different methods of mapping and predicting the distribution of marine habitats within the Baltic Sea region, and it defines habitats and methodologies not covered by BALANCE, which need attention in future studies.

The plethora of habitat mapping and modelling techniques used in BALANCE constitutes the foundation for the key deliverables from the project in terms of support to an improved marine spatial planning and hence sustainable development of the Baltic Sea; guidelines on zoning and management plans.

0.1 Background

BALANCE is a BSR INTERREG IIIB (Neighbourhood Programme) project, which started in 2005 by a consortium consisting of a total of 27 governmental agencies, research institutes, universities, regional authorities and NGOs in 10 countries surrounding the Baltic Sea (including Norway and U.S.A). BALANCE builds upon experiences on habitat mapping and modelling made in Canada (Roff & Taylor 2000, 2003) and initiatives from the UK e.g. the Irish Sea Pilot Project (Vincent et. al 2004) and the UK-SeaMap (Connor et. al. 2007a). The recent interest and effort to produce marine habitat and landscape maps has been spawned by increasing national and international management requirements to provide a trans-national perspective and overview of the marine landscapes present, their extent, and distribution within the Baltic Sea region and other marine regions of the world. The most important management applications of marine habitat and landscape maps in the Baltic Sea region are summarised below.

- **Implementation of the EU Directives** - support implementation of EU Habitats Directive, EU Water Framework Directive, and the proposed EU Marine Strategy Directive. These all, directly or indirectly, require an ecosystem-based approach to the management of the marine environment.
- **European Marine Policy –** aims at a unified policy for the management of all European seas with an emphasis on the ecosystem approach, both as a founding principle for the whole policy and as a key tool in marine spatial planning and in the Common Fisheries Policy.
- **HELCOM Baltic Sea Action Plan** – it aims to apply the ecosystem-based approach to the management of the Baltic Sea, which acts as a pilot project for the EU Marine Strategy Directive.
- **OSPAR/HELCOM working groups on threatened and/or declining species and habitats** – defines species and habitats which are currently threatened or declining in Baltic Sea and thus assists in the implementation of the EU nature Directives
- **Protecting the marine environment** - The maps will present end users with a better understanding of the extent and distribution of the natural values of the Baltic Sea. The maps will feed directly into a BALANCE assessment of the representativity of the network of marine protected areas in the Baltic Sea region.

- **Sea use planning** - The availability of a broad scale ecological map for the Baltic Sea region can provide sea use planners with an opportunity to incorporate an ecosystem-based approach when making planning decisions on a regional scale taking a layer with the natural values into account, and thus help in an assessment of the potential impact of human activities.
- **Strategic planning** - Marine landscape maps provide a baseline study of the complexity within a region providing planners and policy makers with a tool to integrate knowledge of marine ecosystems more efficiently with development strategies.
- **Maritime safety** - Marine landscape maps may be used in regard to maritime safety issues as it provides an ecological input for a region showing the amount and distribution of specific natural values and thus provide a basis for sensitivity mapping of areas considered as emergency harbours in case of shipping accidents.

BALANCE aimed to develop a broad scale and ecosystem-based approach to the mapping of the unique environment and natural landscapes of the Baltic Sea, Kattegat and parts of Skagerrak. This resulted in the definition of coastal physiographic features and seabed features (which includes topographic features and ecological relevant benthic landscapes identified on salinity, sediments, and photic depth) as well as four examples of water column marine landscapes for the Baltic Sea and Kattegat. Thus, BALANCE gratefully builds upon the efforts mentioned above in order to learn from previous experiences as well as to harmonise methods. This report should be seen as the first step towards identifying and mapping the marine landscapes in the Baltic Sea and Kattegat building upon trans-national and cross-sectoral cooperation.

BALANCE has focused on identifying sea bottom marine landscapes (both benthic features and topographic features marine landscapes, as it will be discussed in later chapters) and physiographic marine landscapes in the Baltic Sea region. The identification of pelagic marine landscapes should be developed as a 3D model, which is beyond the scope of BALANCE. In order to avoid confusion on terminology the term “marine landscapes” has been applied to promote the use of a unified terminology in Europe rather than using the term “seascape” as this in the UK describes “the view over coastal feature e.g. the White Cliffs of Dover”. Any reference to the work done by BALANCE should therefore use “marine landscapes”[1].

Lastly, a word of caution - the marine landscape maps are not better than the data used to develop them. In some regions, especially offshore, raw data points are scarce and far between and modelled data (using a grid of 7 km) has been applied. Hence, further refinements need to focus on validation through obtaining new data, continue improving the maps and lastly, providing a confidence rating of the maps. Further, BALANCE focused on the establishment of landscape maps covering mainly benthic habitats, and thus future landscape maps covering all habitats need to be developed to offer end users a fully comprehensive mapping system for the Baltic Sea. In relation to sectoral development planning it is important to stress the need to transfer the habitat and landscape maps to sensitivity maps displaying the degree of resilience and vulnerability of the habitats and landscapes in relation to potential perturbations associated with each sectoral use of the sea.

[1] More information about BALANCE can be found at http://www.balance-eu.org and about MESH on http://www.searchmesh.org. For information of the BSR INTERREG IIIB Neighbourhood Programme please look at http://www.bsrinterreg.net.

0.2 Aims and approach

The aim was to use available and new geological, oceanographic, and chemical data as well as biological information to model and map the distribution of broad-scale marine habitats and landscapes within the four pilot areas in the Baltic Sea region: 1) the Kattegat-Skagerrak, 2) the Bornholm Deep, 3) the Åland Islands, and 4) the Lithuanian coastal zone. The work was carried out through cross-sectoral and trans-national cooperation, thus providing ecological relevant maps for national and trans-national management of the Baltic marine environment. The process undertaken included:

- Identify environmental data needed for mapping of marine habitats of the seabed and in the water column.
- Characterise broad-scale features of the seabed.
- Identify and access available data spanning the entire Baltic Sea region.
- Identify and sign suitable data sharing agreements within the partnership.
- Classify data into uniform categories and process them into an agreed GIS format.
- Identify ecological relevant categories for each environmental factor e.g. photic or non-photic depth and create the data layer in GIS.
- Analyse the data in order to produce the classification of the seabed and coastal zone.
- Begin the validation process.
- Begin the confidence assessment exemplified for specific sub-regions.
- Present the data layers and maps and show examples of potential application.

A focused effort was put on classification of broad-scale coastal and benthic landscapes in the Baltic Sea. The classification involved three different characterisations using GIS-based multi-criteria evaluation techniques. Firstly, the topographic and bed-form features were identified showing the topographic layout of the seabed. Secondly, the physiographic features were identified showing the layout of the coastal area (e.g. archipelagos), and lastly, ecological relevant entities of the seabed were identified showing their broad scale distribution and extent within the Baltic Sea region (e.g. non-photic mud at 30 psu). Another key activity of WP 2 has been predictive modelling of the potential distribution of habitat-building species and essential fish habitat in the four pilot areas. The modelling activities applied data-driven statistical models (Generalised Additive Models, GAM) to regional empirical data and GIS coverages of selected physical parameters.

Within the framework of BALANCE it was not possible to integrate information and models of key processes and ecological significance into the characterisation of Baltic marine landscapes. Therefore, the marine landscape classifications should be seen as qualitative, and future studies will be needed to resolve the linkages between landscapes and ecological functioning and between landscapes and resilience and sensitivity to the growing human activities in the region. Such studies will also be needed to transfer the static maps into dynamic maps incorporating potential structural changes over the short- and medium term.

0.3 Potential application - a stepwise approach to mapping of marine habitats in the Baltic Sea region

1. GIS analyses and mapping of the NATURA 2000 Annex 1 habitats that equal landscapes, as well as analyses and mapping of other marine landscapes.

2. Detection of remaining NATURA 2000 Annex 1 habitats by acoustic and other methods.

3. Sampling of acoustic data of fine grain (multibeam and back scatter, resolution of 1-10 cm) and validation using samples, UW video/photographs, and/or diving observations.

4. Sampling and collation of additional variables, essential for NATURA 2000 Annex 1 habitats. For the OSPAR, HELCOM and essential top predator habitats variables driving ecological significance will be necessary.

5. Predictive modelling of habitats, particularly on habitat-building species of structural importance and species of high ecological significance. Validation of models.

6. Integration of habitat model results and existing marine landscapes in GIS to create a habitat classification system for the Baltic Sea region, with classes of discrete boundaries.

7. Predictive modelling of spatial and temporal variation of each classified habitat.

Application for planning of MPA network, incl. of the NATURA 2000 network:

8. Apply in GIS to create areas of appropriate size and numbers, as well as blue corridors.

9. Use the above data to evaluate the best possible solutions of a marine protected areas network of habitats and corridors, which maintain ecological functioning.

Application for management and evaluation of ecological status of MPA network, incl. of the NATURA 2000 network:

10. Assess the resilience and sensitivity of each classified habitat in relation to major sea uses.

1 INTRODUCTION

Over the past century, the marine environment has deteriorated, and more rapidly so in the last few decades. The growing population of man has increased the global demand for marine resources for construction work, transport, energy, and food. This has placed marine areas under severe pressure and lead to degradation of natural ecosystems, particularly in coastal areas on the continental shelves (e.g. Rosenberg 2003, Tilman 2000). While some pressures can be diminished by development and use of advanced technologies (e.g. drain of sewage, dumping of waste, and disturbance from shipping) other pressures are related to demand of area (e.g. recreation, land reclamation, extraction of raw material, oil and gas exploitation, shipping routes, fishery, natural and cultural conservation). As on land, space has now become a limited source at sea. This asks for careful planning and management of the use of natural ecosystems to ensure development sustainable for both man and nature (e.g. Margules & Pressey 2000, Peterson & Kneib 2003).

Marine spatial planning is a powerful tool for management at sea and the finding of long term solutions as more activities demands more space and the extent of pressures expand. Experiences and best practices from spatial planning and management on land should be included to the extent possible. The first step is mapping in space, including of the geological history and resources, the nature values and ecological processes behind, and the anthropogenic remains and cultural inheritance. The second step is classification of the significance of mapped areas to ecosystem processes and services. The third step involves mapping of pressures and uses and the ecosystem sensitivity to these. Based on this, first generation marine spatial planning can be developed to provide a common strategic perspective for development and use of the sea. To become truly successful marine spatial planning requires a fourth step, unique to the marine environment, which is mapping in time. Thus, features and processes that may change within less than 10 or 100 years need to be dynamically encompassed in the mapping processes, while features that change at a much slower pace (over thousand to millions of years) may be dealt with in a rather static way.

This report is concerned primarily with the first step, while touching on the second and fourth steps. Compared to on land, spatial mapping in the sea is far more expensive and rely heavily on remote sensing. In addition, ecological functioning and processes differs fundamentally between land and sea environments (Martin et al. 2005, Ockelmann & Dinesen submitted, Sale et al. 2006). Thus new methods and approaches must be developed to capture the information necessary for marine spatial planning and management.

Continuous maps of habitats and ecological communities are needed as a basic tool. Detailed knowledge on the spatial extent of essential habitats for algae, plants, invertebrates, fishes, and mammals in the Baltic Sea region is however sparse, as is knowledge on the effects of habitat availability and quality on population size, structure, and recruitment.

Maintenance and restoration of nature and ecosystem functioning require efficient management of the use and exploitation, particularly in coastal areas such as the major part of the Baltic Sea region. Recognising a holistic approach is needed, multiple initiatives and legislation is now being interwoven at the international level. The BALANCE project aims through this report to deliver the first holistic trans-national approach to identifying and mapping seabed features of the Baltic Sea region – an approach with the potential to be further developed into a tool for implementing an ecosystem-based and sustainable approach to sea use planning and management of the marine environment in the Baltic Sea region. The report is based on international and cross-sectoral cooperation with participants from the countries surrounding the Baltic Sea including Norway.

1.1 Requirements for habitat data in marine spatial planning

The variety of current needs for broad-scale information to promote the implementation of an ecosystem-based approach to management at sea is made tangible through various initiatives and legal requirements, such as:

- Implementation of EU directives, such as the EU Habitats Directive, the EU Water Framework Directive, and the proposed EU Marine Strategy Directive, which all, directly or indirectly, requires a broad scale approach to the management of the marine environment.
- The need for identifying marine protected areas (MPAs) and assessing the ecological coherence and representativity of existing MPA networks e.g. the NATURA 2000 and Baltic Sea Protected Areas networks in the Baltic Sea region.
- Providing a trans-national solution to initiatives such as the Baltic Sea Action Plan undertaken by the Baltic Sea States under the Helsinki Convention (HELCOM).
- Delivering ecological relevant information for promoting sea use planning.

Meeting these needs creates several political and technical challenges for the countries sharing the Baltic Sea region, such as:

- Overcoming the historical distrust that shaped the region for half a century.
- Enabling access to existing national environmental data.
- Requisition of e.g. biological information for offshore areas where little biological information is available if it exists at all.
- Overcoming differences in methodology for collecting, storing, and interpreting of environmental data.
- Providing relevant trans-national and cross-sectoral information for various stakeholders utilising the marine environment, such as fisheries, marine aggregates, wind farms, nature conservation, shipping etc.
- Meeting short-term national commitments and targets, such as those required by various EU Directives or international conventions such HELCOM etc.

Furthermore, as described by Laffoley et al. (2000) and Connor et al. (2007a), more and more countries and the EU Commission (*sensu* the proposed Marine Strategy Directive) recognise that in order to improve the management of the marine environment an approach is needed that is operational on the relatively limited amount of data available for offshore areas. Similarly, there has over the last few years been a general realisation that nature conservation and general protection of the marine environment should strive to ensure that a network of marine protected areas are protecting a representative part of the ecological units (marine landscapes and/or habitats) present within a specific region or sub-region hereof rather than the preservation of a few individual habitats or species[2].

Thus, given the needs mentioned above and the wish for an improved, cost-effective approach to management of the marine environment several countries has developed, tested and utilized "the marine landscape concept" in their quest for developing an ecosystem-based approach to management. The marine landscape concept is based on the use of available geological, physical, and hydrographic information in order to yield broad-scale ecologically meaningful maps for marine areas with little or no biological information.

BALANCE has followed and successfully implemented the marine landscape concept for the Baltic Sea, and has additionally undertaken prediction modelling of the distribution of habitat-building species and essential fish habitat in the four pilot areas. When examining the methodologies used for mapping and modelling the marine habitats and landscapes in the Baltic the following should be observed:

- For various reasons it has not been possible to gain access to all existing data sets, including military restrictions on e.g. bathymetric data or due to lack of funds for gaining access to certain data sets etc. This has influenced the "exactness" of the produced map.
- Some of the modelled layers have a grid size of 7 km, which may have influenced e.g. the exact location of a known bio-geographic boundary such as the sill in the Sound between Copenhagen and Malmö.
- It has for some areas been difficult to find metadata for the applied datasets, which has influenced the overall confidence rating of the map for some regions.
- Due to the many different classification schemes (one for each country) for e.g. classifying sediments, it has been necessary to compromise sediment classifications when merging data for the entire Baltic Sea region.
- There exist no coherent biological dataset for benthic biological quality elements for the entire region. This has influenced the validation process of the landscape map adversely.
- It should be noted that expert judgment (and to some extend availability of data) has been applied in deciding which environmental parameters should be included in the identification of the Baltic Sea marine landscapes. It could be argued that the inclusion of environmental parameters should be made on a statistical basis, and that other factors and categories should have been considered. This will be a challenge for future work.
- For these reasons a confidence rating of the marine landscape map was developed providing the end user with information about the usefulness and inherited limitations of the map and the layers used to develop it.

[2] The BALANCE project will make a representativity analysis for the Baltic Sea region based on the work presented in this report. Please refer to www.balance-eu.org for further information.

1.2 What is a marine habitat?

The term 'habitat' was first defined by Darwin (1859) as '*The locality in which a plant or animal naturally lives*'. Thus, a spatial environment inhabited by a specific species was described in its original sense by abiotic variables only (Connor et al. 2004). In this sense habitat is the geographic manifestation of a species' requirements to the environment; the niche. Hutchinson (1957) defined a species' ecological niche as a hyper-volume in the multidimensional space of ecological variables within which a species can maintain a viable population.

Most species live together with other species. Such assemblages in the sea have been described as bio-coenoses (Möbius 1877) and later, as communities (Petersen 1913, 1918, Thorson 1950, review in Mills 1969). The assemblages are often found to live under similar environmental conditions in different areas. Such assemblages may be linked to a specific range of environmental variables, thus the habitats of the species are overlapping. Such a shared habitat is termed a 'biotope', and is defined by the abiotic characteristics of the environment and the associated assemblages of species. While some species occur together simply because they share the same niche, others depend on the assembled species, e.g. as mates, food, substrate, or shelter. Compared to 'biotope', the word 'habitat' was regarded to be more familiar to environmental mangers and policy makers (Connor et al. 2004). For the purpose of mapping and modelling of the habitat distribution of marine organisms, the term 'habitat' is now being used *sensu lecto*, synonymously with and in the sense of the term 'biotope', e.g. in the marine mapping projects MESH (Connor et al. 2007b) and in BALANCE.

Some of the marine habitats of concern in HELCOM are characterised by the presence of habitat-building species, some are defined by abiotic environmental variables, and yet others are defined by geo-morphological and physiographical variables linked to prehistoric, geological events (often regarded as landscapes).

The EC Habitats Directive (Anonymous 1992) provides a legal and administrative definition of the term 'habitat', in which '*Natural habitats means terrestrial or aquatic areas distinguished by geographic, abiotic, and biotic features, whether entirely natural or semi-natural*'. According to the Directive, the conservation status of habitats is evaluated on the basis of their range, the maintenance of their specific structure and functions and the status of their typical species.

From these definitions it is clear that the NATURA 2000 Annex 1 habitats and the priority habitats to HELCOM comprise of entities that science has defined as landscapes, habitats, or biotopes. Thus, these habitats occur at highly different scales in space and time, and differ considerably in ecological structure, function, significance and sensitivity. In addition, the methods required to map the HDA1 habitats differ considerably.

Thus, knowing the scale and degree of dimensionality of what we aim to capture and map is essential, and will together with the choice of techniques for mapping and modelling these habitats control which areas are classified as habitats.

1.3 BALANCE approach to habitat identification

Spatial and temporal scale affects habitat fragmentation and connectivity, and thus influences the inferences that can be made about habitat patterns and processes. Time proceeds on a one-way linear dimension. Spatial scale refers to the two or three dimensions of an object (e.g. a habitat or landscape) or a process (e.g. physical or ecological). Spatial scale may be characterized by both grain and extent (Hilty et al. 2006). 'Grain' is defined as the finest level of spatial resolution possible within a given data set, while 'extent' is defined as the size of the study area (Turner et al. 1999). Here, grain ranges from fine to coarse, while extent varies from small to large. Coarse-scale habitat models may predict the general presence or absence of a species or species assemblages broadly across a region, while fine-scale analyses will show the distribution patchiness of a species within an area or region (Hilty et al. 2006). In BALANCE coarse-grained habitat models and landscape maps were developed to enable the extent of the maps to cover the entire Baltic Sea region.

The temporal and spatial extent of marine features phenomena varies greatly because of the scale of the underlying processes that causes change. For example microbiotic communities change within seconds or hours and extent 1 μm - 50 cm, whereas marine animal communities change at a scale of 1 to ≥100 years and extent 1 m - 1 km. The duration of a habitat *sensu lecto* is linked to the generation time of species, as well as to natural maintenance and persistence of the typical species, colonies and/or assemblages associated with it. The generation times of whales and seabirds is up to 100 years or more, whereas fish live 1-20 years and marine benthic macro-invertebrates live for 3 months to ≥10.000 years. Geological and evolutionary changes may take ≥10.000 to millions of years and oceanographic changes may take 1 day - 1 year (Fenchel 2006). Both are potential components of marine landscapes. The spatial scale of a habitat to fish, marine mammals and seabirds may be characterised as the geographic, abiotic and biotic conditions to which the species respond in a hierarchical manner (Fauchald et al. 1999). Fish habitats, for example, may be described for scales ranging from 1^{-2} - 1^{4} kms. At the large scale distributional responses typically are linked to climate and water mass properties which determine the threshold for the species' range, while responses at medium scale (1-50 km) are linked to foraging habitats (properties like hydrographic fronts enhancing the possibility of prey encounter). Responses at the smallest scale (< 1 km) are associated with social aggregations and interactions with prey.

Habitat mapping and modelling activities in BALANCE focused on developing static maps of the present distribution of marine landscapes with a low to medium resolution. As a result, the medium- and long-term natural oscillations of the identified habitats and landscapes could not be addressed, and the adaptations of species to specific oceanographic and seabed structures where prey detection, shelter and various other key functions are optimised could not be resolved during this project. As the broad-scale landscapes identified in BALANCE are classified irrespective of the gradients in the use of the landscapes by single species or assemblages of species the ecological significance of the marine landscapes has not been dealt with. Thus, assessment of the potential support of the various landscapes to habitat-building species, essential habitats to predators and ecosystem services will be an important follow-up to the mapping activities of BALANCE.

Essentially, in order to encapsulate the multi-dimensionality of marine habitat structures marine landscapes essentially need to contain a comprehensive dataset on both coastal, seabed and water column characteristics. The UKSeaMap project (Vincent et al. 2004) included water column or pelagic habitats based on ‘model derived' datasets for salinity and stratification. Pelagic habitat variables were included to some extent in BALANCE, however a more comprehensive description of the pelagic structures is required in the future to fully compliment the BALANCE landscape map.

In nature, disturbance and stability are highly interconnected and the balance between these opposing forces is the driver of many ecosystems. The interrelation may ship change frequently and quickly, but often anthropogenic activities inflict changes at a must faster pace than would occur by natural forces in the sea (Barnes & Hughes 1999). In addition, human activities may deteriorate marine habitats by destruction and habitat displacement. The sensitivity of marine habitats and landscapes is related both to their vulnerability relative to various activities and the resilience they display to these perturbations. These issues were not dealt with in BALANCE, but will be a priority when the landscape map is implemented in the physical planning process of HELCOM and individual countries.

Obviously, the choice of mapping and modelling influences the identification of habitats and landscapes. Three main sources of variability can be identified in relation habitat mapping and modelling:

1. Spatial scale
2. Choice of parameters
3. Choice of statistical model

Spatial scale was dealt with above. Of the three sources of variability the choice of habitat parameters is the most important. Both GIS-based multi-criteria evaluation and spatial prediction models are sensitive towards the choice of habitat parameters. As an example, habitat models based on only topographic parameters always display suitability gradients similar to the major gradients in topography. In BALANCE marine landscapes were classified following expert judgements using a large number of controlling parameters encompassing a wide range of topographic, geographic, geologic, coastal and pelagic conditions. Thus, the classified landscapes should be regarded as relatively robust and ecological sensible. Classification was made in a raster GIS environment using Multi-Criteria Evaluation (MCE) with overlay analysis and hard classifiers. MCE has been established as the standard GIS method for mapping suitability in relation to environmental conditions, and is used as an integral of decision-support systems worldwide. A number of MCE routines have been developed, including fuzzy logic applications, which lower the risk of fault classification (the risk that a grid point due to lack of detailed data on environmental thresholds falls in the wrong class). The more sophisticated MCE applications were not used in BALANCE.

Generalised Additive Models (GAM) based on known key variables in relation to the distribution of habitat-building species and essential fish habitat was the main method used for spatial modelling. GAM has proven a useful technique to incorporate non-linear relationships commonly found between animals and abiotic habitat features into statistical prediction models. A minor drawback of using GAMs for spatial prediction is the limited possibility to include samples from non-standardised surveys (e.g. satellite tracking, acoustic data and observations by layman, Hirzel et al. 2002). Additionally, GAM has limited capacity compared to suitability models like BIOCLIM and ENFA to describe the basic tenet of the niche theory; that fitness (or habitat suitability) does not bear monotonic relationships with conditions or resources, but instead decreases from either side of an optimum (Guisan & Zimmerman 2002). However, as only data from standardised surveys were used in BALANCE and habitat suitability was not explicitly modelled these drawbacks are not likely to have influenced the habitat models negatively.

Within the four pilot areas, the aims were to:

- Identify data of possible use for mapping and modelling of marine habitats in the Baltic Sea area, including of marine Natura 2000 Annex 1 habitats and/or the marine habitats of selected species.

- Agree on approaches to mapping and modelling of selected marine Natura 2000 Annex 1 habitats and/or of the habitats of selected species.

- Develop models to demonstrate the usefulness of data and various methods for mapping and modelling of selected marine habitats in the Baltic Sea area.

Produce maps of marine benthic and pelagic habitats that can be used for evaluation of boundaries of marine landscapes in WP2 and for modelling of habitat coherence and maps comparison in WP3.

Classification systems (e.g. EUNIS) provide classes with discrete boundaries that can be perceived by the human brain (incl. of mangers and policy makers, Davies & Moss 2004, Backer et al. 2004). Von Nordheim et al. (2000) provided a classification of threatened marine biotopes, yet a full Baltic classification system has not yet been developed, and the existing systems have not been found to apply very well to the Baltic Sea region. The classification system developed in UK waters (Hiscock 1994, Connor et al. 2003) is difficult to apply to the Baltic Sea due to the significant differences in prevailing environmental conditions (tide, salinity, age/geo-history). Classification systems were absolutely required when trying to map habitats by hand or print. However, the strength of classification systems is to organise complex structures in a simplified form that can be perceived by the human brain. Thus, all elements are organised according to a limited number of categories of a few and identical factors make complicated structures. The development from analogue to digital and GIS methods have to some extent made classification systems redundant. Using modern methods in mapping of marine habitats, particularly predictive modelling species distribution, these can be overlaid in a GIS based on multiple and highly different factors.

BALANCE methods to marine habitat mapping are outlined in chapter 2. BALANCE was able to capitalise on the recent national initiatives of mapping of marine benthic habitats in Norway (MAREANO), Finland (VELMU), and Sweden (SAKU). These activities included the measuring of a variety of factors using modern equipment (e.g. multibeam, backscatter, CTD, flow meter, ROV) along with more traditional sampling gear (dredges, grab samplers). These national programmes also provided a srong basis for multi-criteria evaluation of marine landscapes in GIS and predictive modelling of the distribution of several marine organisms, e.g. *Ulva rigida* (Runca et al. 1996), *Laminaria hyperborea* (Bekkby et al. 2002), *Zostera marina* (Ferguson & Korfmacher 1997), and blue mussels (Brinkman et al. 2002).

The BALANCE case studies on predictive modelling of spatial distribution of marine habitat were carried out on selected species. These included benthic habitat-forming kelps, bladder wrack, and other macro algae, submerged plants, reef-forming mussels and habitat-structuring lobsters. Essential fish habitats were modelled for the ecological and economical important species plaice, flounder, sole, pike, perch, pike-perch, roach, and cod. The latter habitats were modelled specifically for juvenile or spawning stages.

1.4 Application in marine spatial planning

Over the last decade there has been some considerations on the potential uses and end users of marine landscape maps. Marine landscape mapping can be used as a tool and source of information in environmental management including proper governance of large sea areas. An approach which provides a tool for an ecological meaningful regulation of human activities (Connor et al. 2007a) and which in regard to environmental protection measures ensures an ecosystem-based approach to management rather than the traditional "*one nation – one approach*". In a semi-enclosed sea, such as the Baltic Sea surrounded by multiple nations with many stakeholders, cross-sectoral and trans-national co-operation is essential in the development of marine landscapes. This is partly in order to gain access to relevant and coherent environmental data covering the territorial waters of many nations, partly to ensure the durability through wide acceptance and lastly, but most importantly, for ensuring an ecosystem-based approach to management and environmental protection. It also provide environmental managers with a practical, cost-effective solution to the managing and planning of large off-shore marine areas as physical and oceanographic information typically are available whereas biological data often are very scarce if available at all.

The main purpose for developing a Baltic marine landscape map is to present a broad scale, trans-national characterisation of the marine environment in the Baltic Sea region creating an ecosystem-based tool which support various national planning and management requirements. Specifically these include:

1.4.1 Implementation of EU Directives

All EU Member States are required to implement the EU Water Framework Directive (WFD), the EC Habitats Directive (HD) and the proposed EU Marine Strategy Directive (pMSD). These all require a more holistic or ecosystem-based approach to the management of the marine environment, which should, directly or indirectly, be based upon a broad scale characterisation of the marine environment as stated in e.g.:

- The EU Water Framework Directive (art. 5.1, Annex II) *"- an analysis of its [river basin district] characteristics"*.
- The EC Habitats Directive (art. 3.2, Annex I): *"- shall contribute to the creation of NATURA 2000 in proportion to the representation within its territory of the natural habitat types and the habitats of species…"*.
- The proposed EU Marine Strategy Directive (art. 7.a, Annex II) *"- an analysis of the essential characteristics and current environmental status of those waters…*

…and covering the habitat types, the biological components, the physio-chemical characteristics and the hydromorphology".

- All three directives (WFD art. 5.1, Annex II; pMSD art. 3.1; HD art. 1.c) also require a trans-national approach covering entire regions such as the Baltic Sea region.

The challenges for the EU Member States are to develop one approach promoting synergies and convergence in the implementation of the directives rather than developing kseveral parallel, potentially conflicting characterisations of the marine environment. Marine landscape maps have the potential to be further developed into just such a tool.

1.4.2 HELCOM actions for ecosystem-based management

The marine landscape maps provide a coherent unified ecological map describing the entire Baltic Sea region disregarding e.g. national boundaries. This gives environmental managers a first time opportunity to gain a holistic overview of national distribution and extent of broad scale ecological units and relate it to a Baltic perspective, thus promoting a true ecosystem-based approach to protection of the marine environment. BALANCE intent to apply these maps in a broad scale assessment of the network of marine protected areas in the Baltic Sea region identifying strengths and weaknesses of current protection schemes. Certain inherited limitations of applying the marine landscape map for this purpose are discussed in section 1.6 and by up-coming BALANCE reports.

The Baltic Sea Action Plan (BSAP) aim to apply the ecosystem-based approach to the management of the Baltic Sea. The BSAP will set a definition of 'good ecological status' for the entire Baltic Sea as well as specific environmental targets and necessary measures, and it will be instrumental to the successful implementation of the EU Marine Strategy in the Baltic Sea ecoregion. It will be difficult to reach these aims without a broad scale characterisation of the marine environment such as the marine landscapes.

The availability of a broad scale ecological map for the Baltic Sea ecoregion can provide sea use planners with an opportunity to incorporate an ecosystem-based approach when making planning decisions on a regional scale taking a layer with the natural values into account, and thus help in an assessment of the potential impact of human activities. E.g. are certain activities that depend on the use of large areas, such as wind farming, unintentionally targeting large proportions of specific ecological units? For more local issues more detailed habitat maps are required. BALANCE is testing this in the Archipelago Sea and is making an overview of habitat mapping activities in 4 pilot areas in the Baltic Sea and Kattegat.

1.4.3 Strategic planning

Marine landscape maps can be applied for several strategic purposes as well. These include an application as a baseline study of the complexity within a region providing field surveyor with a planning tool for areas with limited information. These maps could also provide an informed tool for setting up monitoring programmes, as it would enable a spread of sampling stations across the continuum of ecological units present in a region.

1.4.4 Maritime safety

Marine landscape maps may be used in regard to maritime safety issues. It provides an ecological input for a region showing the amount and distribution of specific natural values. If combined with a sensitivity map this would provide valuable information for handling a major shipping catastrophe or oil spill by supplying a baseline for a prioritisation of effort in regard to natural values. E.g. showing the complexity of a near shore area as a sandy beach will be easier to clean than a more complex stony region.

2 MATERIAL & METHODS

Mapping and modelling of marine habitats in the Baltic Sea region varies in approach between the different types of habitats, between specific areas, and between countries. The challenges of marine habitat mapping and modelling are numerous. Methodologies adopted by BALANCE for meeting some of them are described below. The use of different approaches often reflects genuine difference in physical or ecological structure, or both, although in some cases difference in approach reflects spatial and qualitative difference in available data and lack of a common classification system of marine benthic and pelagic habitats in the Baltic Sea region. Despite these differences a large degree of harmonisation has been achieved in the methods applied in the different pilot areas, and this has to a large extent secured the comparability of results and facilitated a trans-national and cross-sectoral approach.

2.1 The study area

Given the complex structure of the many coastal areas within the Baltic Sea Region, e.g. the fragmented archipelagos and strong environmental gradients, it is very costly to perform surveys that cover all potentially interesting areas for the species/communities of interest. Multi-criteria GIS mapping and spatial predictive modelling, using key habitat characteristics to identify areas of particular interest could provide a tool to circumvent this problem. The underlying concept of these models is that certain habitat characteristics are needed to host specific species, assemblages, or communities. Using these environmental variables as acceptance criteria or predictors maps may be produced that extrapolate field data to GIS surfaces which can be used in marine spatial planning, and that may promote an enhanced understanding of the sensitivity and functioning of coastal ecosystems.

2.1.1 Characteristics of the Baltic Sea region

Characteristics of the Baltic Sea region: The Skagerrak, Kattegat, the Danish Straits, and the Baltic Sea together compose the second largest brackish area in the world (Segerstråle 1981, Thorson 1950) with a number of basins varying from almost fresh water in the north-eastern part, the Bothnian Bay, to the saline waters of the Kattegat and Skagerrak with a distinct salinity gradient in the Danish Straits.

Fig. 1. *Skagerrak, Kattegat and the Baltic Sea as seen from space, with the four pilot areas. The image is used by kind permission of the SeaWiFS Project, NASA/Goddard Space Flight Centre and ORBIMAGE.*

Bathymetry and substrate. The Baltic Sea region has been moulded into its present shape during several glacial and post-glacial periods with the results of often highly complex topography and substrate. In Kattegat large islands, reefs and sandbanks dominate this area with the remnant river channels forming the deepest part. Numerous large inlets, bays, and fiords are located along the coastline (Figure 1). The western Kattegat shores are mostly characterised by a mixed geological composition of mainly sand, gravel, and boulders, while bedrock dominates the eastern shores. The transition from the Kattegat to the Baltic Sea is dominated by the sills in the Sound and at Gedser-Darss. The Baltic Sea is split into a number of deep basins reaching depths of down to 459 m. A striking characteristic of the sub-littoral parts of the Baltic Sea is the existence of a number of large submarine banks extending almost as a string of shallower areas through the central Kattegat, the Straits, western Baltic and the Baltic proper. The southern coast of the Baltic Sea is mainly characterised by exposed sandy shores often with lagoons separated from the sea by gravel and sand banks. More to the north in the Gulf of Finland and the Archipelago Sea numerous skerries and islands span the Baltic Sea almost bridging the area between Åbo in Finland and the Stockholm Archipelago. To the far north the shore is mostly composed of bedrock interspersed with many small gravely bays and lagoons. Furthermore, large areas are influenced by massive land rise with the seafloor rising more than 8 mm per year in the Quarken area, which creates a unique range of habitats in where the sea slowly develops into land.

Hydrography. The total volume of the Baltic Sea including the Danish Straits is approximately 21.700 km^3 with a surface area of 415.200 km^2 reaching depths of up to 459 m with an average depth of 52 m (Andersen & Pawlak 2006). The fairly shallow Kattegat and the Danish straits form the transition zone between the low saline Baltic water and high saline waters of the North Sea and the Atlantic Sea. The Baltic Sea region is characterised by the almost total lack of tide (Hällfors et al. 1983), which makes the salinity regime very stabile in often very large areas. Many areas are temporally or permanently stratified, which together with the intense eutrophication causes large areas to be oxygen-depleted (Ærtebjerg et al. 2003a, b). The permanent stratification is maintained by temperature differences in the water column as well as the large annual efflux of fresh water from the many rivers in the region combined with occasional influx of heavy, high saline water from the North Atlantic and Norwegian Sea through the North Sea into Skagerrak and Kattegat and over the sills in the Danish Straits. The weaker temporal stratification occurring in shallow waters will normally collapse due to storm events during autumn and winter mixing the water column. There are large annual changes in surface temperature which results in up to 4 months of ice coverage during winter in areas of low salinity such as the Bothnian Bay (Jansson 1980).

Biology and organisms. The Baltic Sea region is fairly young, less than 10.000 years of age. Thus the majority of species that now live here have emigrated from adjacent regions, although a few endemic species (e.g. glacial relicts) are known from the areas (HELCOM 2006). In the outer part, North of the Danish Strait, more than 3000 marine macro fauna species are known to occur, in the middle part, the transition zone the number of marine and brackish water species count approximately 200 species, while in the inner parts of the Baltic Sea freshwater species counts 3000 or more, most of which are insects (Remane 1934). Similarly for the vegetation, macro algae species are abundant in the outer parts, while only a few are able to live in the inner, low saline parts of the region. Here instead, the vegetation is made up of salt-tolerant or freshwater emergent and submerged plants and macro algae species.

2.2 The BALANCE pilot studies on habitat mapping and modelling

In total 16 pilot studies on habitat mapping and modelling were carried out within the four pilot areas:

1. Kattegat-Skagerrak
2. Bornholm Deep
3. Archipelago Sea
4. Coastal areas of Estonia, Latvia and Lithuania

Accounts of the pilot studies are provided in Appendix 1 and 2. In chapter 3 the results of the pilot studies are summarised and discussed in relation to data availability, use of GIS and statistical methods, quality of the resulting habitats maps, and the application for marine conservation as well as for general spatial planning and management. Emphasis has been on evaluation of methods that can aid in mapping of NATURA 2000 Annex 1 habitats as well as on mapping of benthic habitats of bottom structuring species and of benthic and pelagic essential fish habitats.

Several of the case studies are rather comprehensive, thus detailed descriptions of those studies are being published as separate Balance Interim reports (see references below).

Table 1. Pilot studies of mapping and modelling of marine benthic and pelagic habitats in the Baltic Sea region in the BALANCE pilot areas 1, 2, 3, and 4.		
Marine habitats	**Mapping and GIS analyses**	**Spatial predictive models**
Pilot Area 1 (PA1): Kattegat - Skagerrak	3.1 Mapping of seabed and habitats in the Norwegian Rauøfjorden-Hvaler	4.1 Spatial prediction of *Laminaria hyperborea* in the Norwegian Skagerrak
	3.2 Mapping of NATURA 2000 habitats in Kattegat using acoustic and ground truth methods	4.2 Spatial prediction of *Nephrops norvegicus* in the Swedish Skagerrak
	3.3 Mapping of NATURA 2000 "reef" between Læsø and Anholt islands, Danish waters	4.3 Spatial prediction of nursery grounds for juvenile flatfish in the Danish Kattegat
	3.4 Mapping of soft bottom fauna between Læsø and Anholt islands, Danish waters	
Pilot Area 2 (PA2): Bornholm deep		4.4 3D-modelling of pelagic habitats of cod and sprat spawned eggs and adults
Pilot Area 3 (PA3): Baltic Archipelago Sea	3.5 Mapping of NATURA 2000 habitats in Swedish and Finnish waters	4.5 Spatial prediction of *Fucus vesciculosus*, *Zostera marina*, and *Mytilus edulis* in the Finnish Archipelago
	3.6 Mapping of EUNIS habitats in Finnish waters	4.6 Spatial prediction of fish habitats in the Swedish Archipelago
Pilot Area 4 (PA4): Coastal areas of Estonia, Latvia and Lithuania	3.7 Mapping of EUNIS habitats in Lithuanian waters	4.7 Modelling of species habitats in Estonian waters
	3.8 Wave impact on coastal habitat along the Latvian Baltic Proper coast	4.8 Spatial prediction of *Furcellaria lumbricalis* in Lithuanian coastal waters

2.3 *Data format (GIS)*

A variety of data were used in mapping and modelling of marine habitats. Some were based on analogue information which had to be digitalised, while others already existed in a digital format. All data were geo-referenced which allowed extensive use of GIS tools. Throughout all studies of benthic habitats, the GIS software packages of ArcView and ArcGIS were used as standard. The datum and projection varied between studies. Coordinates of the final results were translated into WGS84 UTM 34N.

Most of the pilot studies were carried out within smaller areas involving data from a single country. However, in pilot area 3 a time-consuming effort has been put into the harmonising of data and results between Sweden and Finland to create combined maps of the benthic habitats studied.

Temporal variation has not been considered in most of the pilot studies. One exception is the modelling in pilot area 2 of the pelagic habitat of adult cod spawning and of the spawn. However, temporal scale can be added to all the habitat models if and when time series data are available.

Both vector and raster data were used extensively in BALANCE. Vector data were included that consist of data layers based on point, line, or polygon objects including attribute data to each object. Most biological data were point data (e.g. grab samples). Line data may occur (e.g. transect or dredge samples), and were commonly used to describe geo-physical structures (e.g. coastlines). Since the geo-referencing of samples in the field is often somewhat inaccurate biological data could often be interpreted as polygon data (e.g. algal cover of 0.25 m^2). Raster data included digital images, interpolated surfaces and model results, typically with a specific value designated to each cell (or pixel). Transformation between vector and raster data layers and between vector and raster data layers were made frequently using standard routines in ArcGIS. Other data types included ASCII data, - a format that was commonly used for export and import of data between databases, GIS and statistical software.

2.4 Data for mapping and predictive modelling

Mapping and modelling can be made on the basis of point or transect measurements in the form of vector data and as interpolated or modelled surfaces in the form of raster dat. Raster data may include areas of missing or invalid data, as an example a land mask or mask defining area of low coverage.

2.4.1 Bathymetry

Bathymetry data constitute a vital source of information for mapping and modelling of benthic habitats, - both as direct measurements of depth and depth models which again can be used as a basis for calculation of the topography of the seabed (complexity, slope, aspect). One of the key map outputs from BALANCE was a depth model covering the entire Baltic Sea. The resolution of depth measurements used varied depending on the survey methodology. In some areas only scattered depth measurements were available, while in other, typically shallower areas measurements had a relatively high resolution due to intensive survey campaigns using echo-sounders, side-scan sonars and/or multi-beam. High resolution but lower precision in depth measurements were also obtained from aerial photographs and satellite images from coastal areas.

2.4.2 Substrate

The type of substrate is one of the main factors affecting the distribution of organisms that lives in, on or near the bottom. These organisms include representatives of all major kingdoms and classes, including vira, bacteria, micro and macro algae, vascular plants, meio to mega fauna, fish, reptiles, birds, and mammals. Many of the larger phytobenthic, epifauna, and infauna are habitat structuring species and may provide secondary substrate for orther organisms. Type of substrate alone does not explain all patterns of distribution. However, it is a key variable and should be included in the processes of mapping and modelling of benthic habitats.

Sediment core and grab samples has been used to create analogues sediment maps in the past, including information on composition of sediment grain size. In recent years, such samples have proves useful to interpret and validate back-scatter data to create more accurate and fine scale sediment maps.

Back-scatter data from multi-beam surveys has proved to be an efficient and cost-effective method of achieving accurate and high resolution data on spatial distribution of the different types of substrate. Pending on the substrate, back-scatter information can be verified (ground truthing) by sediment samples, video recording, and diving observation. The latter is often required to distinguish between bed rock, boulders, and pebbles at photic depth (≤30 m) where erect macro algae cover the underlying substrate.

Video recording (incl. by ROV) has been used to validate type of substrate at depth down to 100 m or more.

Aerial photograph surveys have been used to establish and monitor spatial changes of the coastline. In recent years, such surveys have been replaced by more the accurate and cost-effective surveys based on satellite images (Interim report). In mapping of marine habitats aerial photographs have been used to map costal sand banks in shallow water in the Northern Kattegat from 0 to 10 m depth. Sand banks slightly covered by water at 0 to 20 m depth represent one of the NATURA 2000 Annex 1 habitats (****). The more shallow habitats, at 0 to 3 m depth, are often difficult to map from the sea (by ship or vessel), why aerial photographs will be useful in mapping of the shallow sand banks. Similarly, shallow boulder reefs will be visible on aerial photographs and thus possible to map (for shallow, biogenic habitats, please see below). The method requires fairly calm weather, and is useful in shallow water only, as it requires high visibility of both the air and the water. The later is rarely the case in the Baltic Sea region.

2.4.3 Oceanography

A third key class of environmental data for mapping and modelling marine habitats are oceanographic variables. In BALANCE only two oceanographic parameters were selected: light and wave exposure. Both variables are known habitat drivers for submerged vegetation. Light exposure (calculated by combining slope and depth values) is the key variable in structuring the distribution of primary producers, in the water column and on the bottom. The quality of light climate on the seafloor is determined by three main factors: solar influx, depth, and water turbidity. The combination of these three parameters determines the light climate at the particular location which is one of the main factors determining the vertical distribution of different algae species enabling the formation of different pelagic and benthic communities. Light intensity has proven an important measure in predictive modelling of macro algae and plant habitats, of e.g. *Laminaria hyperborean*, *Fucus vesiculosus*, *Furcellaria lumbricalis*, and *Zostera marina*.

Wave exposure is a key variable in structuring flora and fauna, particularly on hard substrates, in the littoral zone and in shallow water between 0 and 10 m depth. In the Baltic Sea hydrological parameters in the form of wave action and water-level fluctuations is highly influencing the development of phytobenthic communities (e.g. Kautsky 1988, Kautsky & van der Maarel 1990). Wave activity affects the phytobenthic communities through the formation of substrate quality and direct physical disturbances. Wave exposure may be estimated in many ways and the method chosen was the Simplified Wave Model (SWM), which is fully described by Isæus (2004). The method is called ”simplified” since it uses the shoreline and not the bathymetry as input for describing the coastal shape. This is an adaptation to the fact that detailed bathymetry data is often poor, or restricted, and is therefore usually not available for larger areas such as a national coastline or for an entire regional sea. The method also uses fetch, adjusted for refraction/diffraction patterns, and wind speed from 16 directions. A nested-grids technique is used to ensure long distance effects on the local wave exposure regime. The resulting grids have a resolution of 25 m.As mapping and modelling of submerged vegetation and other habitat-building species was focused on coastal areas exposure and other variables related to currents were not included in the mapping and modelling activities.

2.4.4 Chemistry

Both salinity, temperature and oxygen are important drivers of the variability of benthic vegetation and invertebrates in the Baltic Sea, and were included in most of the mapping and modelling activities. Especially, they played an important role during the mapping of Baltic-wide marine landscapes.

2.4.5 Biological information

No biological data were used as classification or predictor variables. Empirical data were included for all modelled species of submerged vegetation, invertebrates and fish. In most cases modelled distributions were evaluated using independent empirical data.

2.5 Development of multi-criteria evaluation models

Baltic-wide landscape maps and regional maps of NATURA 2000 habitats in Pilot area 3 were developed by GIS-based multi-criteria evaluations (MCE) using the Raster Calculator routine of Spatial Analyst in ArcGIS and the Map Modeller routine in Erdas Imagine. Selection of data layers for the MCE models was based on expert judgements in both studies. Each data layer in the classification was given the same weight and all acceptance criteria were defined as hard classifiers (i.e. criteria defined numerically with precise cut-off values). MCE models of Baltic-wide topography/bedform and coastal physiographic features were developed using the following data layers:

- Topography and bedform features: bathymetry and sediments: troughs, basins, mounds, plains, valeys and holes, slopes, wave/mega ripples
- Coastal physiographic features: estuaries, fjords and fjord-like inlets, bays, sounds, archipelagoes

Baltic-wide broad-scale benthic landscape maps were developed by combing three data layers:

- Surface sediment
- Available light
- Salinity at the seabed

The NATURA 2000 habitat models of Pilot area 3 used the following data layers:

- Bathymetry: Nautical chart, topographic maps (1.25000)
- Coastline: ortophoto (1 m), Landsat ETM (25 m) basic map (1:10000), Esker Island database (vector)
- Substrate and soil: 1:50.000
- Wave exposure: 25 m model
- Secchi depth: national wq database
- Coastal exploitation: 1:10000
- Archipelago zone (50 m)

- Slope: from depth model (25 m)
- Photic depth: modelled from land coverage in 5 km radius and secchi depth.

Table 2. Data and analyses used for MCE-modelling of NATURA 2000 Annex 1 habitats and EUNIS habitats in the Baltic Sea region.

HABITAT	ANALYSES	DATA
1110 Sublittoral sandbanks	Ground-thruthing MCE	Acoustic: Multibeam Depth, substrate, slope
1130 Estuaries	MCE	Depth, freshwater flow, not exposed coast, presence of reeds
1150 Coastal lagoons	MCE	Depth, size, lack of freshwater inflow
1150 Gloes	MCE	Depth, size, lack of freshwater inflow
1160 Large, shallow inlets & bays	MCE	Depth, size, lack of freshwater inflow, geometry
1170 Reefs	Ground-thruthing MCE	Acoustic: multibeam, interferometric and side-scan sonar Depth, exposure substrate, slope
1610 Baltic Esker Islands	MCE	Depth, substrate, slope with submergence
1620 Boreal Baltic & small islands	MCE	Depth, exposure, location in archipelago (outer zone)

Table 3: The criteria used to delineate potential Annex I habitats in Pilot area 3

Habitat	Depth	Size	Exposure to wave action	Sub-strate	Freshwater flow	Other
1110 Sandbanks	<= 30m[1]			Sand >= 70%[2]		Elevated from surrounding sea-floor[2]
1130 Estuaries	<= 3m at the mouth				At least 1 river with >= 1km^2 watershed[1]; >=2 m^3/s[2] flow	Not on an open coast Reed beds present
1150 Lagoons	Max <=6m	< 30 ha			No river inflow	
1160 Large Bays	<20% of area >15m (6m*) deep	>= 20ha (>= 100ha*)			No river inflow	Wider than long at least 1:1
1170 Reefs	<= 6m[1] <= photic depth[2]		Sheltered and higher (algal zonation)	Hard substrate		Elevated from surrounding sea-floor
1610 Esker islands	<= 10m[1] <= photic depth[2]			Sand or moraine (>= 50% of cover)		Submerged part elevated from the surrounding sea-floor[2]
1620 Boreal Islets	<=6m[1] <= photic depth[2]		Sheltered and higher[1]			In outer archipel-ago zone[2]

2.6 Development of predictive habitat models

In this context, predictive modelling covers statistical analyses correlating biological data with physical, oceanographic, and chemical data. Models were built for areas for which environmental data were available for the major part of the area. Response samples of species and habitats were both given as presence-absence and abundance/density. Environmental predictors were selected with known and potential structuring effect, such as depth, substrate, wave exposure, and salinity. A subset of the environmental predictors explaining most of the variance of the response variable was used to predict the species/habitat distribution.

For reasons of comparability Generalised Linear Models (GLM) and General Additive Models (GAM) were used as a basis for all predictive modelling in BALANCE. GLMs and GAMs are the most widely used statistical models in the fields of ecological modeling, biodiversity and conservation. GAM, which is a semi-parametric extension of GLM, has been successfully applied in habitat modelling in terrestrial as well as marine studies, due to its flexibility in resolving complex responses between species and environmental data. In addition, GAM was chosen due to its characteristic smoothed coarse-scale modelled distributions which fitted the coarse-scale focus of BALANCE well. In order to ease trans-national collaboration on the application of GLM/GAM the software package GRASP was used by all WP2 partners. GRASP stands for Generalized Regression Analysis and Spatial Prediction and is a combination of advanced S Plus functions and GIS written R by A. Lehman (Lehman et al. 2002).

GRASP prediction models included both full (all variables forced into the model) and stepwise (only best predictors selected) models. The function GRASP.STEP is the central part of GRASP. It selects significant predictors for each response variable. Several options must be set first from GUI. By default, the stepwise procedure starts with a full model containing all potential variables (selX), and goes in loop to try to eliminate one variable at the time. At each step, the less significant variable is dropped from the model, and the loop starts again with the remaining variables. Tests are used to decide whether a variable should be dropped, kept, or reintroduced.

An important step in setting up the correct variance model is to determine what kind of distribution the dependent variable has. Advanced variance models as GLM (generalized linear model) and GAM (generalized additive model) provides the possibilities to model a range of different distribution besides the Gaussian, such as Poisson, binomial and Negative binomial distributions. Spatial auto-correlation is a serious problem in spatial prediction modelling, and is a current focus of development (Segurado et al. 2006). Given the expected inflation in the estimates of significance when analysing spatially auto-correlated variables (like transect data) these need to be adjusted before predictions can be fully applied in management. Unfortunately, it was not possible within the framework of WP2 to ensure that adjustments for spatial autocorrelation were made.

In many fields, the approach of null hypothesis testing is being replaced by model selection as a means of making conclusion based on evidence. In the latter, several models, each representing one hypothesis, are simultaneous evaluated in terms of support from observed data. Where models have similar levels of support, model averaging can be used to make robust parameter estimates and predictions (review in Johnson & Omland 2004). The Akaike Information Criteria (AIC) is a conservative, model selection approach that considers both fit and complexity. AIC enables models to be compared simultaneously. AIC is available in GRASP.

The modelling results were transferred to a GIS , and a suite of routines were used for comparison and evaluation of resulting maps in ArcGIS, as well as for merging the results of different habitats models and maps into a single, coherent map of the included habitats.

Table 4. Data and data models used for predictive modelling of the spatial habitat distribution of selected marine and brackish species of macro algae, plants, benthic structuring invertebrates, and essential marine and freshwater fish habitat. Predictive models cover distribution of adult, unless otherwise mentioned. Predictor variable in brackets have been used indirectly to generate other variables. Resolution of grid cells is given in meters.

Predicted habitat of species	**Predictor variables**						**Response variables**				
	Depth	**Wave Exposure**	**Light exposure**	**Substrate**	**Curvature**	**Distance**	**Species occurrence**	**Statistics**	**Model selection**	**Comparison**	**Lacking**
Algae and plant habitats											
Laminaria hyperborean, kelp LAMHY full model[1] Pilot 1	(digital elevation model) 25 m	(SWM, Isæus 2004) 25 m	(Slope, Aspect) 25 m		(depth) 500 m		P/A	GAM ?distribution	AIC	Spatial Analyst	Sediment Topography 0.1-10 m res.
Fucus, **seaweed Pilot 3											
Fucus vesiculosus, **seaweed Pilot 4	50 m			Sediment 50 m	Slope 50, 100, 500, 1000, 5000 m	Coastline vector		GAM ?distribution	AIC		?
Charophytes Pilot 4	50 m			Sediment 50 m	Slope 50, 100, 500, 1000, 5000 m	Coastline vector		GAM ?distribution	AIC		?
Furcellaria lumbricalis, red algae Pilot 4	50 m			Sediment 50 m	Slope 50, 100, 500, 1000, 5000 m	Coastline vector		GAM ?distribution	AIC		?
Zostera marina, eel grass Pilot 3											
Zostera marina, eel grass Pilot 4											

Benthic invertebrate habitats	**Depth**		**Slope & Aspect**	**Sub-strate**			**Species occurrence**	**Statistics**	**Model se-lection**	**Com-parison**	**Lacking**
Mytilus trossulus, Baltic blue mussel Pilot 3											
Nephrops norwegicus, Norwegian lobster Pilot 1	Multi-beam ≤10 m		Slope Aspect ≤10 m	Back scatter ≤10 m			SPI, burrow frequency 35 points	GAM binomial dis-tribution, df=2			Video rec. Current velocity

Fish habitats, benthic marine	Distance from shore, m	Wave Exposure	Slope, %	No. of sand banks	Place, m	Year	Species occurrence	Statistics	Model selection	Comparison	Best variables	Lacking
Pleuronectes platessa, plaice, 0 year	(Coastline Depth curve 5 m)	(Fetch Wind condition Topography Wave breaking)	(Depth model?) 100 m	(No./100 m coastline)	(distance: sample site to nearest coastline)	1995	Abundance 40 % CPUE	GLM: Place GAM: other Negative binomial distribution	AIC	?	Slope Wave exp. Sand banks	Sediment Topography
Pleuronectes platessa, plaice, 1 year	X	X	X	X	X	X	Abundance 26 % CPUE	See above	AIC		Slope Wave exp. Year Sand banks	Depth Sediment
Platichthys flesus, flounder, 0 year		X	X	X			Abundance 40 % CPUE	See above	AIC		None	Depth Sediment
Platichthys flesus, flounder, 1 year	X	X	X	X	X	X	Abundance 26 % CPUE	See above	AIC		None Distance & Sand banks slightly negative	Depth Sediment
Solea solea, sole, 1 year Pilot 1	X	X	X	X	X	X	Abundance 26 % CPUE	See above	AIC		Slope Wave exp. Sand banks	Depth Sediment

Essential fish habitats, benthic brackish/freshwater **Pilot 3**	Secchi depth	Wave Exposure (SWM, Isæus 2004)	Depth					**Statistics**	**Model selection**	**Comparison**	**Best variables**	**Lacking**
Perca fluvialis, Eurasian perch, 0 year Pilot 3	GIS model, vert. range 0.3-10 m (25 m) Pos.	SWM (Isæus 2004) (25 m) Neg.	Nautical charts, vert. range 0-6 m (25 m) no cor.						GAM Binomial df=3			High-resolution depth (military restriction) ≤10 m
Perca fluvialis, Eurasian perch, spawning	Pos.	Neg.	optimum at 1 m depth						Same as above			Same as above, Also vegetation cover
Esox lucius, northern pike, 0 year	Pos.	Neg.	(Neg.)						Same as above			See above,
Sander lucioperca, pike-perch, 0 year	Neg.	Neg.	no cor.						Same as above			See above
Rutilus rutilus, roach, 0 year	Neg.	Neg.	optimum at 2 m depth						Same as above			See above
Essential fish habitats, pelagic **Pilot 2**	Salinity	Temperature	Oxygen				**Species occurrence**	**Statistics**	**Model selection**	**Comparison**	**Best variables**	**Lacking**
Gadus morhua, cod spawn (eggs)			> 2 ml/L Cod egg surv. *f*					Bryan-Cox-Semtner model				
Gadus morhua, cod (spawning adults)	> 7 psu no cor.	no cor.	> 60 % sat.					Same as above				
Sprattus sprattus , sprat spawn												
Sprattus sprattus, sprat (spawning adults)		< 5.1 °C	< 1.0 ml/l					Same as above				

2.7 *Classification of marine habitats*

Geographical information systems offer a number of ways to analyse and synthesise spatial data into classes that are meaningful from the viewpoint of a habitat classification system. The approach presented here would be applicable to any hierarchical classification system where data is available on the factors determining classes. At present the Environment Agency's EUNIS classification of marine habitats is the only available classification system for the Baltic. The Baltic classification in the EUNIS classification has been achieved using the classification system of HELCOM for red-listed biotopes (von Nordheim et al. 1998).

An attempt at mapping habitats in the area where substrate data is available in the Archipelago Sea (Pilot Area 3) was made in order to assess the applicability of the EUNIS system for the Baltic Sea. The results indicate that Baltic classes do not follow the hierarchical structure of the classification very well and are somewhat inconsistent to what is included on each level (see Table 1). Improvements to the classification are surely needed regarding marine habitats in the Baltic Sea. The current system is, however, the best available today. The classes used here for BALANCE purposes follow as closely as possible the EUNIS classes on levels two, three and four.

The main aims of the habitat classification in Pilot Area 3 were to combine available GIS data layers to produce habitat maps at EUNIS level 2 and to integrate results from previous modelling exercises on mussels, algae and angiosperms to complete maps at EUNIS level 3. The analysis of EUNIS classification level 2 and 3 habitats used four sets of GIS layers:

I. Substrate data classified according to the BALANCE substrate classification, which corresponds fairly well to the substrate classes used in the EUNIS classification.
II. The photic layer derived from a model where secchi depth is predicted based on the level to which an area is enclosed (see chapter x.).
III. Wave exposure data classified into 3 categories (sheltered, moderately exposed and exposed), first using the cut-off values derived from an analysis of the distribution of lichens and algae on shores (Isaeus, xxxx) and then combining these seven classes into 3.
IV. Raster layers (5 m cell size) with probability of presence of Mytilus trossulus, algae and angiosperms (see chapter x)

Data were combined in GIS using multi-criteria evaluations following the same standards as for the Baltic-wide landscape maps (chapter X). The first two layers (substrate and photic depth) were combined in a GIS overlay analysis to produce maps of EUNIS level 2. The same approach, also including the third dataset (wave exposure), was used for those level 3 habitats that do not include biotic information (Table 1). It was recognised that a combination of the original GIS analysis and additional layers achieved by habitat modelling techniques could be used to complete the maps on level 3. Habitat modelling enables mapping those classes that cannot be done using GIS analysis of abiotic data layers alone. The biological data required for the models, including observations of mussels, algae and angiosperms, was only available for a very small part of the study area. This 100 km^2 area was used as an example area in an attempt to develop and demonstrate the approach. Table X describes the data needed for making a map of the habitat classes in the current EUNIS marine habitat classification that are found in the study area.

Table 5: EUNIS Habitats in the Study Area

Level2		Level3		Level 4	
A3	Infralittoral rock and other hard substrata	A3.4	Baltic exposed infralittoral rock		
		A3.5	Baltic moderately exposed infralittoral rock		
		A3.6	Baltic sheltered infralittoral rock		
A4	Circalittoral rock and other hard substrata	A4.4	Baltic exposed circalittoral rock		
		A4.5	Baltic moderately exposed circalittoral rock		
		A4.6	Baltic sheltered circalittoral rock		
A5	Sublittoral sediment	A5.1	Sublittoral coarse sediment	A5.11	Infralittoral coarse sediment in reduced salinity
				A5.13	Circalittoral coarse sediment
				A5.14	Deep circalittoral coarse sediment
		A5.2	Sublittoral sand	A5.21	Sublittoral sand in low or reduced salinity
				A5.27	Deep circalittoral sand
		A5.3	Sublittoral mud	A5.31	Sublittoral mud in low or reduced salinity
				A5.37	Deep circalittoral mud
		A5.4	Sublittoral mixed sediments	A5.41	Sublittoral mixed sediment in low or reduced salinity
				A5.45	Deep mided sediments
		A5.5	Sublittoral macrophyte dominated sediments	A5.52	Kelp and seaweed communities on sublittoral sediment
				A5.54	Angiosperm communitites in reduced salinity
		A5.6	Sublittoral biogenic reefs	A5.62	Sublittoral mussel beds on sediment
		A5.7	Features of sublittoral sediments	A5.72	Organically enriched or anoxic sublittoral habitats

Can be achieved using existing abiotic GIS layers	Requires habitat models made from biological data overlaid with the habitats from abiotic GIS data	Requires a spatial model of organic enrichment and anoxia overlaid with the habitats from abiotic GIS data

3 RESULTS & DISCUSSION

3.1 Mapping of seabed habitats

Bathymetry. The acoustic surveys in Pilot area 1 documented that both interferometric and multibeam sonars produce very detailed and reliable bathymetric data – multibeam in waters shallower than 50 m and interferometric sonar in water down to 180 m. An acceptable overall agreement was found between the multibeam data and diver ground-truthing with an average difference in depth of ~30 cm. The ground truthed depth measurements, when accurately executed, can be considered as a robust calibrator to the multibeam bathymetric results. In addition post-processing of multibeam data (shading techniques) allows for the identification of subtle aspects of the relief.

Substrate. The interferometric and sidescan sonar surveys in Pilot area 1 showed clearly that both instruments have the capacity to provide information on sediment texture, topography, bed forms, and other discrete objects at the seabed (e.g. boulders) and to certain extend also the degree of macroalgae coverage. The analyses of the sidescan sonar data demonstrated that a minimum diameter of particles in the order of 25 cm can be detected under optimal conditions. The acoustical detection limit in the magnitude of 25 cm is in agreement with the definition of 'boulders' – in the geological sense – as particles of a diameter above 256 mm (Wentworth 1922). For classification purposes a system is proposed using three new classes of reef types a box/cell size of 50 x 50 m, which is judged as the optimal size within which the hard substrate coverage most reasonable can be evaluated into broad scale habitat types on a harmonised set of acoustic/ground-truthed data.

The results from Pilot area 1 also stresses the need for ground-truthing of the acoustic data. Preferrably, validation should be based on diver paravane and spot dives supplemented by ROV (high-resolution camera or video) and van Veen samples. The results indicate that both biological and geological expertise should be onboard to do preliminary interpretations and adjust the survey programme. The sediment ground-truth results agreed considerably with the scattering map interpretation, except in cases of diver bias a thick vegetation cover on stones which rendered a low sidescan backscatter signal (> 15 % coverage).

NATURA 2000 habitats. For broad-scale mapping of habitats (>1 km2) the sidescan sonar combined with the multibeam echo sounder is considered to be the most cost-effective means of discriminating sediment types and dynamic processes. For small-scale habitat classification (> 1 km2), high-resolution sidescan sonar, underwater cameras or videos, and grab-sampling methods are considered to be the most appropriate mapping tool.

The interpretation of the sidescan mosaic and multibeam data from Læsø Trindel has demonstrated that the sandy and hard seabed unambiguously can be distinguished. However, the acoustic maps (e.g. the sidescan sonar mosaic) reflecting the variable acoustic properties of the seabed has to be 'ground truthed' by seabed sediment samples for calibrating the acoustic classes into sediment types and biological samples to characterise the flora and fauna present.

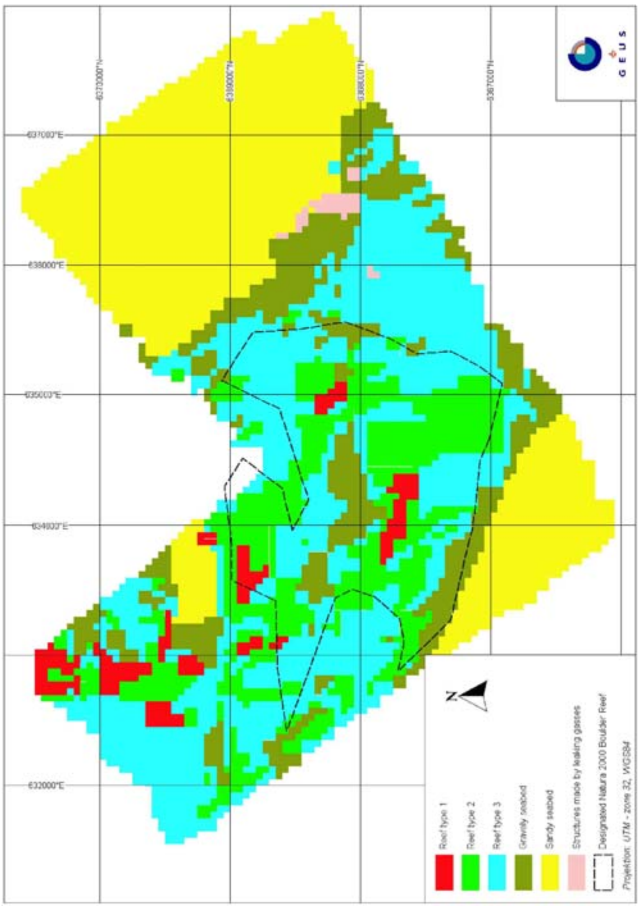


Figure 2. The seabed habitat classification map from the seabed mapping in Pilot area; Læsø Trindel. The present designated Natura 2000 boulder reef area (dashed line) covers only partly the newly mapped reef area.

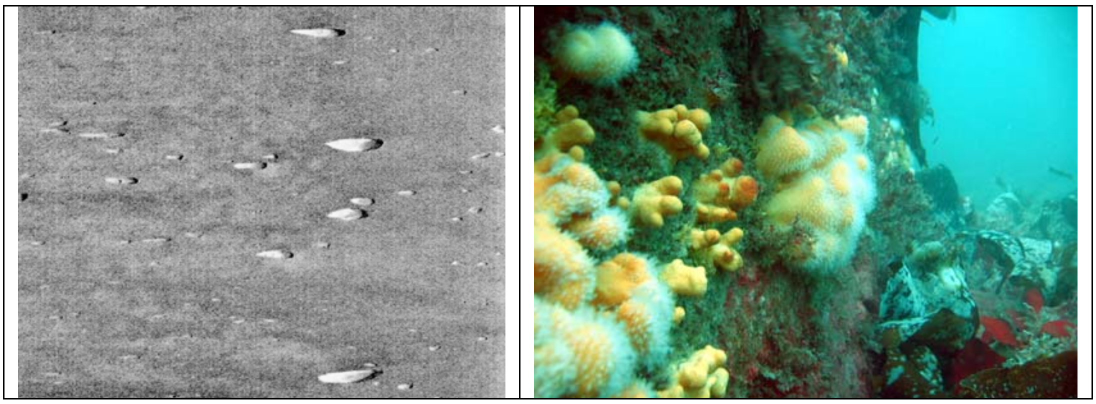


Figure 3. Habitat type reef (1170). To the left sidescan picture from Pilot area 1 showing scattered boulders on a flat, pre-dominantly gravely seabed. The largest boulders in the picture raise 3-4 m above the seabed. Approximate dimension of the sec-tions: Height 150 m and width 50m. To the right photo from the same area showing a stone wall with a variety of species (Photo: Jan Nicolaisen).

3.2 Multi-criteria evaluation

3.2.1 Baltic-wide benthic landscape map

The development of Baltic-wide broad-scale benthic landscapes succeeded in establishing the first delineation of individually distinct coarse-scale regions reflecting broad-scale species assemblages (Figure X). The results underline that marine landscape maps covering entire ecoregions are potentially strong tools providing a basis for a broad-scale spatial approach to the planning and management of the marine environment. The approach presented here is fully applicable for an ecologically relevant characterisation of the Baltic Sea. However, end users might find it necessary to validate and further refine and improve the maps. Such validations and refinements are necessary in order to fully exploit the potential application of the maps and for linking them to the implementation of national legislation, EU Directives and other policy documents such as the Baltic Sea Action Plan and the EU Maritime Policy.

Validation could be made either by carrying out an independent classification based entirely on a statistical approach or based on a classification with fuzzy logic acceptance criteria. The future success of producing marine landscape maps with a higher accuracy and precision and with information on ecological significance and sensitivity depends on access and availability of existing data as well as a transnational and cross-sectoral approach to this spanning the Baltic Sea. As such, the work presented in this report should be seen as a first step towards the broad-scale mapping of the marine landscapes in the Baltic Sea to be further developed by EU Member States for implementing EU maritime policy and legislation.

3.2.2 MCE models of NATURA 2000 habitats in Pilot area 3

The multi-criteria evaluation of NATURA 2000 habitats in the archipelago sea between Sweden and Finland also proved the potential for using MCE models with existing GIS data on key habitat drivers at a somewhat smaller scale. However, the results show that MCE are very sensitive to the quality of existing GIS data. Basic landscape data that outlines water and land have sufficient quality, as they are available in very detailed resolution (scale 1:20.000) that is needed to identify small habitats like small islands and lagoons, although maps in scale 1: 50.000 may well suite the purpose for most habitat modelling. Similarly, the data on wave exposure and land cover also have sufficient quality to be used in MCE modelling of NATURA 2000 habitat. However, better data on water flow from small rivers are wanted, as well as data on water quality. The main datasets missing to produce high-quality detailed habitat maps for the region are high resolution maps of the sediment characteristics and depth (e.g. from multibeam surveys). Besides resolution, the current depth and substrate data available has two major problems; a) There are quite large areas with very limited or no information available due to military restrictions and b) the shallow areas of 0-6 meter are not well outlined. A separation between large shallow inlets and bays and long narrow inlets are not done. Additionally, the mapped habitats give no information about the vegetation cover in the areas.

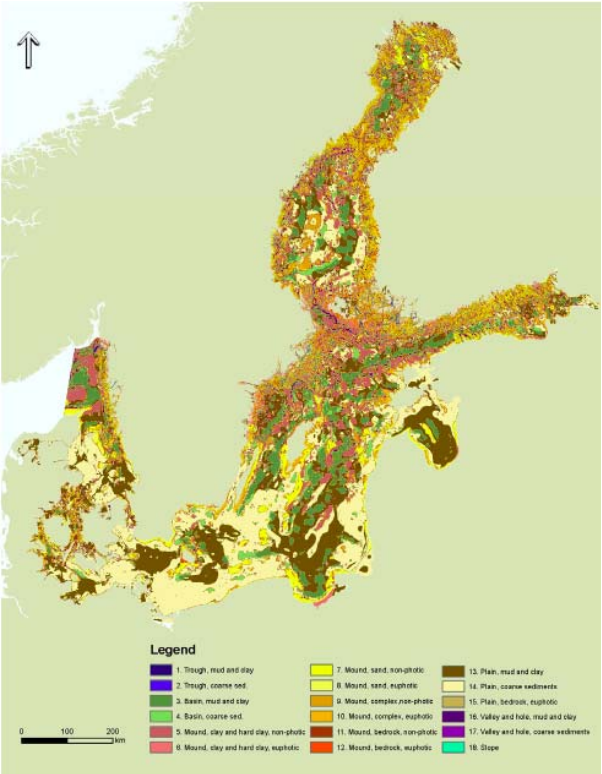


Figure 4. Topographic and bedform features identified during the preparation of the benthic marine landscape map of the Baltic Sea.

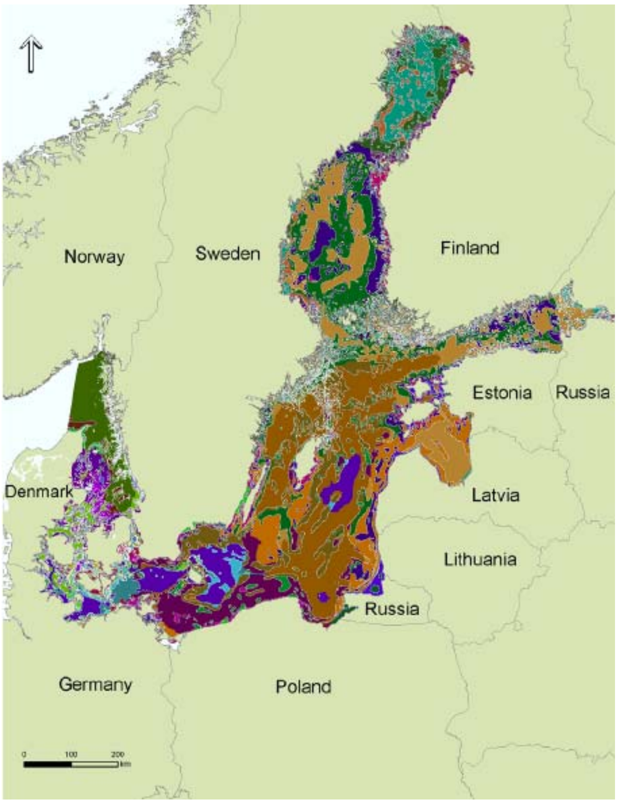


Figure 5. Benthic marine landscape map of the Baltic Sea using MCE modelling.

To get a more complete overview for management in Pilot area 3 these results could be complemented with models showing:

- Vegetation cover in shallow soft bottoms
- Vegetation cover around islands
- Shallow hard bottoms with zoned vegetation
- Deeper hard bottoms with zoned vegetation
- Deeper soft bottoms separated into sandy/muddy areas without oxygen deple tion
- Areas with high values for fish

3.3 Predictive modelling

3.3.1 Data requirements

The predictive habitat modelling activities of BALANCE have high-lighted some general requirements for the GIS datasets to be used in predictive spatial modelling of marine habitats in the Baltic Sea. Firstly, it is clear that predictive modelling should be preced by the design of a conceptual model outlining the potential regulating mechanisms before conducting a survey or utilising data from older surveys. This will enable a better and more conservative selection of environmental variables concentrating initially on the potentially most important ones (Guisan & Zimmerman 2000, Pont et al. 2005).

Secondly, it is important that the sampling design comprises large or entire parts of the environmental gradient that governs the distribution of the target species, habitat or assemblage in question. Many of the datasets collected in the Baltic monitoring programs do not meet this criterion. They are often, instead, targeted against the core areas of the distribution of the species. When producing habitat maps over large geographical areas, it is important to keep in mind that there may be regional differences in the species-environment relationship. If the statistical models are based only on a smaller, restricted area, biases may occur when applying the models to larger areas, especially if the environmental variables used as predictors are not primary drivers for the distribution of the target species. A trade-off in this respect is the tendency for the strength of the model predictions to decrease with increasing prediction area, thus demanding more precise and accurate descriptions when increasing prediction area.

Thirdly, it is important that the coverage of samples is relatively even and not too restricted to certain habitats. The latter is potentially a problem for some organism types that are only possible to sample in certain habitats. For example, many biological sampling methods are restricted either to soft or to hard bottom substrates. It may therefore be difficult to cover the potential distribution of species that are not closely associated with a certain bottom type.

Fourthly, it is, especially when considering the objectives of BALANCE project and the requirements of the work package 3 (connectivity and coherence) and 4 (MPA selection and network development), important that the explanation models can be used for predicting habitat distributions over larger spatial scales. This condition does not only affect the requirements of the field data but also, equally important, demands high resolution maps of the environmental variables that cover the whole areas of interest.

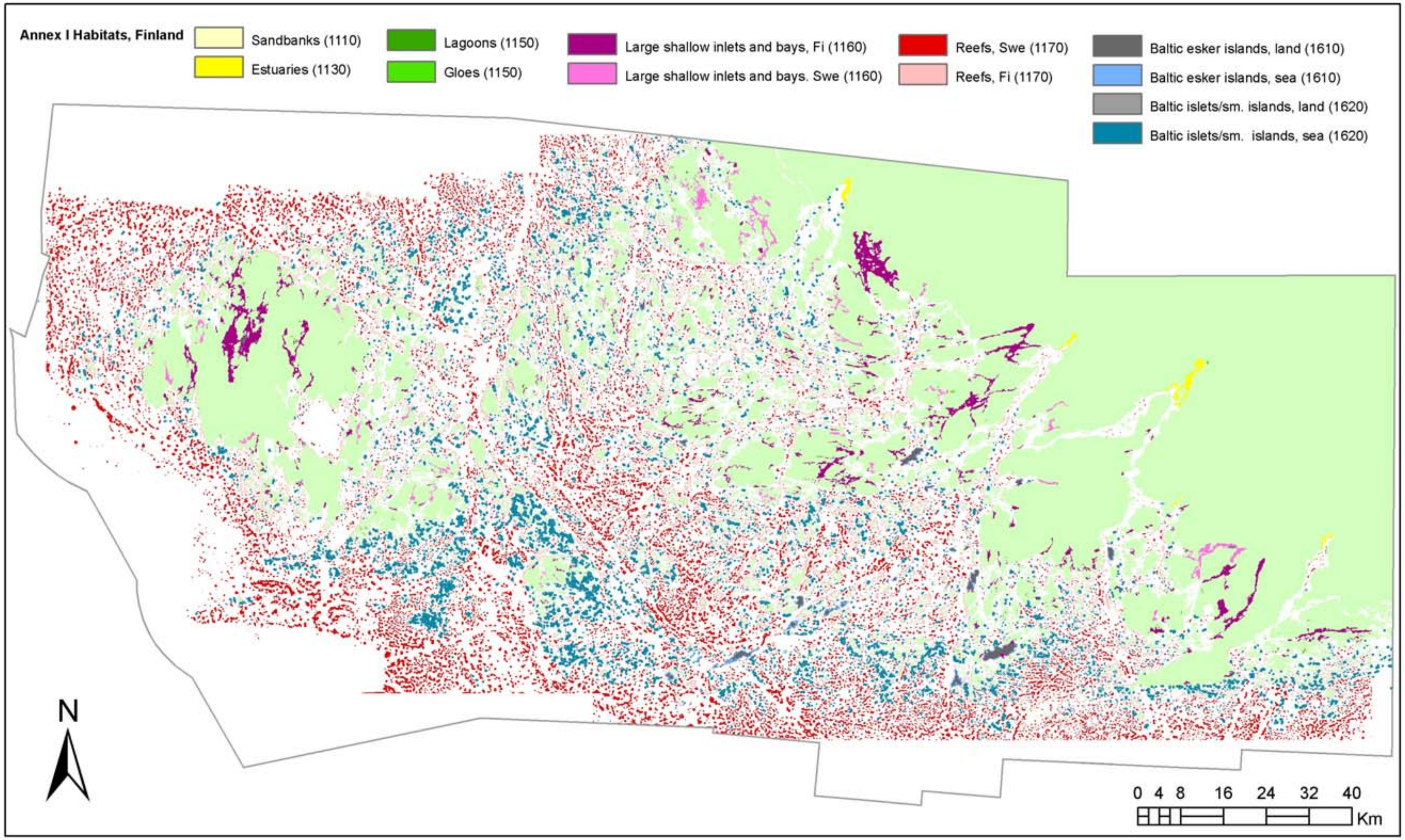


Figure 6. Results of the MCE model of NATURA 2000 Annex 1 habitats in the Finnish part of Pilot area 3.

3.3.2 Predictor variables

As mentioned above a common feature in the design of habitat models in BALANCE has been the application of conceptual models and model calibration using all available knowledge of the ecological requirements of the modelled species. This allowed for the careful selection of a few variables with known structuring functions in relation to the species' habitats. Although the case studies gave satisfactory results in terms of coarse-scale predictions of species distributions they indicated that some of the selected variables act as true habitat drivers while others play a minor role in shaping the habitat of the modelled species. In addition, the results of the case studies pointed at the needs for additional predictor variables, especially at the smaller scale, to establish more comprehensive habitat models.

Currently, a lack of high-resolution maps of for example bathymetry, surface sediments, hydrography, and in the case of young fishes, vegetation coverage, is limiting the production of accurate habitat maps. For bathymetry, this deficiency may be alleviated for example by opening access to classified maps, and by easing restrictions on collection and usage of bathymetric data. For other environmental variables, and for bathymetry in some areas, additional high-resolution mapping is needed. Development of new techniques, such as remote sensing for identification of coastal habitat characteristics (Bergström et al. 2007), as well as GIS-modelling techniques similar to those used within this BALANCE report, may provide efficient tools for producing high-resolution maps at reasonable costs.

The modelling of *Laminaria* in Pilot area 1 indicated that the significance of seabed curvature in the model reflect the importance of sub-strate, as curvature indicates presence of submarine elevations (here rocky bottom) vs. level bottom. The precision of benthic habitat models in this region is expected to increase as fine-scale substrate information becomes available. The surprisingly low significance of the two parameters slope and exposure appeared to be a bias, as field data failed to cover the whole gradient in the region.

Modelling of lobster habitat in Pilot area 1 showed that backscatter data on bottom substrates is a useful predictor variable when modelling this species, as it describes the bottom characteristic that is most important to the lobster distribution. However, the study suggested that the habitat model could be strengthened by adding data on local currents which relate directly to food supply for the species. It was further clear that for the model to accurately describe the variation of lobsters, it is necessary to cover the whole range of the environmental variables.

Modelling of the habitat to juvenile flatfish in Pilot area 1 indicated sampling problems; as trawling is not usable on several types of substrates, e.g. it does not sample efficiently in sea weed or eel grass beds, areas with patches of stones, or in muddy areas. Hence, potentially essential fish habitats may be not included in the final map. Another striking results of this case study was the fact that the three predictors showed great variability between response variables (species).

The 3-dimensional GIS study of suitable fish spawning areas in Pilot area 2 did not include spatial modelling. However, modelling was evaluated as a potential important supplement to be used to evaluate if environmental variables are randomly related to the spawning locations or if they present significant habitat choices.

In Pilot area 3 depth turned out as the most important factor in all models. Other important factors were distance to sandy shores and either exposure or the density of shoreline. With Mytilus the distance to submerged and emergent rocks was also significant, whereas aspect explained a large amount of de-viance in the algae model. The results from this case study also demonstrated that explanatory and predictive power may show different results. The Mytilus model got the highest score for explaining variance in the response variable as well as the lowest Generalized Cross Validation score, while the deviance explained by the angiosperm model was fairly low. However, the ROC plots showed that the models with best predictive capability were in fact the algae and angiosperm models, falling into the category 'excellent and 'outstanding', respectively.

The fish models in Pilot area 3 showed that the predictor variables chosen were important components when determining the habitats of the fish species, and that GIS-modelling could develop into an indispensable tool in large-scale mapping of essential fish habitats. Two of the predictor variables, wave exposure and the visibility proxy, which were completely GIS-derived and may be considered as indirect variables displayed the strongest explanatory power. For future modelling work it was suggested to use an alternative approach to attaining large-scale maps of turbidity by applying satellite imagery. A separate study showed that turbidity can be accurately interpreted from SPOT 5 images at a resolution of only 10 m (Bergström et al. 2007). In addition, including vegetation coverage as a predictor variable would most likely increase the predictive power of the models. Producing high-resolution, large scale GIS-layers of vegetation coverage would therefore constitute an important step towards increasing the precision of many fish habitat models.

The Fucus model i Pilot area 4 (Estonia) underlined the need for using key habitat drivers as predictor variables, and the problems associated with obtaining these data at a suitable scale. The Fucellaria model in Pilot area 4 (Lithuania) had relativily low predictive power caused by small-scale environmental heterogeneity. Point measurements of the depth at the observation sites (input data) do not provide information on local conditions of exposure (sheltered or not) caused by seabed elevations. This effect may also be captured by data at a finer spatial resolution of sediment data and using polygon based data for statistical model rather than point observations.

3.3.3 Model selection

The Akaike information criterion (AIC) has proven useful in selection of models and model complexes predicting habitat distribution in the Baltic Sea region. AIC estimates the Kullback-Liebler information lost by approximating full reality with the fitted model. This model selection approach involves terms representing lack of fit and a bias correction factor related to model complexity. AIC has a second order derivative, AICc, which contains a bias correction term for small samples size, and should be used when the number of free parameters, p, > n/40 (where n is sample size) (see details in Johnson & Omland 2004).

AIC has the advantage over maximizing fit, such as adjusted R^2 that do not consider model complexity, and thus always favours fuller models, while neglecting the principles of parsimony. In comparison with AIC, the commonly used null hypothesis tests, such as the likelihood ratio tests, compare pairs of nested models. The latter method has drawbacks, among others it cannot be used to quantify relative support among competing models (Johnson & Omland 2004).

3.3.4 Validation and ground truthing

Within the framework of MCE modelling and habitat classification in BALANCE it has not been possible to carry out validation nor ground truthing of the model results. Both statistical and GIS-based validation of the Baltic-wide marine landscapes are planned in the years to come. A wide range of validation techniques were applied for the habitat prediction models to evaluate the explanatory and predictive power of the models. There has been a rapid increase in available methods for species distribution modelling, and there is yet no consensus among the scientific community on how to best describe the potential and limitations of a model (Vaughan & Ormerod 2005). Many authors recommend the use of separate data sets when building and evaluating models (Chatfield 1995, Fielding & Bell 1997, McPherson et al. 2004, Vaughan & Ormerod 2005), although such an approach runs the risk of comparing sampling occasions or methods rather than model results. In BALANCE the predictions were not assessed by using independent samples. Whether or not new data has been used in validation, components of accuracy and level of generalization need to be specified to aid comparisons with different models and assess model usefulness in different situations (Carroll et al. 1999, Justice et al. 1999, Pearce & Ferrier 2000). It is not enough to specify only e.g. overall accuracy or sensitivity if end-users/managers are to draw appropriate conclusions about the usefulness and limitations of a model.

3.4 Classification of marine habitats

The attempt to make habitat classifications in line with the EUNIS classification in Pilot area 3 gave varied results. Several problems presented themselves. Presently, the Baltic Sea area is poorly represented in the EUNIS system. The Baltic classes do not follow the hierarchical structure of the classification very well and are somewhat inconsistent to what is included on each level. Improvements to the classification are surely needed. The Baltic has several gradients that do not play a significant role in the truly marine environment from where EUNIS originates. The most obvious differences are the lack of tides, the salinity gradient, ben-thic substrate complexity, and the enclosed nature of the sea.

The lack of tides means there is a narrow or no intertidal zone (<0.5 m). However, some of the species found in the intertidal zone on marine shores, form a similar zonation subtidally in the Baltic. This is currently not laid out in the existing EUNIS hierarchy. In the Archipelago Sea salinity changes from almost freshwater in the innermost archipelago and near river mouths to approximately 5-7 psu where it joins the Baltic Proper. On the scale of the whole Baltic Sea the salinity gradient is much larger, from 0 to 34 psu. The enclosedness limits fetch and consequently wave exposure. Although wave exposure in the Baltic may be small compared to Atlantic shores, the variation within the Baltic plays an important role in structuring communities.

In general the modelled layers satisfy the needs for large scale planning of the coastal sea. All maps show the potential occurrence of the habitats and can be used to derive habitat complexity maps, estimate the proportion of protected versus unprotected areas of the habitats and can be used as a first selection of areas of interest for more detailed surveys. The Natura 2000 habitats do not cover deeper habitats or shallow hard bottom habitats that may have high nature value.

In the future there is a need to recalibrate wave exposure specifically for the Baltic Sea area. There is also a need to create true classes based on the flora and fauna communities and the special abiotic factors at play in the Baltic Sea area, to be incorporated into the hierarchical structure of EUNIS.

3.5 Application of habitat maps for management

Although splitting nature into man-made categories is not the most accurate representation of nature, it is often necessary for management purposes. The maps produced using this methodology, will give a basic view of the types of habitats that are found in an area. The ecological considerations related to these habitats can be inferred from existing knowledge. In nature habitats are never static, and the habitats depicted in the maps may have seasonal or multi-annual cycles (e.g. annual algae, bottom fauna on soft bottoms), but if these are acknowledged and included in decision making, the maps can be a good addition to the sustainable management of marine areas and to marine conservation.

It is important that end users are aware of the inherited limitations of the developed marine habitat maps:

- The resultant maps are no better than the information on which they were developed. For some areas data are scarce and/or only available in low resolution with large distances between points with actual data. The maps are thus not suitable for fine scale planning unless further improved.
- Due to the relatively coarseness of most of the data available for the mapping and modelling exercises it has not been possible to identify fine-scale features and the resultant maps only present the most dominant features.
- The maps should be regarded as beta-versions, which need further refinement and validation before they are applied in spatial planning. Most important is the validation of the MCE-based habitat maps and correction for auto-correlation in the habitat prediction models as well as ground-truthing of all maps.

Despite these limitations the results of the habitat mapping and modelling were very encouraging and successfully demonstrated the potential to develop basin-wide coarse-scale maps of benthic and pelagic habitat maps on the basis of few key variables. The Baltic-wide models of topographic and benthic landscapes represent highly needed datasets for the implementation of the EU Habitats Directive in the Baltic Sea. These maps are also expected to become highly valuable for the succesful launch of ecosystem-based management in the region, which will require a basemap on structural habitat entities. At the sub-regional level the modelling results are expected to provide a useful tool in developing integrated solutions for nature conservation and sustainable fisheries, coastal development, transport and other sea uses. As an example the modelling of

Norwegian lobster habitat in Skagerrak may be used in the management of the lobster fishery for information on where important lobster habitat can be found. Similarly, the 3-dimensional modelling of spawning areas to Baltic cod and sprat can be used in the characterization of the spatial and temporal variability of eastern Baltic cod spawning habitats in the light of implemented closed areas to ensure undisturbed spawning. The usefulness of the BALANCE models and the examples for habitat modelling applications they provide is also stressed by the high demand for detailed maps of essential habitats for fish, marine mammals and other top predators. The habitat maps presented in this report are already used by several regional authorities, for example in fisheries restoration and management plans and in the design and zonation of forthcoming MPAs.

4 RECOMMENDATIONS

The following recommendations are directed at policymakers, scientists and environmental managers for the future refinement of the modelling and mapping of marine habitats with the long-term goal of a supporting a sustainable development in the Baltic Sea Region through an informed trans-national approach to the management of the marine ecosystem.

4.1 Data and methodology

The following recommendations are made in regard to marine information issues within the Baltic Sea region:

1. Geo-morphological data on land features, e.g. the coast line, exists and are accessible at both fine and coarse scales. These data are highly useful for mapping of several Natura 200 Annex 1 habitats.

2. Data models on a range of envirionmental variables, such as light attenuation, oxygen levels, salinity, temperature, are available and should be used when improving models for mapping the distribution of species habitats.

3. Bathymetric data exists at a fine scale from multiple areas, but are often not accessible due to e.g. military restrictions. Effort should be made to retrieve data from the areas, where they do exist. In areas where such data are lacking, effort should be made to collect new data. Even at a course scale, e.g. in the form of slope retrieved from nautical charts, have proven highly valuable for mapping of Natura 2000 Annex 1 habitats and for modelling and mapping of habitats of several marine species.

4. Substrate data exists at a coarse scale (grid size of 1-2 km) from most areas, while fine scale data (grid cell of 10-50 cm) exists only from a few areas. Effort should be made to collect such data covering the entire Baltic Sea areas. When available, these data have proven highly useful for mapping and modelling of most plants and animal species.

5. GIS analyses are appropriate for mapping of Natura 2000 “habitats” using physiographic and geological features.

6. Predictive modelling, for example using GAM, is a cost-effective way of developing fine grained, large extent distribution maps of marine habitats of species and species assemblages.

7. It is highly important that the biological data cover the entire gradient and extent of the environmental variables!

8. At present, modelling of habitats in time requires detailed knowledge of the limits of environmental variables at the species level.

4.2 A Stepwise approach to mapping of marine habitats in the Baltic Sea area

1. GIS analyses and mapping of the NATURA 2000 Annex 1 habitats that equal landscapes, as well as analyses and mapping of other marine landscapes.

2. Detection of remaining NATURA 2000 Annex 1 habitats by acoustic and other methods.

3. Sampling of acoustic data of fine grain (multibeam and back scatter, resolution of 1-10 cm) and validation using samples, UW video/photographs, and/or diving observations.

4. Sampling and collation of additional variables, essential for NATURA 2000 Annex 1 habitats. For the OSPAR, HELCOM and essential top predator habitats variables driving ecological significance will be necessary.

5. Predictive modelling of habitats, particularly on habitat-building species of structural importance and species of high ecological significance. Validation of models.

6. Integration of habitat model results and existing marine landscapes in GIS to create a habitat classification system for the Baltic Sea region, with classes of discrete boundaries.

7. Predictive modelling of spatial and temporal variation of each classified habitat.

Application for planning of MPA network, incl. of the NATURA 2000 network:

8. Apply in GIS to create areas of appropriate size and numbers, as well as blue corridors.

9. Use the above data to evaluate the best possible solutions of a marine protected areas network of habitats and corridors, which maintain ecological functioning.

Application for management and evaluation of ecological status of MPA network, incl. of the NATURA 2000 network:

10. Assess the resilience and sensitivity of each classified habitat in relation to major sea uses.

4.3 Confidence of the habitat maps

The following recommendations are made in regard to habitat mapping issues within the Baltic Sea region:

1. The maps of Baltic-wide benthic marine landscapes and some of the selected Natura 2000 Annex 1 habitats have not yet been validated. The majority of the maps of species habitats from the predictive modelling have been validated, however it is recommended to test the predictive power of the models on inde-

pendent data and to take account of auto-correlation effects. All maps need ground-truthing prior to application for management.

2. The current habitat maps are not to be regarded as all inclusive and full coverage, but rather pieces of the Baltic Sea marine patchwork. Also, the maps are products that demonstrate the value of using different methods for mapping of marine habitats.

4.4 Application for nature conservation

Habitat maps are a pre-requisite for nature conservation management. While Natura 2000 Annex 1 habitats are legal entities, many of them are not ecosystem based. Thus, maps of species habitats are needed to evaluate ecosystem functioning of the Natura 2000 network at sea. Development of Baltic-wide habitat maps of ecological relevance requires a co-ordinated approach. The following recommendations are made in regard to nature conservation issues within the Baltic Sea area:

3. Harmonisation of habitat characteristics of both Natura 2000 Annex 1 habitats and of species habitats, and their relation to marine landscapes. If a classification system is developed, it should be coherent with systems of adjacent seas, e.g. the EUNIS system.

4. Development of guidelines for a common approach to modelling and mapping of marine habitats. The guidelines should be in line with those developed for adjacent seas, e.g. as part of the MESH project.

5. To increase the cost-efficiency, harmonisation of data collection and monitoring methods should be made to the extent possible without loosing valuable information needed for other purposes. Supplementary surveys should be made for the specific purpose of mapping of marine habitats and ecosystem function.

6. Within the framework of BALANCE it was not possible to integrate information and models of key processes and ecological significance into the characterisation of Baltic marine landscapes. Therefore, the marine landscape classifications should be seen as qualitative, and future studies will be needed to resolve the linkages between landscapes and ecological functioning.

4.5 Application for marine spatial planning

The following recommendations are made in regard to marine spatial planning issues within the Baltic Sea:

1. Maps of habitats and their connectivity form the basis for ecosystem-based marine spatial planning and management. Thus, full coverage habitat maps should be developed for the entire Baltic Sea area. This could be done in a patchwork approach, where new, validated and ground truthed maps are added to the master-map as they become available.

2. Further, BALANCE focused on the establishment of landscape maps covering mainly benthic habitats, and thus future landscape maps covering all habitats need to be developed to offer end users a fully comprehensive mapping system for the Baltic Sea.
3. In relation to sectoral development planning it is important to stress the need to transfer the habitat and landscape maps to sensitivity maps displaying the degree of resilience and vulnerability of the habitats and landscapes in relation to potential perturbations associated with each sectoral use of the sea.

5 ACKNOWLEDGEMENTS

The BALANCE partnership is grateful for the support received from partners in the MESH project in methodology on how to classify and map marine habitats of the Baltic Sea as well as marine institutes and data holders in the Baltic Region who kindly has provide access to relevant environmental data.

BALANCE is also grateful for the original support of the Nordic Council of Ministers who funded the Nordic Workshop on Marine Protected Areas enabling John Roff (University of Guelph, Canada), Jon Day (Great Barrier Reef Marine Park Authority, Australia), Jon Davies (the Joint Nature Conservation Committee, UK) and John Baxter (Scottish Natural Heritage, Scotland) to inspire the workshop participants into formulating the Nordic project NorGIS, "Geographic analysis for mapping and modelling and coastal and marine habitats". The NorGIS project made the application for the BALANCE project possible, incl. the applicability of marine habitat models and maps in environmental management.

Specifically, we are grateful of the support received from:

- The BALANCE Steering Committee consisting of David Connor from the Joint Nature Conservation Committee UK (Chair), Juha Markku Leppanen from the HELCOM Secretariat and Jacek Zaucha VASAB 2010 for providing valuable advice throughout the project period.

- The staff at the BSR INTERREG IIIB Secretariat in Rostock for their always available and professional advice throughout the project period.

- In regard to providing access to data, models and results we are grateful for the support received from **NN at the **XX,.

- ** for providing photos illustrating marine life and habitats.

6 REFERENCES

Aaltojen alla, 2006. A web site describing the Baltic Sea and its underwater life. Accessed on 7.2.2006 at http://www.aaltojenalla.fi/

Allouche, O., Tsoar, A., Kadmon, R. 2006. Assessing the accuracy of species distribution models: prevalence, kappa and the true skill statistic (TSS). Journal of Applied Ecology 43:1223-1232.

Andersen, J. H., Pawlak, J. 2006. **

Andrushaitis A, Seisuma Z, Legzdina M, Lenshs E. 1995. River load of eutrophying substances and heavy metals into the Gulf of Riga. In: Ojaveer E (ed) Ecosystem of the Gulf of Riga between 1920 and 1990. Estonian Academy Publishers, Tallinn, pp 32–40.

Anonymous 1992. EU HD. 92/43/EEC

Anonymous 1999. **European Comission, dg Environment 1999: Interpretation manual of European Union Habitats. EUR 15/2

Anonymous 2001. DN-håndbok Kartlegging av marint biologisk mangfold, Report 19-2001, ** pp.

Anonymous 2006. **Coastal Guide to Europe, 2006. Archipelago Sea Biosphere Reserve, a web page of the Coastal Guide to Europe. Accessed on 7.2.2006 at http://www.coastalguide.to/archipelago/main.html.

Anonymous **Swedish environmental protection agency 2006: Sammanställning och analys av kustnära undervattenmiljö. Report no. 5591 (in Swedish)

Anonymous ** The county administrative board of Stockholm 1991: Trösklade havsvikar. Rapport nr 1991:9 (in Swedish)

Anonymous **The county administrative board of Stockholm 2003: Skyddsvärda grundområden I Svealands kärgårdar. Rapport nr 2003:05 (in Swedish)

Aps, R. 1989. Sprat stock dynamics in the Northern Baltic, 1950-1987. Rapp. P. -v. Réun. Cons. int. Explor. Mer 190: 219-222.

Astok V, Otsmann M, Suursaar Ü (1999) Water exchange as the main physical process in semi-enclosed marine systems: the Gulf of Riga case. Hydrobiologia 393: 11–18

Austin, M. P. 2002. Spatial prediction of species distribution: an interface between ecological theory and statistical modelling. Ecological Modelling 157:101-118.

Axelsson, S. 2003: Kartering av vissa kustbiotoper som utpekas i EU:s habitatdirektiv. Rapport för Naturvårdsverket. Metria Miljöanalys.

Aaltojen alla, 2006. A web site describing the Baltic Sea and its underwater life. Accessed on 7.2.2006 at http://www.aaltojenalla.fi/.

Aarup, T. 2002. Transparency of the North Sea and Baltic Sea – a Secchi Depth data mining study. *Oceanologia* 44(3): **-**.

Bagge, O. and Thurow, F. 1994. The Baltic cod stock: fluctuations and possible causes. ICES mar. Sci. Symp. 198: 254-268.

Backer, H., Leinikki, J., Oulasvirta, P. 2004. Baltic Marine Biotope Classification System (BMBCS) – definitions, methods and EUNIS compatibility. Alleco Ltd., Technical Report. May 2004, 49 pp.

Bailey, N., Chapman, C.J., Kinnear, J., Bova, D. and Weetman, A. (1993). Estimation of *Nephrops* stock biomass on the Fladen Ground by TV survey. ICES-CM K:34

Barnes, **., Hughes, **. 1999.

Begon, Harper, Townsend, **. 2001.

Bekkby, T., Eriksstad, L., Bakkestuen, V., Bjørge, A. 2002. A landscape ecological approach to coastal zone applications. *Sarsia* 87: 396-408.

Bergstrøm, S., Carlsson, 1994. River runoff to the Baltic Sea: 1950-1990. Ambio 23 (4-5): 280-287.

Bergström, U., Evertson, J., Karås, P., Sandström, A., Sundblad, G. and Wennberg, S. 2007. Evaluation of satellite imagery as a tool to characterise shallow habitats in the Baltic Sea. Balance interim report no. 5.

Berzinsh V (1995) Dynamics of hydrological parameters of the Gulf of Riga. In: Ojaveer E (ed) Ecosystem of the Gulf of Riga between 1920 and 1990. Estonian Academy Publishers, Tallinn, pp 8–31

Bio, A.M.F., Alkemade, R., Barendregt, A. (1998). Determining alternative models for vegetation response analysis: a non parametric approach. J. Veg. Sci. 9, 5 _/16.

Boyd, **. et al. 2006.

Breman, **. 2002.

Brinkman, A. G., Dankers, N., Stralen, M. v. 2002. An analysis of mussel bed habitats in the Dutch Wadden Sea. *Helgol Mar Res* 56: 59-75.

Bryan, K. 1969. A numerical method for the study of the circulation of the world ocean. J. Phys. Oceanogr., 15: 1312-1324.

Buckling A., Kassen R., Bell G., Rainey P.B., 2000, *Disturbance and diversity in experimental microcosms*, Nature, 408, 961–964.

Carroll, C., W. J. Zielinski, and R. F. Noss. 1999. Using presence-absence data to build and test spatial habitat models for the fishes in the Klamath region, U.S.A. Conservation Biology 13:1344-1359.

Cato, I., Kjellin, B. och Zetterlund, S. 2003: Förekomst och utbredning av sandbankar, berg och hårdbottnar inom svenskt territorialvatten och svensk ekonomisk zon (EEZ).SGU Rapport 2003:1. Sveriges Geologiska Undersökning (SGU). Uppsala.

Chapman, C. J., and Bailey, N. (1987). Biological research on fish and shellfish stocks. Recent progress in Norway lobster research. *In* Developments in fisheries research in Scotland, pp. 99–111. Ed. by R. S. Bailey and B. B. Parrish. Fishing News Books Ltd, Farnham, England.

Chatfield, C. 1995. Model uncertainty, data mining and statistical inference. Journal of the Royal Statistical Society, Series A 158:419-466.

Codyssey (2007) Cod spatial dynamics and vertical movements in European waters and implications for fishery management. Final report (EU contract Q5RS-2002-00813).

Collie, **. et al. 2004.

Connor, D. W., Allen, J. H., Golding, N., Lieberknecht, L. M., Northern, K. O., Reker, J. B. 2003. The National Marine Habitat Classification for Britain and Ireland, Version 03.02 Copyright JNCC, Peterborough ISBN 1 86107 546 4. Available at: http://www.jncc.gov.uk/MarineHabitatClassification

Connor, D. W., Golding, N., Robinson, P. Todd, D., Verling, E. 2007a. *UKSeaMap: The mapping of marine seabed and water column features of UK seas*. Consultation report. Joint Nature Conservation Committee, Peterborough, ** pp.

Connor D. W. et al. **. 2007b. Mapping of marine habitats. MESH report , ** pp.

Cox, M.D., 1984. A primitive equation 3-dimensional model of the ocean.GFDL Ocean Group Tech. Rep. No. 1, GFDL/Princeton University.

Christensen, O., Totland, O., Olsen, H. 2006: Bunntypekartlegging i forbindelse med bruk og vern av sjøområdene mellom Rauøy og Hvaler, Østfold. NGU Rapport 2006.035, 26 s. + 4 kart.

Dahl, K., Lundsteen, S. & Helmig, S., (2003): Stenrev – havets oaser. *Miljøbiblioteket. Gads Forlag*. ISBN 87-12-04019-3.

Danielsson A, Carman R, Rahm L, Aigars J (1998) Spatial estimation of nutrient distributions in the Gulf of Riga sediments using cokriging. Estuar Coast Shelf Sci 46: 713–722.

Darwin, C. 1859. **

Davies, C., Moss, D. 2004. EUNIS habitat classification. Marine habitat types: Revised classification and criteria, September 2004. *European Environmental Agency. September 2004*,82 pp.

Day, J. C., Roff, J. C. 2000. Planning for Representative Marine Protected Areas: A framework for Canada's Oceans. *Report prepared for World Wildlife Fund Canada, Toronto*, ** pp.

DeLeo, J. M. 1993. Receiver operating characteristic laboratory (ROCLAB): Software for developing decision strategies that account for uncertainty. Pages 318-325 in Uncertainty Modeling and Analysis, 1993. Proceedings., Second International Symposium on, College Park, MD, USA.

den Hartog, C., 1970. The Seagrasses of the World. North Holland Publ. Co, Amsterdam. 275 pp.

Denny, M. W., B. Helmuth, G. H. Leonard, C. D. G. Harley, L. J. H. Hunt, and E. K. Nelson. 2004. Quantifying scale in ecology: lessons from a wave-swept shore. Ecological Monographs **74**: 513–532.

Diaz, **. et al. 2006

Dickson, R., and Brander, K. 1993. Effects of changing wind field on cod stocks of the North Atlantic. Fish. Oceanogr. 2: 124-153.

Erlandsson, C., Lindeberg, G. 2007. Harmonising marine geological data with EUNIS classification. *BALANCE Interim Report No.* 13.

Fielding, A. H., and J. F. Bell. 1997. A review of methods for the assessment of prediction errors in conservation presence/absence models. Environmental Conservation 24:38-49.

Fenchel, T. 2006. **

Ferguson, R.L., Korfmacher, K. 1997. Remote sensing and GIS analysis of seagrass meadows in North Carolina, USA. *Aquatic Botany* 58: 241 - 258.

Flöder S., Sommer U., 1999, *Diversity in planktonic communities: An experimental test of the intermediate disturbance hypothesis*, Limnol. Oceanogr., 44, 1114–1119.

Fossum, P. 1986. A staging system for larval cod (*Gadus morhua* L.). Fiskeridir. Skr. (Havunders.) 18: 69-76.

Francis, M. P., Morrison, M. A., Leathwick, J., Walsh, C., Middleton, C. 2005. Predictive models of small fish presence and abundance in northern New Zealand harbours. *Estaurine, Coastal and Shelf Science* 64: 419-435.

Franklin, J. (1998). Predicting the distribution of shrub species in southern California from climate and terrain-derived variables. J. Veg. Sci. 9, 733_/748.

Garza-Pérez, J. R., Lehmann, A., Arias-González, J. E. 2004. Spatial prediction of coral reef habitats: integrating ecology with spatial modeling and remote sensing. *Mar Ecol Prog Ser* 269: 141-152.

Gibson, R.N., 1994. Impact of habitat quality and quantity on the recruitment of juvenile flatfishes. Neth. J. Sea Res. 32, 191-206.

Gislén, T. 1930. **

Grauman, G.B. and Yula, E. 1989. The importance of abiotic and biotic factors in early ontogenesis of cod and sprat. Rapp. P-v Réun. Cons. Int. Explor. Mer 190: 207-210.

Guisan, A., T. C. Edwards Jr., and T. Hastie. 2002. Generalized linear and generalized additive models in studies of species distributions: setting the scene. Ecological Modelling 157:89-100.

Guisan, A., and W. Thuiller. 2005. Predicting species distribution: offering more than simple habitat models. Ecology Letters 8:993-1009.

Guisan, A., S. B. Weiss, and A. D. Weiss. 1999. GLM versus CCA spatial modeling of plant species distribution. Plant Ecology 143:107-122.

Guisan, A., and N. E. Zimmerman. 2000. Predictive habitat distribution models in ecology. Ecological Modelling 135:147-186.

Gutt, J., and D. Piepenberg. 2003. Scale-dependent impact on diversity of Antarctic benthos caused by grounding of icebergs. Marine Ecology Progress Series **253**: 77–83.

Hansen, J.L.S., Josefson, A.B., Pedersen, T.M. 2004. Genindvandring af bundfauna efter iltsvindet 2002 i de indre danske farvande. Danmarks Miljøundersøgelser. *Faglig rapport fra DMU* 506: 61 pp. Available at: http://www2.dmu.dk/1_Viden/2_Publikationer/3_Fagrapporter/rapporter/fr506.pdf

Hastie, T.J., Tibshirani, R.J. (1990). Generalized Additive Models. Chapman & Hall, London.

Heip, **. 2006.

HELCOM (1996) Third periodic assessment of the state of the marine environment of the Baltic Sea. 1989–1993: Background document. Balt Sea Environ Proc 64B.

Herkül, K., Kotta, J., Kotta, I., Orav-Kotta, H. 2006. Effects of physical disturbance, isolation and key macrozoobenthic species on community development, recolonisation and sedimentation processes. Oceanologia, 48S, 267–282.

Hermansen and Jensen, (2000): Digitalt kort over havbundssedimenter omkring Danmark 1:500.000. *Danmarks og Grønlands Geologiske Undersøgelse Rapport 2000/68.*

Hilty, J. A., Lidicker Jr, W. Z., Merenlender, A. M. 2006. Corridor Ecology. The science and practice of linking landscapes for biodiversity conservation. Island Press, London, pp 323.

Hinrichsen, H.-H., Lehmann, A., St. John, M.A. Brügge, B. 1997. Modelling the cod larvae drift in the Bornholm Basin in summer 1994. Cont. Shelf Res., 17 (14): 1765-1784.

Hiscock, **. 1994.

Holmberg, J., Nyberg, J., Lindberg, T., Zetterlund, S., Elhammer, A. 2007. The Balance Data Portal: Background, documentation and perspectives. *BALANCE Interim Report* No. 17, ** pp.

Holthuis, L.B. (1991). FAO species catalogue. Vol. 13. Marine lobsters of the world. An annotated and illustrated catalogue of species of interest to fisheries known to date. FAO Fisheries Synopsis. No. 125, Vol. 13. Rome, FAO. 1991. 292 p.

Häkkinen, A., 1990: Saaristomeren vedenalaisten maa-ainesvarojen kartoitus Gullkronan selällä 1989. Varsinais-Suomen seutukaavaliitto, Turku. 58 p. In Finnish.

Hällfors, G., Leskinen, E., Niemi, A. 1983. Hydrography, chlorophyll and nutrients at Tvärminne Storfjärd, Gulf of Finland, in 1979/80. *Sci. Reps*. Walter and Andree de Nottbeck Foundation, Helsinki 4: 1-19.

ICES 2006. Report of the Baltic Fisheries Assessment Working Group. ICES CM 2006/ACFM:18.

Isæus, M. 2004. *Factors structuring* Fucus *communities at open and complex coastlines in the Baltic Sea*. PhD Thesis, Dept. of Botany. Stockholm, Sweden, Stockholm University**,** 40 pp.

Isæus, M. 2004: A GIS-based wave exposure model calibrated and validated from vertical distribution of littoral lichens. In thesis: Factors structuring Fucus communities at open and complex coastlines in the Baltic Sea. Doktorsavhandling vid Botaniska Institutionen, Stockholms Universitet: Stockholm.

Isæus, M., Carlén, I., Wibjörn, C., Hallén, S. 2007. *Svenska högarna. Marinbiologisk kartläggning och naturvärdesbedömning*, Stockholm, Stockholm Administrative County Board**:** 50.

Jamieson, A., and Otterlind, G. 1971. The use of cod blood protein polymorphisms. Rapp. P.-v. Réun. Cons. Int. Explor. Mer 161: 55-59.

Jansson, B.O. 1980. Natural systems of the Baltic Sea. *Ambio* 9: 128-136.

Jevrejeva S (2000) Long-term variability of sea ice and air temperature conditions along the Estonian coast. Geophysica 36: 17–30.

Johnson J. B., Omland K. S. 2004. Model selection in ecology and evolution. *TRENDS in Ecology and Evolution* 19: 101-108.

Johnston, T. H. 1924. The relation of climate to the spread of prickly pear. Transactions of the Royal Society of South Australia 48:269-295.

Jones, **. 1950.

Josefson, A. B., Hansen, J. 2004. **

Justice, A. C., K. E. Covinsky, and J. A. Berlin. 1999. Assessing the generalizability of prognostic information. Annals if internal medicine 130:515-524.

Kangas, P., H. Autio, G. Hällfors, H. Luther, Å. Niemi & H. Salemaa, 1982. A general model of the decline of *Fucus vesiculosus* at Tvärminne, south coast of Finland in 1977–1981. Acta Botanica Fennica 118: 1–27.

Karås, P., and R. Hudd. 1993. Reproduction areas of fresh-water fish in the Northern Quark (Gulf of Bothnia). Aqua Fennica 23:39-49.

Kautsky, H. 1988. Factors structuring phytobenthic communities in the Baltic Sea. Akademitryck, Edsbruk: 1-30.

Kautsky, H. 1995. Quantitative distribution of sublittoral plant and animal communities in the Baltic Sea gradient. In Elefteriou, A., A. Ansell & C. Smith Jr. (eds), Biology and Ecology of Shallow Coastal Waters. Olsen & Olsen, Fredensborg: 23-31.

Kautsky, H., Maarel, E. van der 1990. Multivariate approaches to the variation in phytobenthic communities and environmental vectors in the Baltic Sea. *Mar. Ecol. Prog. Ser.* 60: 169-184.

Kautsky, U. and H. Kautsky, 1995. Coastal productivity in the Baltic Sea. In Eleftheriou, A., A. D. Ansell & C. J. Smith (Eds), Biology and Ecology of Shallow Coastal Waters. Olsen & Olsen, Fredensborg: 31–38.

Kenny, A. J., Cato, I., Desprez, M., Fader, G., Schüttenhelm, R. T. E., and Side, J., (2003):

An overview of seabed-mapping technologies in the context of marine habitat classification.

ICES Journal of Marine Science, 60: p. 411–418.

Keynäs, ** (ed.). 2001. Kystbiotoper i Norden. Hotade och representative biotoper. *TemaNord 2001*, No. 536, ** pp.

Kiirikki, M. 1996a. Dynamics of macroalgal vegetation in the northern Baltic Sea – evaluating the effects of weather and eutrophication. Walter and Andrée de Nottbeck Foundation Scientific Reports.12: 1-15.

Kiirikki, M. 1996b. Experimental evidence that *Fucus vesiculosus* (Phaeophyta) controls filamentous algae by means of whiplash effect. Eur. J. Phycol. 31: 61-66.

Kiirikki, M., Ruuskanen, A. 1996. How does *Fucus vesiculosus* survive ice scraping. – Botanica Marina 39: 133-139.

Killworth, P.D., Stainforth, D., Webbs, D.J., Paterson, S.M. 1991. The development of a free-surface Bryan-Cox-Semtner ocean model. *J. Phys. Oceanogr*. 21: 1333-1348.

Kristiansen, **., Greve, **. 2003. GIS interpolation. Kotta, J., V. Lauringson, and I. Kotta. 2007. Role of functional diversity and physical environment on the response of zoobenthos communities to changing eutrophication. Hydrobiologia, 580, 97–108.

Kotta, J., Torn, K., Martin, G., Orav-Kotta, H., Paalme, T. 2004. Seasonal variation of invertebrate grazing on *Chara connivens* and *C. tomentosa* in Kõiguste Bay, NE Baltic Sea. Helgoland Mar. Res., 58, 71–76.

Kändler, R. 1944. Untersuchungen über den Ostseedorsch während der Forschungsfahrten mit dem R.F.D. „Poseidon" in den Jahren 1925-38. *Ber. dt. wiss. Komm. Meeresforsch. N.F.* 11: 137-245.

Köster, F.W., Hinrichsen, H.-H., Schnack, D., St. John, M. A., MacKenzie, B. R., Tomkiewicz, J., Möllmann, C., Kraus, G**.**, Plikshs, M., Makarchouk, A., Eero, A. 2003. Recruitment of Baltic cod and sprat stocks: identification of critical life stages and incorporation of environmental variability into stock-recruitment relationships. *Sci. Mar.* 67: 129-154.

Laffoley, D. d'A., Baxter, J., Bins, T., Bradley, M., Connor, D.W., Hill, M., Tasker, M.L., Vincent, M.A. 2000. An implementation framework for conservation, protection and management of nationally important marine wildlife in the UK. Prepared by the statutory nature conservation agencies, EHS and JNCC for the DETR working group on the Review of Marine Nature Conservation. Peterborough, *English Science Report* 394, ** pp.

Laier, T., Jørgensen,. N.O., Buchardt, B., Cederberg, T. & Kuijpers, A., (1992): Accumulation and seepage of biogenic gas in northern Denmark. *Cont. Shelf Research, Vol.12*, p. 1173-1186.

Laier, T., Kuijpers, A., Dennegård, B. & Heier Nielsen, S., (1996): Origin of shallow gas in Skagerrak and Kattegat – Evidences from stable isotopic analysis and radiocarbon dating. *NGU Bulletin, Vol. 430*, p. 119-125.

Larsen, Birger, 1996: Sand, grus og sten. Ressource undersøgelse Læsø Trindel-område 574. Marin-geofysisk undersøgelse. Detail opmåling for SNS, Maj 1996. *Danmarks og Grønlands Geologiske Undersøgelse Rapport 1996/106*. 15p.

Lehmann, A. 1995. A three-dimensional baroclinic eddy-resolving model of the Baltic Sea. *Tellus* 47A: 1013-1031

Lehmann, A., Hinrichsen, H.-H. 2000a. On the thermohaline variability of the Baltic *Sea. J. Mar. Syst.* 25: 333-357.

Lehmann, A., Overton, J. M., Leathwick, J. R. 2002a. GRASP: generalized regression analysis and spatial prediction. *Ecological Modelling* 157: 189-207.

Lehmann, A., J.R. Leathwick, JMcC. Overton. 2002b. Assessing New Zealand fern diversity from spatial predictions of species assemblages. *Biodivers. Conserv.* 11: 2217-2238.

Lek, S., and J.-F. Guegan. 1999. Artificial neural networks as a tool in ecological modelling, an introduction. *Ecological Modelling* 120:65-73.

Leth, J.O. & Alhamdani, Z., (2006): Multibeam and sidescan sonar survey of the Natura 2000 site 168: Læsø Trindel and Tønneberg Banke, Kattegat, Denmark. *Danmarks og Grønlands Geologiske Undersøgelse Rapport 2007/1*. 62p.

Lewis, J. R. 1964. *The ecology of rocky shores*. London, The English Universities Press, ** pp.

MacKenzie, B.R., Hinrichsen, H.-H., Plikshs, M., Wieland, K. and Zezera, A.S. 2000. Quantifying environmental heterogeneity: habitat size necessary for successful development of cod *Gadus morhua* eggs in the Baltic Sea. *Mar. Ecol. Prog. Ser.* 193: 143-156.

Maggini, R., A. Lehmann, N. E. Zimmerman, and A. Guisan. 2006. Improving generalized regression analysis for the spatial prediction of forest communities. *Journal of Biogeography* **0:1-21.

Margules, C. R., Pressey, R. L. 2000. Systematic conservation planning. *Nature* 405: 243-255

Marin Mätteknik AB (2003). Report 30911 – Sjömätning i norra Bohuslän.

Marrs, S.J., Atkinson, R.J.A., Smith, C.J., Hills, J.M. (1996). Calibration of the towed underwater TV technique for use in stock assessment of Nephrops norvegicus. Final Report European Commission DG-XIV Study Contract Report Number: 94/069, 155 pp

Marrs, S.J., Atkinson, R.J.A., Smith, C.J. and Hills, J.M. (1998). The towed underwater TV technique for use in stock assessment of *Nephrops norvegicus*. 88–98 ICES CM 1998\G: 9.

Martin, G. et al. 2005. Review on the Blue Corridors concept. Balance interim report.

Mattisson, A. 2005: Mapping Marine Habitats. Pilot study for the coastal areas of the Stockholm County. County administrative Board of Stockholm. Available at http://www.ab.lst.se/upload/dokument/publikationer/M/Rapportserien/2005/R2005_21_Marine_mapping.pdf

McPherson, J. M., W. Jetz, and D. J. Rogers. 2004. The effects of species' range sizes on the accuracy of distribution models: ecological phenomenon or statistical artefact? Journal of Applied Ecology 41:811-823.

Mccann K. S. 2000. The diversity-stability debate. Nature 405: 228-234

Mills 1969. Community concept.**

Minns, C. K., R. G. Randall, J. E. Moore, and V. W. Cairns. 1996. A model simulating the impact of habitat supply limits on northern pike, *Esox lucius*, in Hamilton Harbour, Lake Ontario. *Canadian Journal of Fisheries and Aquatic Sciences* 53:20-34.

Molander, **. 1929.

Moy, F., Nielsen, R., Kristiansen, Aa., Mathiesen, L., Mathiesen, H. 1995. Distributional index of the benthic macroalgae of the Baltic Sea area. *Acta. Bot. Fenn. 155*: 1-51.

Moy, F., Magnusson, J., Aure, J., Johnsen, T., Lømsland, E., Dahl, E., Omli, L., Falkenhaug, T., Rygg, B., Green, N., Walday, M., Pedersen, A. 2004. Langtidsovervåking av miljøkvaliteten i kystområdene av Norge. Kystovervåkingsprogrammet. Årsrapport for 2003. Oslo, Statens forurensningstilsyn, 79 pp.

Moy, F., Magnusson, J., Aure, J., Johnsen, T., Lømsland, E., Falkenhaug, T., Rygg, B., Norderhaug, K. M., Pedersen, A. 2007. Langtidsovervåking av miljøkvaliteten i kystområdene av Norge. Kystovervåkingsprogrammet. Årsrapport for 2006. Oslo, Statens forurensningstilsyn, 95 pp.

Mägi L, Lips U (1998) Budget components of water, salt, and nutrients in the Gulf of Riga in 1993–95. Proc Estonian Acad Sci Biol Ecol 47: 14–31.

Mäkinen, J., Saaranen, V., 1998: Determination of post-glacial land uplift from the three precise levellings in Finland. Journal of Geodesy 72, 516–529.

Möbius, **. 1877.

Nilsson, H.C. and Rosenberg, R. (1997). Benthic habitat quality assessment of an oxygen stressed fjord by surface and sediment profile images. Journal of Marine Systems 11:249-264.

Nilsson, H.C. and Rosenberg, R. (2006). Collection and interpretation of Sediment Profile Images (SPI) using the Benthic Habitat Quality (BHQ) index and successional models. Report No. 5200-2006, NIVA.

Nissling, A., Müller, A., Hinrichsen, H.H., 2003. Specific gravity and vertical distribution of sprat (Sprattus sprattus) eggs in the Baltic Sea. J. Fish Biol. 63, 280-299.

Ockelmann, K. W., Dinesen G.E. (submitted). Life histories of marine invertebrates in the Baltic sea region. *Marine Biology Research*.

Ojaveer E (1995) Large-scale processes in the ecosystem of the Gulf of Riga. In: Ojaveer E (ed) Ecosystem of the Gulf of Riga between 1920 and 1990. Estonian Academy Publishers, Tallinn, pp 268–277.

Otsmann M, Astok V, Suursaar Ü (1997) A model for water exchange between the Baltic Sea and the Gulf of Riga. Nordic Hydrology 28: 351–364.

Otsmann M, Suursaar Ü, Kullas T (2001) The oscillatory nature of the flows in the system of straits and small semienclosed basins of the Baltic Sea. Cont Shelf Res 21: 1577–1603.

Parmanne, R., Rechlin, O. and Sjöstrand, B. 1994. Status and future of herring and sprat stocks in the Baltic Sea. Dana 10: 29-59.

Pearce, J., and S. Ferrier. 2000. Evaluating the predictive performance of habitat models developed using logistic regression. Ecological Modelling 133:225-245.

Pedersen, M. F., Borum, J. 1996. Nutrient control of algal growth in estuarine waters. Nutrient limitation and the importance of nitrogen requirements and nitrogen storage among phytoplankton and species of macroalgae. *Mar. Ecol. Prog. Ser*. 142: 261–272.

Persson, L., and P. Eklöv. 1995. Prey refuges affecting interactions between piscivorous perch and juvenile perch and roach. Ecology 76:70-81.

Petersen, C. G. J. 1913. **

Petersen, C. G. J. 1918. **

Philipson, P. och Lindell, T. 2003: Nationell kartering från satellitbilder av strandtyper längs svenska havskusten. Rapport för Naturvårdsverket. Centre for Image Analysis, Swedish University of Agricultural Sciences, Uppsala University. (in Swedish)

Plikshs, M., Kalejs, M. and Grauman, G. 1993. The influence of environmental conditions and spawning stock size on the year-class strength of the Eastern Baltic cod. ICES CM 1993/J:22.

Põder T, Maestrini SY, Balode M, Lips U, Béchemin C, Andrushaitis A, Purina I (2003) The role of inorganic and organic nutrients on the development of phytoplankton along a transect from the Daugava River mouth to the open Baltic, in spring and summer 1999. ICES J Mar Sci 60: 827–835.

Pont, D., B. Hugueny, and T. Oberdorff. 2005. Modelling habitat requirement of European fishes: do species have similar responses to local and regional environmental constraints? Canadian Journal of Fisheries and Aquatic Sciences 62:163-173.

Poulsen, E.M. 1931. Biological investigations upon the cod in Danish waters. Medd. Komm. DFH, Ser. Fiskeri 9 (1): 150 pp.

Purvis, A., Hector, A. 2000. Getting the measure of biodiversity. *Nature* 405: 212- 220.

Raudsepp U (2001) Interannual and seasonal temperature and salinity variations in the Gulf of Riga and corresponding saline water inflow from the Baltic Proper. Nordic Hydrology 32: 135–160.

Reker, J. B., et al. ** 2007. Marine landscapes in the Baltic Sea region. *BALANCE Interim Report* No. **, ** pp.

Remane A. 1934. Die Brackwasserfauna. *Verh. Dt. Zool. Ges.*, pp. 34-74.

Roff. J. C., Taylor M. E. 2000. Viewpoint. National frameworks for marine conservation – hierarchal geophysical approach. *Aquatic Con. Mar. Freshw. Ecosys.* 10: 209-223.

Roff, J.C., Taylor, M.E., Laughren, J. 2003. Viewpoint: Geophysical approach to classification, delineation and monitoring of marine habitats and their communities. *Aquatic. Con. Mar. Freshw. Ecosyst.* 13: 77-90.

Romare, P. 2000. Growth of larval and juvenile perch: the importance of diet and fish density. *Journal of Fish Biology* 56: 876-889.

Rosenberg, A.A. 2003. Multiple uses of marine ecosystems. In: Sinclair M, Valdimarsson G (eds.) *Responsible Fisheries in the Marine Ecosystem*, pp 189-196.

Runca, E., Bernstein, A., Postma, L., Silvio, G.D. 1996. Control of the macroalgae bloom in the Lagoon of Venice. *Ocean and Coastal Management* 30: 225-257.

Ruth, **, Lindholm, **. 2002.

Sale P. F, Hanski I., Kritzer J. P. 2006. The merging of metapopulation theory and marine ecology: Establishing the historical context. In: Kritzer J. P., Sale P.F. (eds.) *Marine metapolulations*, pp. 3-28.

Sandström, O., I. Abrahamsson, J. Andersson, and M. Vetemaa. 1997. Temperature effects on spawning and egg development in Eurasian perch. *Journal of Fish Biology* 51:1015-1024.

Schmieder, K., Lehmann, A. 2004. A spatio-temporal framework for efficient inventories of natural resources: A case study with submersed macrophytes. *Journal of Vegetation Science* 15: 807-816.

Schmidt, C. 2000. Populationsgenetische Untersuchungen am Ostseedorsch (*Gadus morhua* L.). *Dipl. Thesis, University of Kiel*, 83 pp.

Schramm, W., Nienhuis, P. H. (eds.) 1996. Marine Benthic Vegetation, recent changes and the effects of eutrophication. Springer, Berlin: pp 465.

Segerstråle, S. G. 1957. The Baltic Sea. In J. W. Hedgpeth (ed.). Treatise on Marine Ecology and Paleocology I. Ecology. *Geol. Soc. Am. Mem*. 67: 751-800.

Seinä, A. et al., 2001. Ice seasons 1996-2000 in Finnish sea areas. MERI – Report Series of the Finnish Institute of Marin Research, No. 43.

Skidmore, **. 2002.

Smedberg 2006: Brygginventering I flygbilder längs Sveriges kust. Rapprt för Naturvårdsverket. Metria Miljöanalys.

Smith, C., A. Douglas, and P. Jurajda. 2001. Oviposition site selection and embryo mortality in perch. *Journal of Fish Biology* 58:880-882.

Smith, C.J., Marrs, S.J., Atkinson, R.J.A., Papadopoulou, K.N. and Hills, J.M. (2003). Underwater television for fisheries-independent stock assessment of *Nephrops norvegicus* from the Aegean (eastern Mediterranean) Sea. *Mar. Ecol. Prog. Ser.* 256: 161-170

Snickars, M., A. Sandström, A. Lappalainen, and J. Mattila. 2007. Evaluation of low impact pressure waves as a quantitative sampling method for small fish in shallow water. *Journal of Experimental Marine Biology and Ecology* 343:138-147.

Sparrevohn and Støttrup, (In press). Diet, abundance and distribution can be used as indices of turbot (Psetta maxima L.) habitat suitability. *Rev. Fish. Sci.* In press.

Stepputtis, D. 2006. Distribution patterns of Baltic sprat (*Sprattus sprattus* L.) - causes and consequences. *PhD thesis, Univ. Kiel*.

Støttrup, J.G., C.R. Sparrevohn, J. Modin and K. Lehmann. The use of releases of reared fish to enhance natural populations. A case study on turbot *Psetta maxima* (Linné, 1758). *Fish. Res.* 1361: 1:20 (2002).

Stålnacke P, Vagstad N, Tamminen T, Wassmann P, Jansons V, Loigu E (1999) Nutrient runoff and transfer from land and rivers to the Gulf of Riga. Hydrobiologia 410: 103–110.

Suursaar Ü (1995) Nutrients in the Gulf of Riga. In: Ojaveer E (ed) Ecosystem of the Gulf of Riga between 1920 and 1990. Estonian Academy Publishers, Tallinn, pp 41–50.

Suursaar Ü, Kullas T, Otsmann M (2002) Flow modelling in the Pärnu Bay and the Kihnu Strait. Proc Estonian Acad Sci Eng 8: 189–203.

Suursaar Ü, Kullas T, Otsmann M (2003a) A model study of the sea level variations in the Gulf of Riga and the Väinameri Sea. Cont Shelf Res 22: 2001–2019.

Suursaar Ü, Kullas T, Otsmann M, Kõuts T (2003b) Extreme sea level events in the coastal waters of western Estonia. J Sea Res 49: 295–303.

Suursaar, Ü., Astok, V., Otsmann, M. 1998. The front of Väinameri. In EMI Report Series, 9: 23-33.

Sørensen, T. K. 2005. Workshop on the development and management of marine protected areas. *TemaNord 2005*, No. 587, 111 pp.

Thompson, B.M., Riley, J.D. 1981. Egg and larval developmental studies in the North Sea cod (*Gadus morhua* L.). Rapp. P.-v. Réun. Cons. perm. int. Explor. Mer 178: 553-559.

Tenson, J. 1995. Phytoplankton of the Pärnu Bay. In: Ojaveer E (ed) Ecosystem of the Gulf of Riga between 1920 and 1990. Estonian Academy Publishers, Tallinn, pp 104–126.

Thorpe, J. 1977. Synopsis of biological data on the perch *Perca fluviatilis* Linnaeus, 1758 and *Perca flavescens* Mitchill, 1814. FAO, Rome.

Thorson, G. 1950. **

Tilman D. Causes, consequences and ethics of biodiversity. Nature 405: 208- 212

Treasurer, J. W. 1983. Estimates of egg and viable embryo in a lacustrine perch, *Perca fluviatilis*. Environmental Biology of Fishes 8, 3-16.

Trei, T., Kukk, H., Kukk, E. 1987. Phytobenthos as indicator of the degree of pollution in the Gulf of Finland and in neighbouring areas. - Meri. 13: 63-110.

Tuck, I.D., Atkinson, R.J.A. and Chapman, C.J. (1994). The structure and seasonal variability in the spatial distribution of *Nephrops norvegicus* burrows. Ophelia 40:13-25.

Tuck, I.D., Chapman, C.J., and Atkinson, R.J.A. (1997A). Population biology of the Norwegian lobster, *Nephrops norvegicus* (L.) in the Firth of Clyde, Scotland – I: Growth and density. – ICES Journal of Marine Science, 54: 125–135.

Tuck, I.D., Chapman, C.J., Atkinson, R.J.A., Bailey, N. and Smith, R.S.M. (1997B). A comparison of methods for stock assessment of the Norwegian lobster, *Nephrops norvegicus*, in the Firth of Clyde. Fisheries Research 32:89-100.

Turner, M. G. 1989. Landscape ecology: The effect of pattern on process. Ann. Rev. Ecol. Syst. 20: 171-197

van der Veer, H., R. Berghahn, J.M. Miller, A.D. Rijnsdorp. 2000. Recruitment in flat-fish, with special emphasis on the North Atlantic species: Progress made by the Flatfish Symposia. ICES. J. Mar. Sci. 54, 202-215.

Vaughan, I. P., and S. J. Ormerod. 2005. The continuing challenges of testing species distribution models. Journal of Applied Ecology 42:720-730.

Vincent, C., Heinrich, H., Edwards, A., Nygaard, K., Haythornthwite, **. 2002. Guidance on typology, Reference Conditions and Classification Systems for Transitional and Coastal Waters. CIS Working Group 2.4 (Coast), Common Implementation Strategy of the Water Framework Directive, European Commission. Final draft, 119 pp.

Vincent, M. A., Atkins, S. M., Lumb, C. M., Golding, N., Lieberknecht, L. M., Webster, M. 2004. Marine nature conservation and sustainable development – the Irish Sea Pilot. *Report to Defra by the Joint Nature Conservation Committee, Peterborough*, ** pp.

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, 103-122.

Waern, M. 1952. Rocky-shore algae in the Öregrund archipelago. - Acta Phytogeogr. Suec. 30: 1-298.

Wallentinus, I. 1991. The Baltic Sea gradient. In: Mathieson, A. C., Nienhuis, P. H. (eds) Intertidal and littoral ecosystems. Goodall, D. W. (ed) The ecosystems of the world, vol. 24. Elsevier, Amsterdam: 83-108.

Wennberg, S., Lindblad, C., Albertsson, J., Bergström, U., Isæus, M., Kilnäs, M., Mattisson, A., Sandman, A. 2006. Sammanställning och Analys av Kustnära Undervattenmiljö (SAKU). *Stockholm, Naturvårdsverket Rapport* nr. 5591, 99 pp.

Wentworth, C.R., (1922). A scale of grade and class terms for clastic sediments. *Jour. Geology, 30*: 377-392.

Westernhagen, H.v. 1970. Erbrütung der Eier von Dorsch (Gadus morhua), Flunder (Pleuronectes flesus) und der Scholle (Pleuronectes platessa) unter kombinierten Temperatur- und Salzgehaltsbedingungen. Helgoländer wissenschaftliche Meeresuntersuchungen 21: 21-102.

Wieland, K. 1988. Distribution and mortality of cod eggs in the Bornholm Basin (Baltic Sea) during two patch studies in 1986. Kieler Meeresforsch. Sonderh. 6: 331-340.

Wieland, K. and Köster, F.W. 1996. Size and visibility of Baltic cod eggs with reference to size-selective and stage-dependent predation mortality. J. Appl. Ichthyol. 12: 83-89.

Wood RD, Imahori K (1965) A revision of the Characeae. First Part. Monograph of the Characeae. Verlag von J. Cramer, Weinheim.

Worm B., Lotze H.K., Hillebrand H., Sommer U., 2002, *Consumer versus resource control of species diversity and ecosystem functioning*, Nature, 417, 848–851

Wright, **. 2002.

Yee, T.W., Mitchell, N.D. (1991). Generalized additive models in plant ecology. J. Veg. Sci. 2, 587_/602.

Yurkovskis A (2004) Long-term land-based and internal forcing of the nutrient state of the Gulf of Riga (Baltic Sea). J Mar Syst 50: 181–197.

Zaniewski, A. E., Lehmann, A., Overton, J. M. 2002. Predicting species spatial distributions using presence-only data: a case study of native New Zealand ferns. *Ecological Modelling* 157: 261-280.

Ærtebjerg, G., Andersen, J. H., Hansen, O. S. (eds.) 2003a. *Nutrients and Eutrophication in the Danish Marine Waters. A challenge for Science and management*. National Environmental Research Institute, 126 pp.

Ærtebjerg, G., Carstensen, J., Axe, P., Druon, J.-N., Stips, A. 2003b. The 2002 Oxygen Depletion Event in the Kattegat, Belt Sea and Western Baltic. Thematic Report (HELCOM 2003). Helsinki Commission. *Baltic Sea Environment Proceedings* 90: 64 pp. Available at: http://www2.dmu.dk/1_viden/2_Publikationer/3_Ovrige/rapporter/Helcom2003_BaltSeaEnvironProcNo90.pdf

A P P E N D I C E S

A P P E N D I X   A
Case studies: Mapping and MCE Modelling of Marine Habitats

Pilot Area 1

Pilot area 1 comprise of the outer, North-western part of the Baltic Sea area, where the diversity of marine species is the highest. The area is highly heterogenic regarding, salinity, benthic substrate, bathymetry and light attenuation. Hence two sub-areas were selected, for modelling of different habitats, 1A along the Norwegian and Swedish rocky coast and deep, muddy bottom, and 1B including Danish sandy coasts and off-shore boulder reefs and soft bottoms at greater depth.

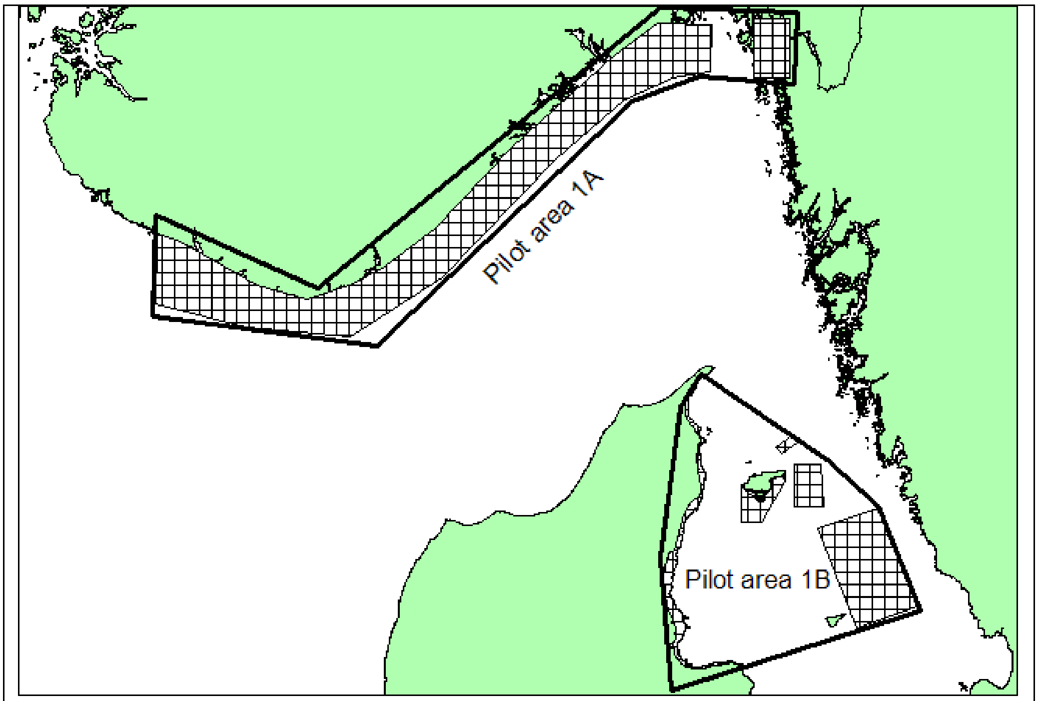


***Fig. 1.** Map showing Pilot Area 1 located in the Kattegat/Skagerrak area between Denmark, Norway, and Sweden (by C.R. Sparrevohn).*

The pilot studies on mapping of marine NATURA 2000 Annex 1 habitats were carried out in three of the four pilot areas (PA): 1) Kattegat-Skagerrak in Norwegian, Swedish and Danish waters, 3) Baltic Archipelago in Finnish and Swedish waters, and 4) Gulf of Riga in Lithuanian waters. In addition, wave impact on coastal habitats was studied in Latvian waters.

6.1 PA1. Seabed mapping in the Rauøyfjord-Hvaler, Norway

Authors: Valérie Bellec, Reidulv Bøe, Ole Christensen, Oddbjørn Totland, Heidi Olsen

6.1.1 Introduction

NGU has performed seabed mapping in Rauøyfjorden, around the islands Missingen and Søstene, and in the sound Akersundet and southwest of the Hvaler islands (Fig. 1). The mapping was performed in the framework of the BSR Interreg IIIB project

BALANCE and partly in the framework of the NGU project Geologi i Osloregionen (GEOS).

The area was chosen for studies because it is located within area I of the Balance project, and because parts of it are located within the proposed Hvaler Marine Protected Area. The area exhibits large topographical and geological variations over short geographical distances. The northern part of Rauøyfjorden has been closed by the military, and has not been affected by human activities for several decades. It can thus be used as a calibration area. The area is ideal for investigating the effects of fisheries and other activities, which have taken place in surrounding areas.

Aims

1) Investigate the resolution and possibilities of the interferometric sonar for habitat mapping. Interferometric sonar data were first used as a stand alone for interpretation. Interpretations were then refined with grab sample descriptions and video inspection data of the seafloor.

2) Provide other partners with geological interpretations and visual observations of the seabed from a) areas unaffected for decades by human activities and b) from nearby areas strongly influenced by fisheries and other human activities.

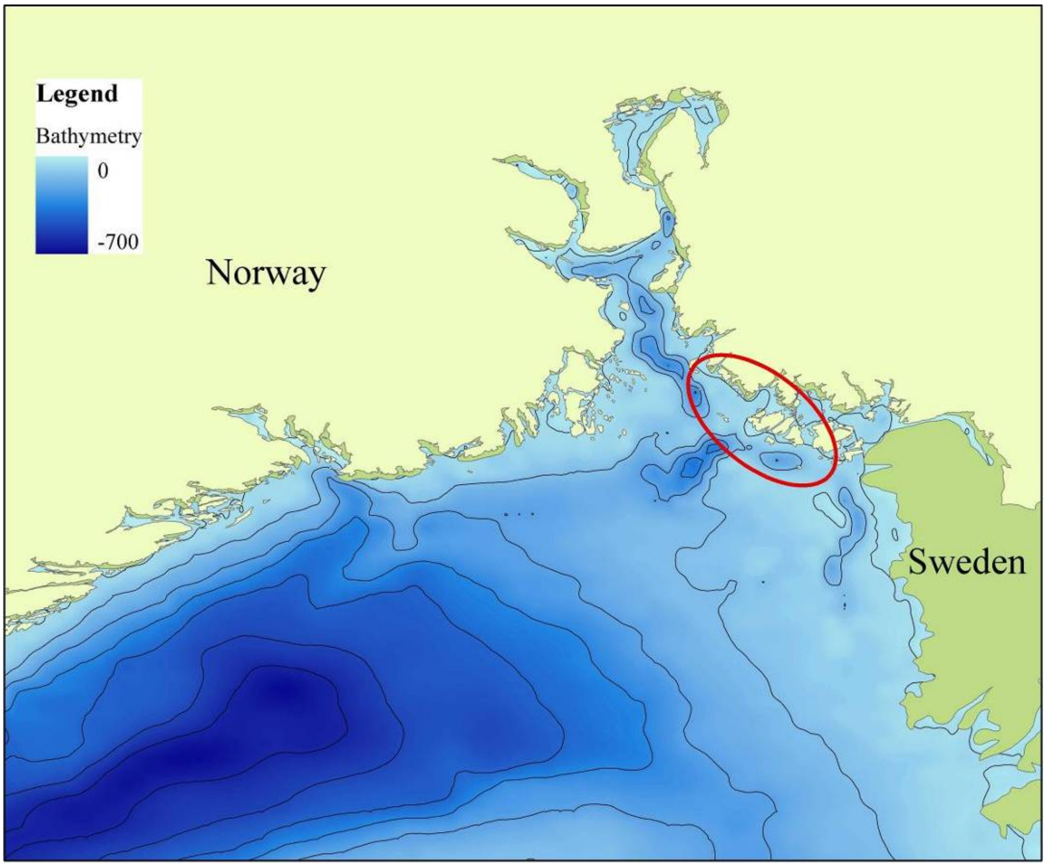


Figure 1. Balance area, red circle marks the approximate location of areas mapped by NGU. Details of the areas are presented in Figure 2.

6.1.2 Material & Methods

Interferometric sonar bathymetry data were processed to ascii format in a resolution of 1x1 m grids. Interferometric sonar backscatter data were produced as georeferenced tiff images of 1x1 m resolution. Video records of the seabed were logged and stored digitally. A GeoSwath interferometric sonar was used to collect batymetric data and reflectivity data. Standard routines and processing algorithms were used to process the data. All bathymetry maps and backscatter maps are in a resolution of 1x1 metres. Habitat and geological interpretation maps are in 10x10 metres resolution.

Digital video records are available on request, but systematic biological registration or interpretation was not performed. It can be mentioned, however, that in Rauøyfjorden videos display an interesting biological diversity including up to 1 m tall *Funiculina quadrangularis.*

6.1.3 Results

Geological maps were made according to the classification scheme discussed in Copenhagen, with a grid resolution of 10x10 meters. Fig. 2 show the interpretations

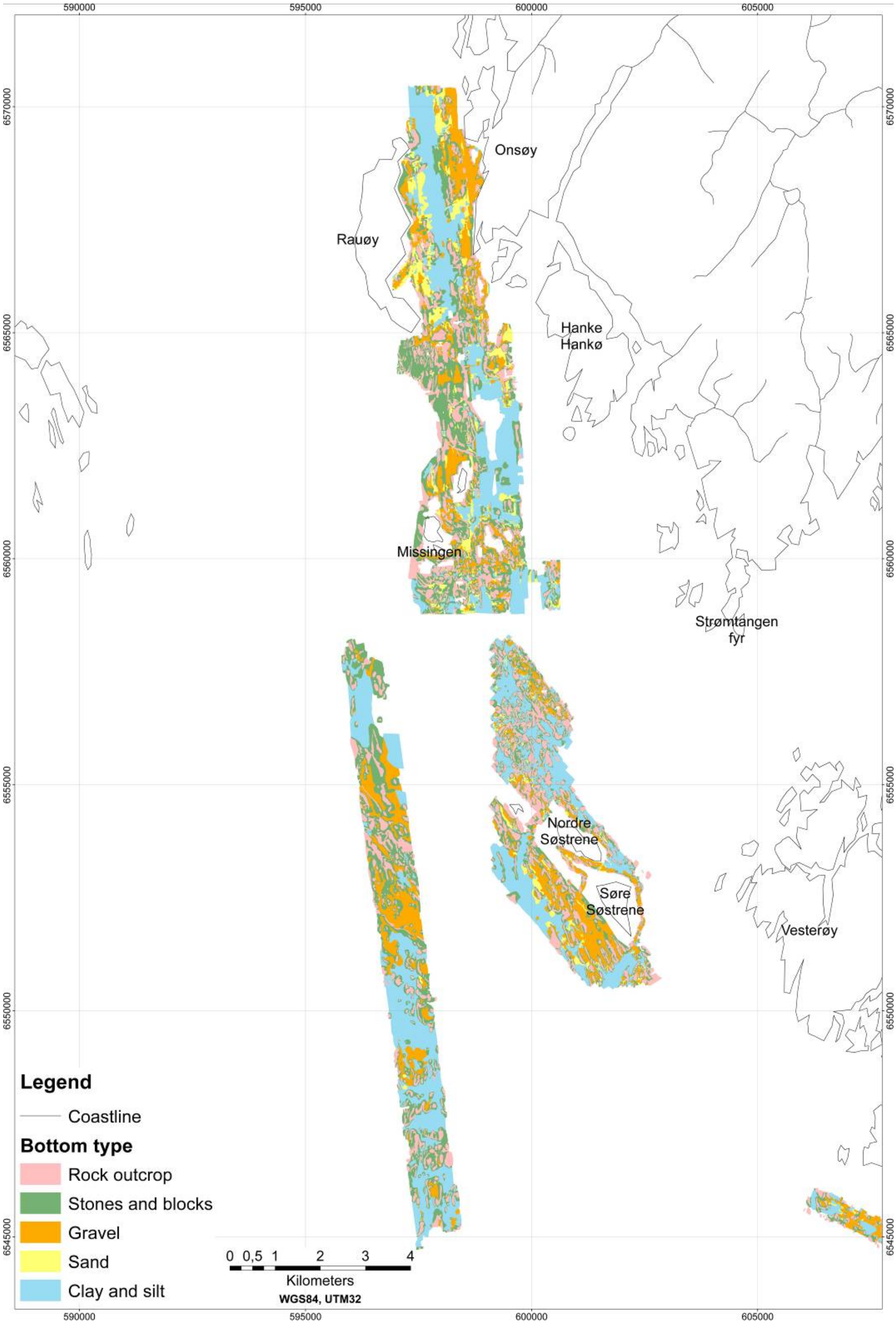


Figure 2. Interpreted bottom types, Rauøyfjorden, Missingen and Søstrene.

6.1.4 Discussion

The interpretation was tested using a Van Veen grab with a mounted video system (Fig. 3). Eleven video lines were run in various areas in order to calibrate the interpretations.

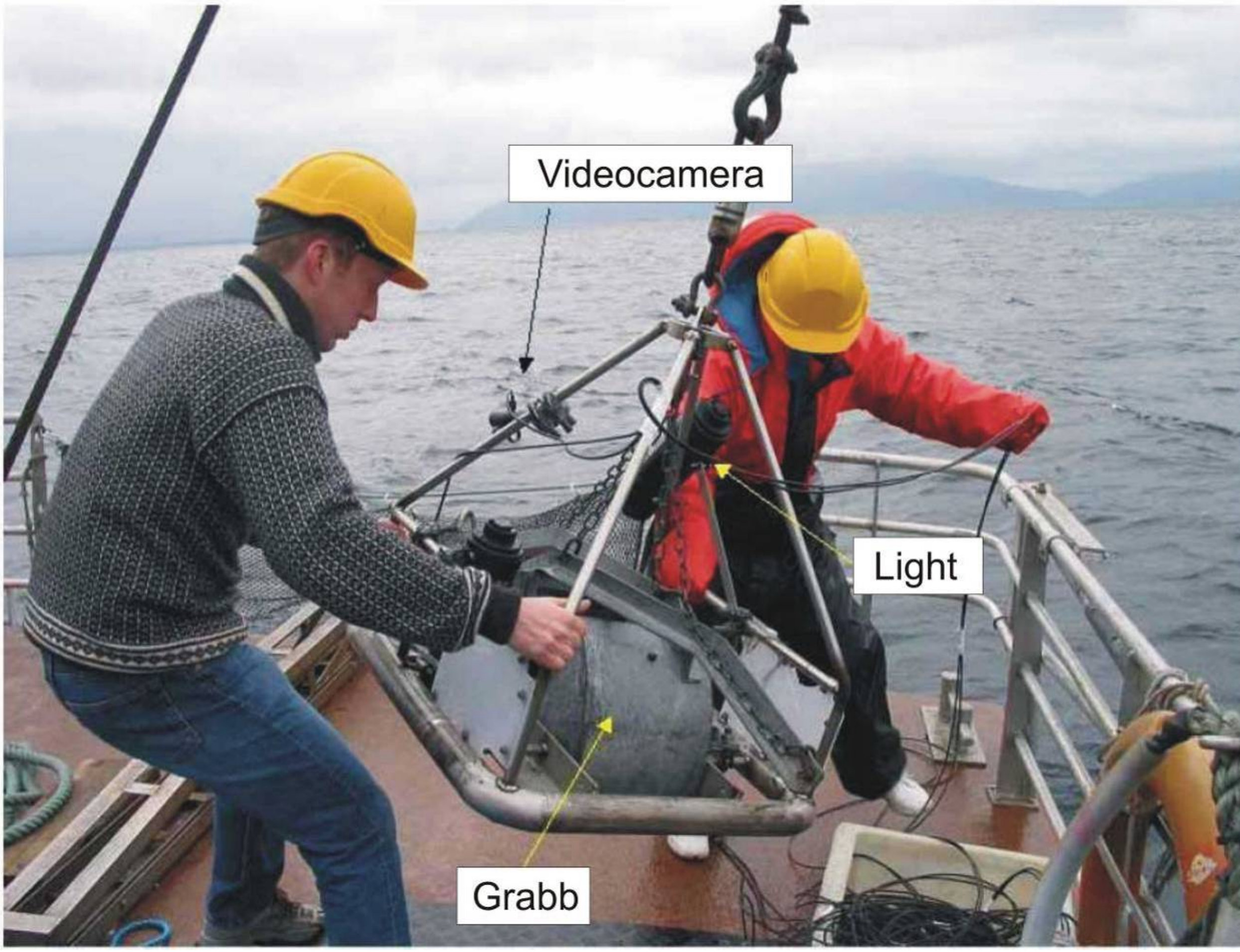


Fig. 3. Picture of the Van Veen grab used for collecting video data.

Data acquisition with interferometric sonar covers large areas in shallow waters, and results in very detailed bathymetry and high quality sonar images.

The GeoSwath interferometric sonar used for bathymetric mapping worked well, but in central Rauøyfjorden, northeast of Missingen, northwest of Søstrene and southwest of Hvaler, bad weather and water depths in excess of the limits for the equipment (ca. 180 m) caused reduced data quality and small holes in the datasets.

It is essential to have high resolution bathymetry and backscatter information available for detailed studies and understanding of seabed morphology and habitats in shallow waters. The detailed maps produced in this project are ideal for relating species to certain habitats.

The usefulness of the Van Veen grab with an attached video camera was limited due to low camera resolution. A high-resolution camera is recommended. The advantage of the system is that it is possible to record videos and obtain samples at the same time.
There was a good correlation between the bathymetry and sonar interpretations and the video inspections/grab samples.

6.1.5 Conclusion and perspectives

We recommend obtaining and processing acoustic data prior to sampling, preferably during two separate cruises. Even if the acoustic data could be processed offshore and used during the sampling/video recording, the details of the high resolution acoustic data only stand forward when interpreted and displayed by visualisation tools. A two-cruises strategy is recommended; the first cruise mainly focusing on acoustic acquisition, possibly with a limited sampling program. The second cruise should focus on video recording and sampling. On the second cruise, there should be both biological and geological expertise onboard to do preliminary interpretations and adjust the cruise programme.

6.2 PA1. Mapping of the NATURA 2000 Annex 1 habitats 1170 and 1180 in N Kattegat, combining acoustic and ground truth methods

Authors: J. O. Leth and Z. Al-Hamdani.

6.2.1 Introduction

For implementing the EC Habitats Directive broad scale marine habitat mapping based on acoustic techniques combined with ground truth verification is expected to be a useful tool. The technique can be customised to fit at least three purposes. These include a) broad scale scanning of large areas of seafloor for which little information exist in order to establish a baseline habitat map for designating sites, b) more detailed surveys within individual Natura 2000 sites for delineating the area of individual habitats, and c) identifying locations and area of habitats with a limited distribution, such as '*Submarine structures made by leaking gases (1180)*'. The multibeam swath bathymetric and the sidescan sonar devices used in the study at Læsø Trindel are the most highly developed and versatile available systems. They offer great data control and supporting real-time visualisation of sonar data as true geo-corrected mosaic seabed maps. The techniques applied here have demonstrated the usefulness of combining acoustic methods with ground truthing to produce maps revealing the physical and biological characteristics of the seabed. Sidescan sonar provides information on sediment texture, topography, bed forms, and other discrete objects at the seabed (e.g. boulders) and to certain extend the degree of macroalgae coverage. The multibeam data system provides depths of centimetre resolution. Multibeam data processing enhances subtle aspects of relief elements through shading techniques for an understanding of erosive and depositional processes. Based on the understanding of the sediment dynamics and geological structure the marine scientist produce maps of the seabed, which help managers of the marine environment to predict the impacts on those habitats which may be of high nature conservation and ecological value.

Background. The Balance project aims to develop informed marine management tools to improve spatial planning in the Baltic Sea based on cross-sectoral and transnational co-operation. One of the main objectives is to develop marine landscape maps for the entire Baltic Sea as well as marine habitat maps in four pilot areas. Like many of the existing Natura 2000 habitat areas in the Baltic region the Læsø Trindel has been designated based on for the time being available but scattered physical and biological data sets. The Læsø Trindel/Tønneberg Banke has been designated as a NATURA 2000 area due the presence of boulder reefs and calcareous reefs made by leaking gasses. By that, the purpose of the investigations at Læsø Trindel was to increase the knowledge of the physical and biological conditions to qualify and improve the basis for the habitat designation of the latter area. Furthermore, the results add new knowledge for the developing habitat mapping methods to make the production of habitat maps more efficient with a higher degree of confidence.

Aims. The aim of the project is to assess the combined use of marine acoustics and ground truthing by diving to identify, classify, and map marine HD Annex 1 habitats. Where the methodology enables further distinction of the individual habitat categories a sub-division into habitat sub-groups will be proposed and its relevance to characterise the HD Annex 1 habitats evaluated accordingly. The existing definitions introduced by Dahl et al. (2003) have solely been based on the diver's observation. In acknowledgement of the complexity of the Danish boulder reefs the present study will adapt and elaborate the latter definitions to extend its applicability in the characterisation of the boulder reef habitat when the background data is a combination of acoustic data and ground truth data. The project aims at providing evidence on the intercalibration of newly acquired acoustic data with other geological and biological information acquired from a dive survey within the NATURA 2000 area.

6.2.2 Material & Methods

The study area. The Nature 2000 site 168, Læsø Trindel and Tønneberg Banke is located about12 km northeast of the island Læsø in the northern Kattegat between Denmark and Sweden (fig. 1). The bathymetry of northern Kattegat around the island of Læsø is very irregular with depths reaching 123m only 12km east of Læsø, whereas flat areas and reefs with more shallow water depths less than 10m extend in north-easterly and north-westerly direction. The top of the Læsø Trindel plateau is at only 3.8m depth of water. This plateau has an extension of approximately 2x2 km where the water depth varies between 3.8m and 10m. The sediment on the plateau consists is known to consist mainly of gravel and minor stones with patches of larger boulders in between, though there are no cavernous elements left. Aggregate extraction from the reef, especially by the removal of boulders by man, had a provable negative influence on the amount of stones present.

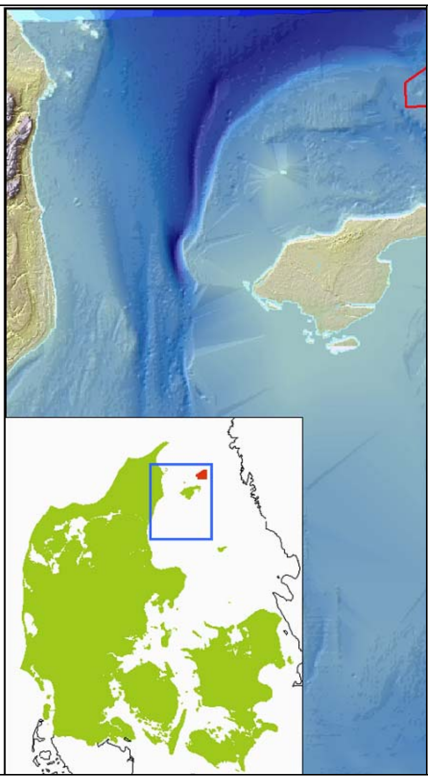


Figure 1. Bathymetric map showing the location of the Natura-2000 habitat area 168, Læsø Trindel and Tønneberg Banke (red box) northeast of the island Læsø in the northern Kattegat. The study area is located inside the shown habitat area. The modelled bathymetry is based on data from The Royal Dan-

6.2.3 Material and methods

During the project, the applicability of the combined use of multibeam sonar and sidescan sonar systems has been tested as a tool for mapping of marine habitats. The acoustic survey was performed in Sept 2005. Based on a preliminary interpretation of the acoustic data, features and sites for the subsequent ground truthing by diving were decided by GEUS' geologists and performed in Oct 2005. Dependent on the type of acoustic features and the state of the substrate the geologists suggested either paravane diving or point diving to be performed. I.e. objects or other pronounced local features were inspected by point diving, while larger areas of specific types and change of substrate were inspected by paravane diving. The results of the ground truthing were subsequently integrated with the acoustic data set and the initial interpretation of the substratum was adjusted and extrapolated within the project area.

The multibeam sonar system (MBS). The used system is a high resolution EM3002 dual head seabed mapping system. Each head delivers a 1.5° beam for transmission and reception, where the swath coverage of the dual head system can reach up to 10 times the water depth. In the high-density mode of operation, each head acquires up to 254 soundings per ping. The operating frequencies are 293 and 307 kHz to avoid interference between the two heads. The operation range of the system is from 1 m to 150 m, which is also a function of salinity and temperature. The depth resolution is very high (~1cm), the across track measurement accuracy is a function of depth and the distance from nadir position, a nominal range resolution of 5cm is reported.

The multibeam transmits across track fan shaped beam, which can be electronically stabilized for pitch, and the received beams are electronically stabilized for roll (fig. 2a).

The pitch, roll, heave, heading, and the applied stabilization are all taken into account when calculating the sounding depths and positions.

Data was collected and stored using the Kongsberg SIS software. Return signal depth and amplitude was recorded for further processing. All other information concerning the installation, calibration and navigation data were stored in as well as the sound velocity profile taken during the survey. The raw data was then processed by the Kongsberg software (Neptune), where all raw data converted to survey data, which then can be processed for depth and backscattering data files (details in Leth et al. 2006).

After performing the required processing on the multibeam data, the data for Læsø Trindel were pooled and gridded together. The resulting sun illumination map (fig. 5) was printed and reveals a highly detailed manifestation of the seabed in the survey area. The depth of the seabed varies between 3.5 m down to 42 m approximately. The structures are well pronounced in the map and places of stone reefs and flat sediment areas can be readily distinguished.

The sidescan sonar (SSS). The used equipment was the EdgeTech DF-1000 dual frequency digital sidescan sonar. The system operates at two frequencies; 100 and 500 kHz corresponding to a standard and high-resolution operation respectively. The system generates a fan shape beam in the cross track direction with 50 m beam width (fig. 2b). In the along track direction the beam width is 1.2° for the 100 kHz operation and 0.5° when the 500 kHz option is used. A nominal operating range of 200-300 m is reported and that depends on the type of the seabed sediments, and to a minor extent on temperature and salinity. A survey cruise of 6 knots was found to be adequate for the survey. The resulting sidescan picture is of high resolution (0.25 m) and is considered being very useful for seabed habitat mapping (fig. 5). The Triton Elics ISIS Sonar software was used for collecting as well as processing sidescan data. The processed data was build into a mosaic by the same software and displayed in DelphMap, where it could be configured and merged and exported in different format acceptable by the GIS software for presentation.

Ground truthing by diving. Transects of relevance to the aim of the survey and crossing areas of interest were chosen for diving conducted shortly after the acoustic survey. Video and still camera were used as well as the observations of the experienced divers. Based on the acoustic interpretations a series positions were chosen by the geologist and put in the order of priority for the ground truthing by diving (fig. 5). The listed positions were sub-divided into point dive positions and paravane dive positions. By the point diving procedure the ship was anchored within a distance of a few metres from the chosen position. The primary task of point diving was to recognise the objects or substrate features pointed out by the geologist on the acoustic data, and to confirm if the interpretation was correct. Finally, the diver should document the substrate features by still photos and/or underwater digital video recordings. The paravane diving was conducted along pre-defined survey lines using the GPS system to ensure the exact position of the diver. The uncertainty was estimated to a few metres off the line. The survey speed of the paravane diver (2 – 4 km/t) was appropriate for the diver to register the overall substrate and biological features. The parameters registered by the diver are: the type of substrate (sediment type), the degree of stone coverage (%), and the type and degree of vegetation covering the stones.

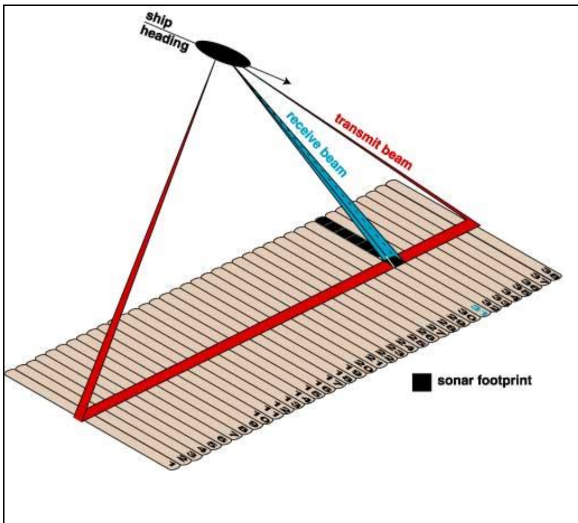


Figure 2a. Schematic diagram of multibeam system operation.

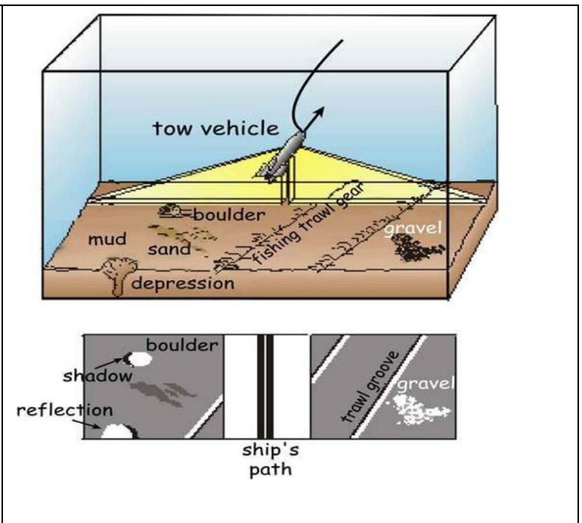


Figure 2b. Sketch showing the sidescan sonar in operation. The yellow colour illustrates the swath width of the sidescan beams and the coverage of the seabed.

Seabed geology. The mapping of seabed structures and sediment types in the Northern Kattegat is mainly based on the marine geological surveys performed by GEUS, Danish Forest and Nature Agency and the Swedish Geological Survey (SGU) (Hermansen & Jensen 2000). The presence of the boulder reefs at Læsø Trindel and its variability is closely linked to the geological development of the area. The geology of the Læsø Trindel area is described as a vast accumulation of glacigenic deposits. The type and distribution of the coarse grained sediments giving rise to the stone reef indicates deposition and deformation in the ice marginal zone during the last glacial period. Based on the interpretation of seismic data deformation by thrusting and folding has caused complex layering of the sediments. Furthermore, the morphology of the glacial surface is quite undulating with a relief of up to 10m throughout the entire area. At the Læsø Trindel proper, the supposed glacigenic formations outcrop with a high frequency of cobbles and boulder in the surface layers. In general, thin layers of reworked residual sandy and gravely sediments and marine postglacial sand cover the glacigenic deposits (Larsen 1996). The detailed mapping of the morphology and the seabed sediment distribution, however, indicates a considerable variation of the stone coverage throughout the area. This expresses different depositional processes in the glacial or late glacial period, e.g. intense erosion, sub-glacial processes, or deposition in front of the glacier during the late glacial period. More of these processes could explain the presence of cave-forming layers of cobbles and boulder. Moraine deposits have never been recognised in the area, neither onshore nor offshore (Fredericia 1987).

Broad scale seabed habitat classification. A Danish boulder reef is in a geomorphological sense “an elongated area, or bank, rising from the surrounding seabed”. However, a further characterisation of the reef is important in order to recognise the variation in reef morphology. This includes recognising i.e. the amount and density of the hard substrate. Dahl et al. (2003) suggested a definition (fig. 3) illustrating three different kind of reefs. As their definition is solely based on divers observations there is a need for extending this classification to make it applicable for the characterisation of boulder reefs and for the production of broad scale habitat maps on the basis of combined acoustic and ground truth data. The sub-division proposed by Dahl et al. (2003) has been adapted and elaborated in the present study and a new classification is proposed and tested. Despite the resolution of individual objects is as high as 0.25 m using sidescan data, a more general evaluation of the reef substrate is needed for the study area of about 12 km^2. The mosaic sidescan data and the high-resolution bathymetry map (fig. 5) were analysed to classify the entire area giving rise to three new re-defined reef sub-divisions. These are:

Reef-1: Coherent formation of stones with high cover (75 – 100%) of hard substrate. When surrounded by sandy and/or gravely seabed the boundary is sharp.

Reef 2: Scattered formation with high to medium cover (25 – 75%) of hard substrate. When surrounded by sandy and/or gravely seabed the boundary is gradual.

Reef 3: Smaller individual banks of stones each at least 10m^2 forming a low cover (5-25%) of hard substrate. When surrounded by sandy and/or gravely seabed the boundary is gradual.

It has been found that reef areas of 10 m^2, as defined by Dahl et al. (2003) "*hard substrate covering at least 5% of the seabed within a area of 10m^2*" is difficult to recognise from a general acoustic classification. However, we acknowledge their method developed based on diver's investigations. An area of 10m^2 is beyond the limit of resolution of the sidescan mosaic and much too detailed in relation to the idea of using the acoustic method for the characterisation of larger seabed areas. Boxes on the side-scan mosaic of different sizes have been tested (side length of 10 m, 30 m, 50 m and 100 m) to evaluate the relationship between the resolution of individual objects versus the characterisation of the defined reef types (see fig. 4). We conclude, that for the purpose of the seabed classification into three new classes of reef types a box/cell size of 50 x 50 m is the optimum size within which the hard substrate coverage most reasonable can be evaluated. At the same time the chosen cell size has to be considered as a pragmatic way to classify the seabed systematically and to delineate broad scale habitat types on a harmonised set of acoustic/ground truth data.

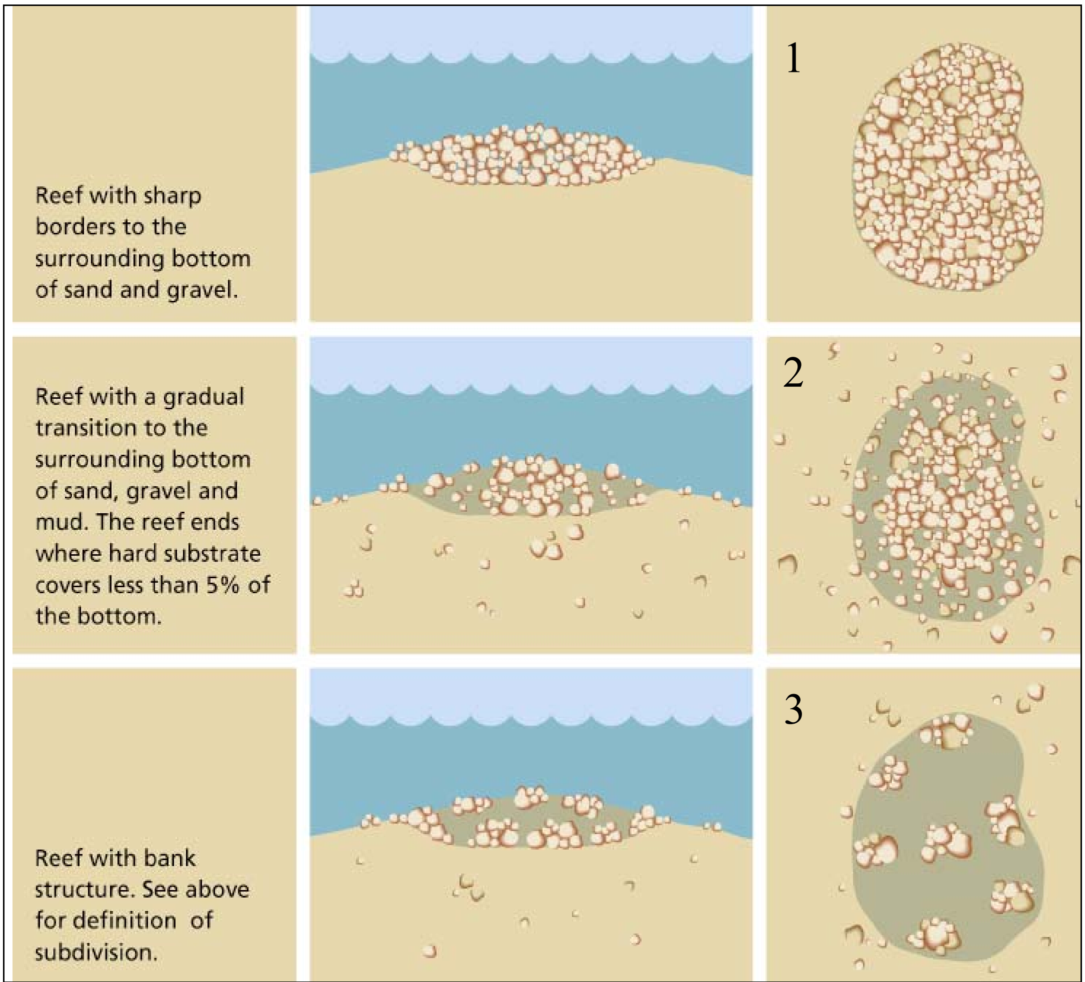


Figure 3. Definition of hard substrate reefs with a schematic presentation of three different reef types and their delineation to other types of seabed habitats based on divers observation. The middle column shows a vertical cut and the right column shows the reef seen from above. The subdivision into type 1, 2, and 3 has been adapted to the classification of boulder reefs in the present study. Modified after Dahl *et al*. 2003.

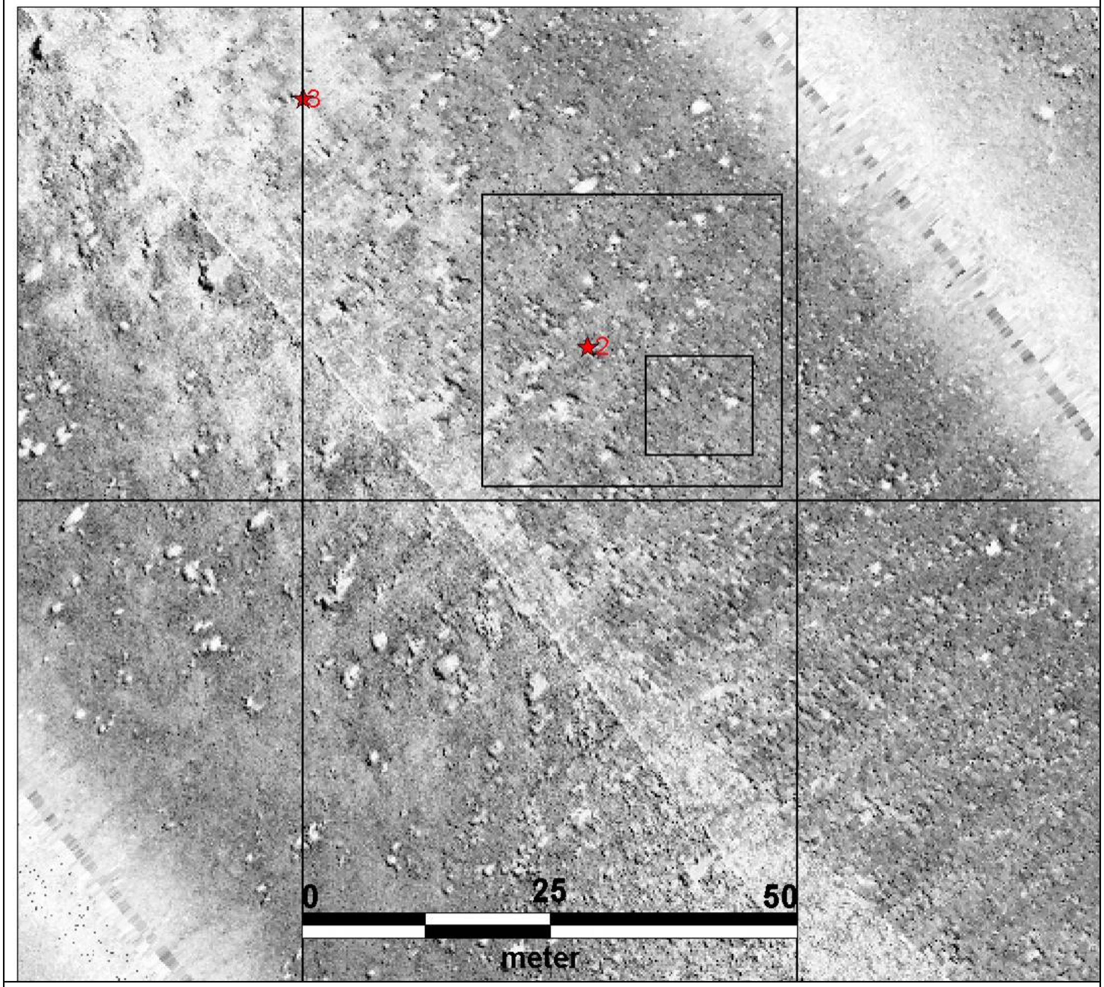


Figure 4. A section of the sidescan mosaic showing different cell sizes used for the evaluation: 10m, 30m and 50m. The 50m cell size has been applied to the present classification of the seabed into three reef types. The red stars indicate ground truth positions.

6.2.4 Results

Bathymetry and sediment map production. After performing the required processing on the multibeam data, the data for Læsø Trindel were pooled and gridded together. The resulting sun illumination map was printed and reveals a highly detailed manifestation of the seabed in the survey area (fig. 5). The depth of the seabed varies between 3.5m down to 42 m approximately. The structures are well pronounced in the map and places of stone reefs and flat sediment areas can be readily distinguished.

The ground truthing by diving was then conducted in the Læsø Trindel to verify the bathymetric results of the multibeam system as well as the backscatter results of the sidescan sonar system. The divers used a high precession depth-meter for measuring the exact depth from sea level. The position of the diver was estimated from the position of the survey boat and the length of the towing cable. The positions of the ground truth reporting were plotted on top of the bathymetric map and a comparison was made between the reported depth and the multibeam calculated depth. The results were very encouraging at some places, whereas a noticeable difference is reported in other areas. But an acceptable overall agreement is found between the two measurements with an average difference in depth of ~30 cm.

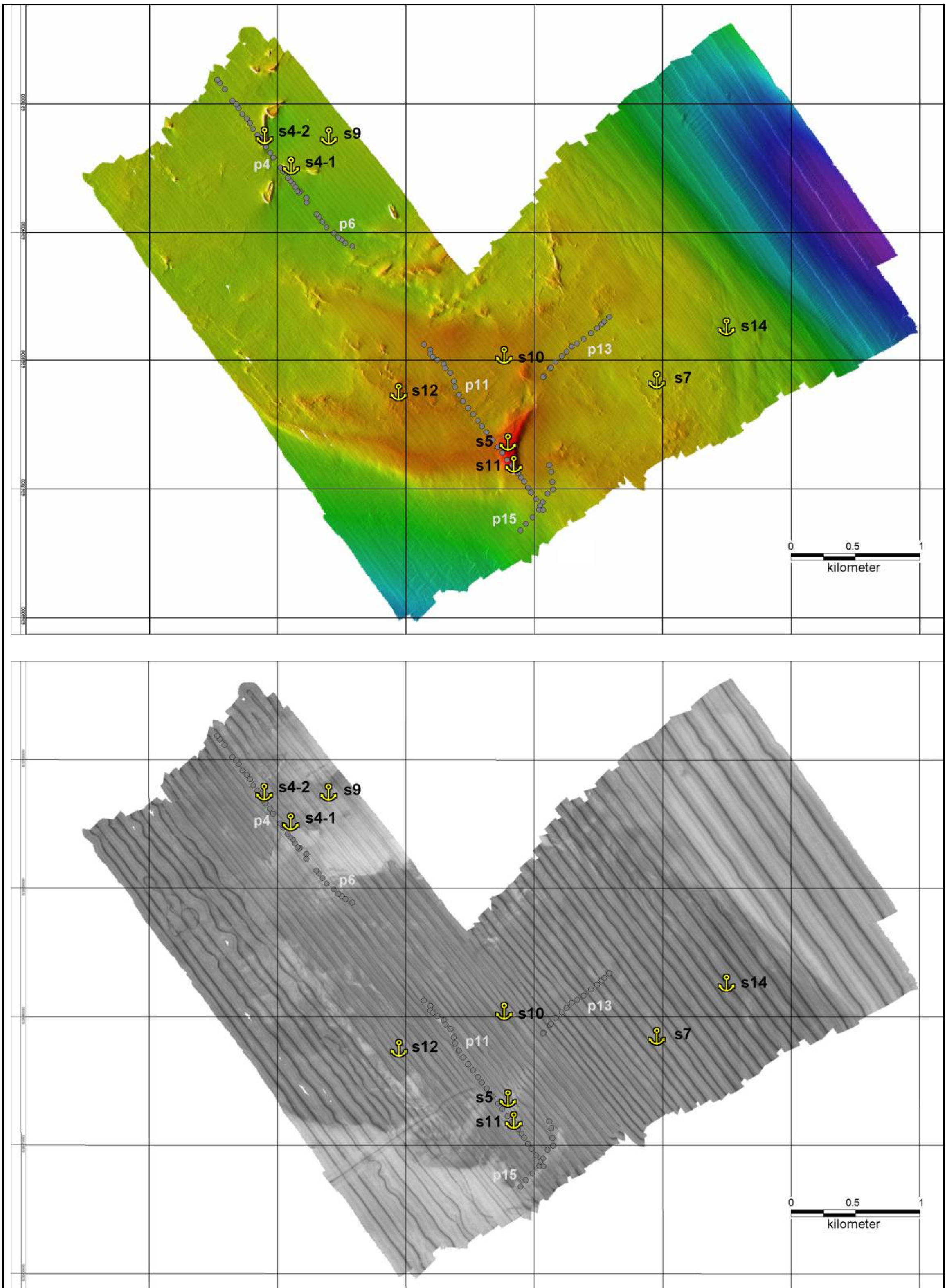


Figure 5. Multibeam bathymetric map (upper) and backscatter sidescan mosaic (lower) of the surveyed area with ground truth station indicated. The water depth interval is approximately between 3.5 and 42m. Dots = paravane tracks with track numbers in white; Anchors = point dive stations with station numbers in black.

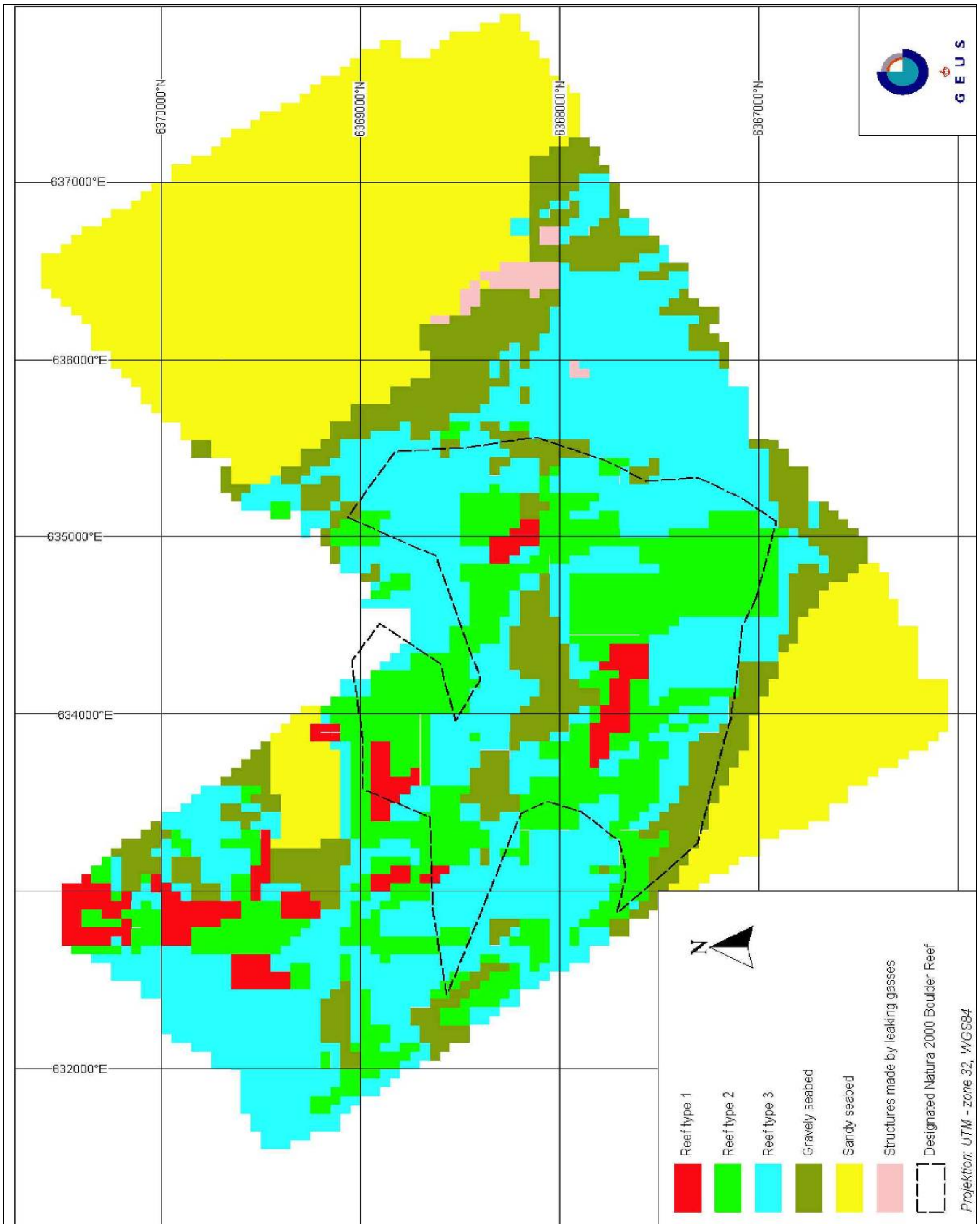


Figure 6. The seabed habitat classification map. The present designated Natura 2000 boulder reef area (dashed line) covers only partly the newly mapped reef area.

The ground truth depth measurements, when accurately executed, can be considered as a robust calibrator to the multibeam bathymetric results. The sediment ground truth results considerably agree with the scattering map interpretation, except in few rather important areas. A thorough investigation has been carried out to find the reason for discrepancy. A number of observed discrepancies were noticed during interpretation, these are:

1. Depth discrepancies: Relatively large difference in depth registered at few points. It appears that when the diver climbs a large and high stone, he sometimes do not wait for the depth measuring system to settle down before taking the reading, so errors can appear. Also sometimes, the drift by currents is so high that the reported position is not precisely accurate and can cause an error, especially in areas covered by large stones causing a hummocky seabed.

2. Substrate discrepancies: The sidescan backscatter interpretation, in general, is very consistent with the ground truth observations from the diving paravane when the areas are composed of soft sediments with no or some scattered stones. However, some controversial results have appeared. When one study the bathymetric map of a specific area in the southern part of Læsø Trindel an elevated accumulation of large stones that comprise an elevated rough seabed (a boulder reef) is noticed. The sidescan backscatter reveals a low backscattering value, which is much less than the expected for a hard bottom. Inspecting the ground truth results, one can clearly notice that this particular area is stony but 100 % covered with vegetation. The low sidescan backscattered signal in this area is presumably due to the presence of this thick vegetation cover that obviously obscures the stones. The reported vegetation cover is reported 5 % cover of stones and 15 to 5 % vegetation coverage respectively.

6.2.5 Discussion

Boulder reefs (HD Annex 1, 1170 Reefs). Experiences from our analysis of the sidescan sonar data has demonstrated that a minimum diameter of particles in the order of 25 cm can be detected under optimum conditions, i.e. no vegetation cover to blur the shape of individual objects. The acoustical detection limit in the magnitude of 25 cm is in agreement with the definition of 'boulders' – in the geological sense – as particles of a diameter above 256 mm (Wentworth 1922).

In the classification of the Læsø Trindel seabed habitats it is found that the three newly defined types of boulder reefs frequently borders to each other rather than bordering to sandy seabed types as stated by Dahl et al. (2003). The new classification does not define a minimum sediment diameter, because the minimum diameter relevant for macroalgae vegetation or epiphyte community depends on the long-term stability of the substrate. The long-term stability of e.g. smaller stones depends on the energy level (wave exposure and/or current strength) on the individual site. Neither do the above definitions take the photic depth of the features into account though differences in biomass can be expected between similar features depending on the depth of the photic zone. Where relevant photic depth as determined by the lower distribution limit of perennial macroalgae should be used to delineate and distinguish between various reef areas. However, no reefs below photic depth were identified during this field survey.

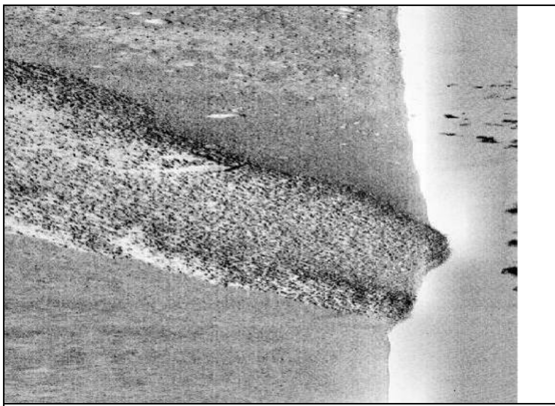


Figure 7a. Habitat type reef (1170). Sidescan picture showing the transition from a flat, sandy seabed at the bottom of the picture to an elongated reef type 1 structure. Approximate dimension of the sections: Height 150 m and width 50.

Figure 7b. Photo showing reef type 1 on top of the elongated bank structure. The boulders have a dense cover of brown and red algae like *Dilsia carnosa*. (Photo: Jan Nicolaisen).

At Læsø Trindel, in general, the kelp forest is characterised by *Laminaria hyberboria and Laminaria digitata* where the boulder reef forms well-developed cave-forming structures (Reef type 1). Underneath the brown algae canopy foliose red algae such as *Phycodrys rubens, Membranoptera alata, Dilsea carnosa* and filamentous tufts such as *Ceramium rubrum* and *Coralina officinalis* typically is present as well as the foliose brown algae *Desmarestia aculeata.* Where the reef area is characterised by scattered large boulders and smaller stones (Reef type 2 and Reef type 3) with pebbles and gravel dominating the seafloor in between the boulders, the large boulders are covered with large the kelp *Laminaria hyberboria,* while the smaller boulders and stone are dominated by the kelp *Laminaria saccharina,* filamentous brown algae such as *Desmarestia viridis* or foliose red algae such *Dilsea carnosa*. The pebbles have no cover of large algae though various encrusting species might be present.

Figure 8a. Habitat type reef (1170). Sidescan picture showing scattered boulders on a flat, predominantly gravely seabed (reef type 3). The largest boulders in the picture raise 3-4 m above the seabed. Approximate dimension of the sections: Height 150 m and width 50m.

Figure 8b. Photo showing a stone wall with a variety of species. (Photo: Jan Nicolaisen).

Submarine structures made by leaking gases (1180). The distribution of the submarine structures made by leaking gases in the northern Kattegat area is directly linked to methane seeps in shallow waters. They form spectacular submarine landscapes due to carbonate-cemented sandstone structures, which are colonised by brightly coloured animals and plants (fig. 10). In the Northern Kattegat evidences indicate that these formations cover up to 500 m^2 of the seabed and consist of pavements, complex formations of overlying slab-type layers, mushroom like or vertical pillars up to a height of 5m high above the surrounding seabed. The carbonate cement consisting of high-magnesium calcite, dolomite or aragonite indicates that it originated from a microbial methane oxidation (Laier et al. 1992, 1996). The methane most likely originated from the microbial decomposition of plant material deposited during the Eemian and Early Weichselian periods 100.000 to 125.000 years before present. It is believed that the cementation occurred in the subsurface and that the rocks were exposed by subsequent erosion of the surrounding unconsolidated sediment.

The formations are interspersed with gas vents that intermittently release gas, primarily methane. Many animals live within these formations in holes bored by sponges (example of typical species), polychaetes (example of typical species) and bivalves (example of typical species). Within the sediments surrounding the seeps there is a poor metazoan fauna, in terms of abundance, diversity, and biomass. This may be a result of toxicity due to hydrogen sulphide input from the gas escape.

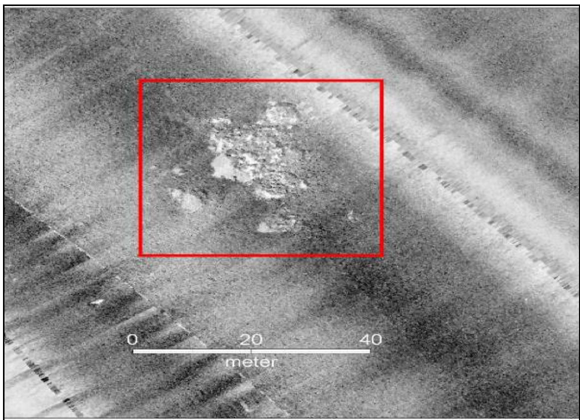


Figure 10a. Habitat type (1180) Submarine structures made by leaking gases. Sidescan sonar mosaic from Læsø Trindel showing the carbonate cemented sandstone structures rising from the surrounding sandy seabed. The size of the red box is 40 x 40 meter.

Figure 10b. Photo from The Læsø Trindel showing the Habitat type (1180) 'the bubbling reef' with a high diversity of species present. Note the escaping gas bubbles in the water. Photo: Jan Nicolaisen.

The analysis of the sidescan and multibeam data has discovered a hitherto unknown occurrence of structures made by leaking gasses. From the sidescan mosaic the Læsø Trindel area has been delimited from the surrounding seabed dominated by sand and gravel. The subsequent ground thruthing has confirmed the existence of the structures and furthermore characterised the detailed structure of the feature including a description and assessment of the biota. It can be concluded that the acoustic method is very suitable for discovering new areas of structures made by leaking gasses. Once the acoustic characteristics of this type of structures were recognised is allows the geologist easily to register and delimit similar areas. The existing knowledge of the performance of structures made by leaking gasses indicates that they may occur as either reef-like structures or as cemented sandy plates within or at the seabed. So far, the acoustic method has demonstrated itself as a promising method for mapping structures made by leaking gasses emerging from the seabed. We still need to demonstrate its efficiency to mapping the plate-like structures. Finding more structures by diving alone has a long perspective due to the low coverage and limited visibility by this method.

6.2.6 Conclusion and perspectives

The differences between the various available remote acoustic sensing techniques, irrespectively of the post-processing being used, often make it difficult to judge, which sonar device is most suitable to the actual need. In the context of habitat mapping the geophysical characteristics of the seabed area is essential; hence it allows the wide-scale geology and the modern-day sedimentary processes to be understood. Based on the understanding of the sediment dynamics and geological structure the marine scientist produce maps of the seabed, which help managers of the marine environment to predict the impacts on those habitats which may be of high nature conservation and ecological value.

The techniques applied in this study have demonstrated the usefulness of combining acoustic methods with ground truthing to produce maps revealing the physical characteristics of the seabed. The multibeam swath bathymetric and the sidescan sonar devices used in the Læsø Trindel study are the most highly developed and versatile available systems. They offer great data control and supporting real-time visualisation of sonar data as true geo-corrected mosaic seabed maps. Sidescan sonar provides information on sediment texture, topography, bedforms, and other discrete objects at the seabed (e.g. boulders). The multibeam data system provides depths of centimetre resolution. Multibeam data processing enhances subtle aspects of relief elements through shading techniques for an understanding of erosive and depositional processes. The maps produced and interpreted such as seabed geology, relief and processes provide the foundation for assessment and mapping of seabed habitats.

There is a wide range of technologies capable of mapping the seafloor including acoustic systems and ground truth devices and methods. The choice of system will depend on survey objectives and scale of the area to be mapped. For inshore areas <50 m water depth where identification of small (<10 m) habitat features may be required, a combination of multibeam echo sounder and sidescan sonar ensures that both quantitative high resolution bathymetric data (1-10 cm scale) and qualitative, high-resolution habitat relief data (decimetre resolution) is obtained (Kenny et al. 2003).

A number of conclusions may be put forward in relation to the advantages and disadvantages of the various devices for habitat mapping. Nevertheless, based on numerous experiences from biotope and habitat mapping reported the recent years there no doubts that the combination of acoustic systems and ground truth verification is recommended. Swath systems offer the availability to discriminate small habitat features (0.3 – 1 m) together with providing information on sediment dynamics and geological development make them most suited for detailed biotope mapping. By contrast, single beam echosounder systems are most useful for detecting gross differences in substrate type i.e. between rock, sand and mud, but often requires intensive ground truthing limiting their utility as a tool for broad-scale biotope mapping.

For broad-scale mapping of habitats (>1 km^2) the sidescan sonar combined with the multibeam echo sounder is considered to be the most cost-effective means of discriminating sediment types and dynamic processes. For small-scale habitat classification (> 1 km^2), high-resolution sidescan sonar, underwater cameras or videos, and grab-sampling methods are considered to be the most appropriate mapping tool.

The interpretation of the sidescan mosaic and multibeam data from Læsø Trindel has demonstrated that the sandy and hard seabed unambiguously can be distinguished. However, the acoustic maps (e.g. the sidescan sonar mosaic) reflecting the variable acoustic properties of the seabed has to be 'ground truthed' by seabed sediment samples for calibrating the acoustic classes into sediment types and biological samples to characterise the flora and fauna present. Dependent on the complexity and distribution of the acoustic classes a relevant sampling programme will ensure the optimum amount of samples to verify the present seabed types.

6.2.7 Acknowledgement

The project has been conducted by GEUS and The Danish Forest and Nature Agency as a part of the Danish obligations to the EU BSR INTERREG IIIB project BALANCE. The ground truthing by diving and the biological investigations was conducted in corporation with Jan Nicolaisen, Orbicon A/S. Thanks to Jan for his great effort.

6.3 PA3. Mapping of the NATURA 2000 Annex 1 habitats in Finnish and Swedish waters using GIS analyses

Authors: Sandra Wennberg, Anna Nöjd, and Cecilia Lindblad.

6.3.1 Introduction

The EU Habitat Directive is a Community legislative instrument in the field of nature conservation that establishes a common framework for the conservation of wild animal and plant species and natural habitats of Community importance. It provides for the creation of a network of special areas of conservation, called NATURA 2000, to 'maintain and restore, at favourable conservation status, natural habitats, and species of wild fauna and flora of Community interest'. Many of the actions associated with the Habitats Directive require extensive information on the spatial distribution of the habitat types. Assessments of favourable conservation status require an estimate of how much of the entire habitat type is protected. Little actual information exists on the spatial distribution and total cover of the Annex I habitats to base assessments on.

In the BALANCE pilot area 3 eight types of the EU Habitat Directive Annex I marine habitats can be found (including those coastal habitat types, which include the subtidal part). The coastal habitats that, in the Baltic, occur primarily above the waterline, such as sandy beaches, stony banks, and vegetated cliffs were excluded from this study.

Aims. The aim of the modelling exercise was to create maps of the spatial distribution of the EU Habitat Directive Annex I habitats: 1) 1110 Sublittoral sandbanks, 2) 1130 Estuaries, 3) 1150 Coastal lagoons, 4) 1160 Large shallow inlets and bays, 5) 1170 Reefs, 6) 1610 Baltic esker islands and 7) 1620 Boreal Baltic islets and small islands. No attempt was made to model type 1650 Boreal Baltic, as they are distinguished by a sill at the mouth, which is not identifiable from the coarse depth models available. The maps should be comparable over the nation border between Sweden and Finland.

6.3.2 Material and Methods

The study area. The study area for mapping the Annex I habitats is the entire pilot area 3 (chapter **), although lack of substrate data limits the study area of 1110 to the Swedish part and a small section of the Finnish part.

Data sources. The GIS-data sources used are shown in table 1. From these an additional seven datasets were derived (tab. 2). The few data available used for validation and evaluation are described in chapter 3.**.4.

Table 1: Data sources

Coutry Ref. no.	Dataset	Spatial scale / cell size	Source	Data owner
[F1]	Shoreline with land, sea, lakes and rivers	1:20,000	Basic map	NLS
[S1]	Shoreline with land and water	1:10,000	GSD-property map	SLS
[S2]	Shoreline, land, water and elevation curves	1:50,000	GSD-topogephic map	SLS
[S3]	Lakes and rivers with waterflow	1:150,000	GSD-general map	SLS
[F2]	River flow	N/A	Flow monitoring database	YH
[F3]	DEM	Scale?/25m	Elevation curves and shoreline in topographic map	NLS
[F4]	Exposed bedrock	1:5,000 – 1:10,000/25m	2001 topographic database	NLS
[F5]	Land use	1:100,000/25m	CORINE Land Cover 2000	YH
[S4]	Land use	1:50,000/25m	GSD-Land and Vegetation Cover (National CORINE)	SEPA
[S5]	Depth surfaces and sufs	1:50,000	Nautical chart	SMA
[F6],[S5]	Depth points, submerged and surface rocks	1:50,000	Nautical chart	FMA/SMA
[F6]	Depth isolines	1:50,000	Nautical charts	FMA
[F7],[S6]	Bottom substrate	1:100,000/100m (F) 1:100,000-500,000(S)	Cato et al. (2003) (S)	GTK/SGU
[F8],[S7]	Soil type on islands and	1:1,000,000/200m(F) 1:100,000 (S)	Soil map	GTK/SGU
[F9],[S8]	Wave exposure	25 m	Isaeus 2004	YH/SEPA
[F10]	Secchi depth	N/A	National water quality database	YH
[S9]	Coastal exploitation	1:10,000/25 m	Smedberg 2006	SEPA

[S9]	Satellite data	25m	Landsat/Image2000	SEPA
[S10]	Arerial photo	1m	GSD-ortophoto	SLS

NLS/SLS = The Finnish/Swedish National Land Survey; FMA/SMA = The Finnish/Swedish Maritime Administration; GTK/SGU = Geological Survey of Finland/Sweden; YH = Finnish Environmental Administration, SEPA = The Swedish Environmental Protection Agency

Table 2: Derived datasets

Country Ref. no.	Dataset	Spatial scale / cell size	Source, used information
[F11],[S11]	Depth model	25m	[F6],[S5] Depth as points and isolines, shoreline, elevation isolines
[F12],[S12]	Land and sea rasters	50, 15, 10m (F) 25, 15 m (S)	[F1],[S1],[S2] Vector shoreline
[F13]	Esker islands, polygon	1:20,000	Digitised from National esker survey (Kontturi & Lyytikäinen, 1987)
[F14]	Percentage of land in a 5 km neighbourhood	50m	[F12] 50m land and sea raster
[F15]	Archipelago zones	50m	[F14]
[F16],[S13]	Benthic terrain model	25m	Depth models
[F10]	Photic depth model	50m	Linear model of the value in [F14] and average July secchi depth.

The model of Photic depth was created on the Finnish part of the pilot area. To predict secchi depth a simple linear regression model was used based on the value in each cell of the percentage of land dataset [F14]. Secchi depth was found to have a negative correlation with the percentage of land in a five kilometre radius neighbourhood (fig. 1). Photic depth was estimated at twice the secchi depth.

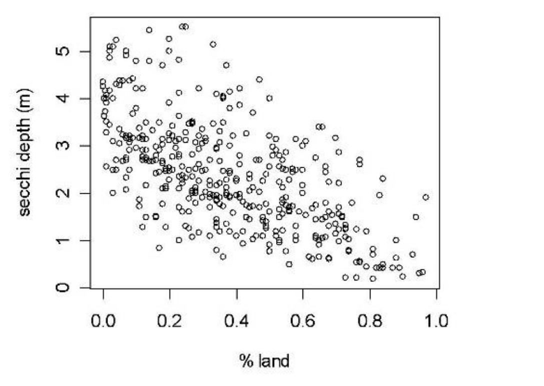


***Fig. 1** Secchi depth against the percentage of land within a five kilometer radius..*

Benthic terrain models were created using different methods in each country although both terrain models distinguish depressions, elevations, and flat areas in the bathymetric data. Finland used the Benthic Terrain Modeller, an ArcGIS extension available from the NOAA website (http://www.csc.noaa.gov/products/btm/). The broad scale Bathymetric Position Index (BPI) tool was used to detect broad scale terrain variation appropriate for locating habitat features such as reefs. The BPI is created using an annulus shaped neighbourhood with a calculation comparing a cells depth value to that of surrounding cells to measure if it is on average

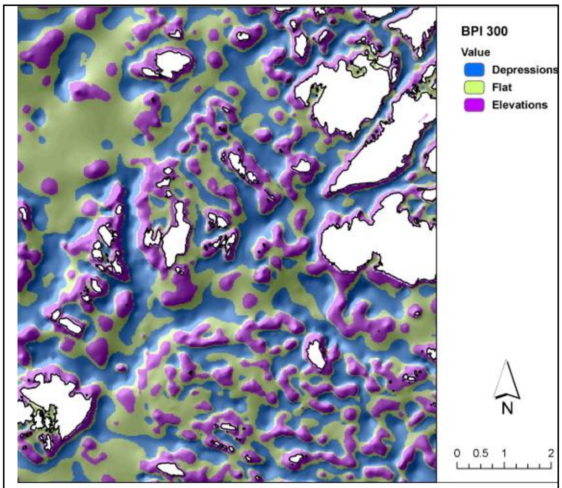


***Fig. 2** Benthic terrain model draped over a hill-shaded depth model*

higher or lower than its neighbours. Sweden used the tool Focal Statistics to calculate a mean depth for a circular neighbourhood and then subtracted the result from the actual depth. In both countries the 25m raster depth model and a radius of 300m for the neighbourhood was used.

Habitat analysis criteria. The selection criteria for each habitat type were derived from the Interpretation manual of European Union Habitats (EC 1999) and the national descriptions of the habitats. No actual datasets that outline the habitats area available. The criteria used to delineate the habitats in the models are described in the table below.

Table 3: The criteria used to delineate potential Annex I habitats

Habitat	Depth	Size	Exposure to wave action	Sub-strate	Freshwater flow	Other
1110 Sandbanks	<= 30m[1]			Sand >= 70%[2]		Elevated from surrounding sea-floor[2]
1130 Estuaries	<= 3m at the mouth				At least 1 river with >= 1km^2 watershed[1]; >=2 m^3/s[2] flow	Not on an open coast Reed beds present
1150 Lagoons	Max <=6m	< 30 ha			No river inflow	
1160 Large Bays	<20% of area >15m (6m*) deep	>= 20ha (>= 100ha*)			No river inflow	Wider than long at least 1:1
1170 Reefs	<= 6m[1] <= photic depth[2]		Sheltered and higher (algal zonation)	Hard substrate		Elevated from surrounding sea-floor
1610 Esker islands	<= 10m[1] <= photic depth[2]			Sand or moraine (>= 50% of cover)		Submerged part elevated from the surrounding sea-floor[2]
1620 Boreal Islets	<=6m[1] <= photic depth[2]		Sheltered and higher[1]			In outer archipel-ago zone[2]

* According to Finnish guidance on identifying Annex I habitats
[1)] Approach 1, [2)] Approach 2

GIS Analyses (methods, software, and routines). The methods are based on tools in ArcGIS with the Spatial Analyst extension. Sweden also used the model maker in ERDAS Imagine and Finland used the Benthic Terrain Modeller extension from NOAA. The resulting layers are presence maps of the Annex I habitats.

The GIS-analyses are different for each habitat and described separately. For some habitats the analyses also differ between the countries, described as approaches. The table below gives an overview of the data used to map each habitat. The numbers in the table and of approaches links to the countries.

Table 4. Predictions layers used in each model. The scale of the input data and and information used in each layer are described in each habitat model.							
Prediction layer	**1110**	**1130**	**1150**	**1160**	**1170**	**1610**	**1620**
Sea	1, 2	1, 2	1, 2	1, 2	1, 2	1, 2	1, 2
Lakes			1, 2				
Rivers		1, 2	1	1, 2			
Land		1, 2	1, 2	1, 2		1, 2	1, 2
Elevation			1, 2				
Depth model	1, 2	1, 2	1, 2	1, 2	1	1	1, 2
Terrainmodel/BPI	2				1, 2	2	
Wave exposure					1. 2		1, 2
Bottom substrate	1, 2						
Subsurface and surface rocks					1, 2		
Exposed bedrock shores					2		
Soil type					1, 2	1	
Land cover: forest							1, 2
Land cover: wetlands		2					
Land cover: estuaries		1					
Land cover: lagoons			1				
Coastal exploitation			1				
Satellite images /aerial photos		1					
Photic depth model					2	2	2
River flow		2					

1) Sweden, 2) Finland

Sublittoral sandbanks (1110)

Approach 1: Submerged areas classified as sand and with a maximum depth of 30 meters are considered sandbanks.

Approach 2: The BPI dataset was used to select cells that were clearly elevated above their surrounding seafloor. The extracted cells were grouped into continuous patches, or "mounds". The total area and the area of sandy substrate on each mound were calculated. Mounds larger than three pixels that consisted of a minimum of 70% sand and were at least partly located above 20m depth were selected as sandbanks.

GIS Analysis of Estuaries (1130)

Approach 1: Sheltered areas with a freshwater influx (from a river with a watershed > 1 km^2) are a by-product of the analyses of 1150 and 1160. These areas of potential estuaries are visually classified with reference data from satellite images, aerial photos and maps. Objects were deselected if they are artificial pools, if the river mouth is located on an exposed coast or in waters deeper than 3 m. The remaining objects are manually outlined in GIS. The outline towards the river is at the point of the river mouth.

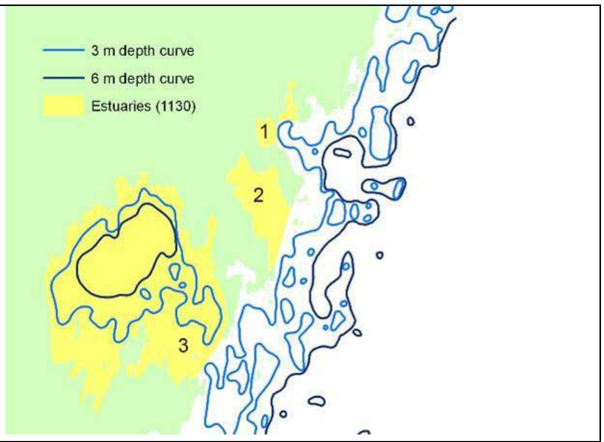


Fig. 3 Three examples of how the outline of estuaries towards the sea was drawn (se text).

The outline towards the open sea is either at a threshold (Fig 3, 3) or the 3 meter depth curve (Fig. 3, 1) but not further out than were "sheltering land" ceases (Fig. 3, 2). Boundaries were drawn with special emphasis on placing the inner border up stream and the outer border towards the sea correctly. Between these two borders the polygon was drawn well up onto the surrounding land and the polygon was then cut against the shoreline. All estuaries from the Swedish Land Cover [S4] are included.

Approach 2: Embayments or basins surrounded by land or shallow water of a depth of 3 metres or less, with an input of fresh water and sediment from at least one river with an average flow of 2 m^3/s or above and with wetlands present were selected using the following methodology. Separate reed lined shallow "basins" were extracted by initially splitting the depth data [F11] into two categories: >= 3m and > 3m. The waters deeper than 3m were reclassified into the same class as land [F12]. The land and "deep" water class was expanded, resulting in many separate bodies of shallow water. Wetlands were extracted from Corine 2000 landcover data and added to the shallow waters to help define the edges. Potential estuaries were selected from the shallow reed lined basins based on rivers running into them.

Estuaries have priority over the other enclosed water body types, so areas selected as estuaries can not be selected as any other type (both approaches).

***Fig. 4a.** Estuaries Photo: NN, Affiliation.*

***Fig. 4b.** Coastal lagoons. Photo: NN, Affiliation.*

Coastal lagoons (1150)

Approach 1: For gloes, lakes were selected from the older map ([S1] date from 1990-1996). A buffered shoreline was produced by expanding the sea surface one pixel up towards land ([S12], 15 m pixel size). Lakes that intersect with the new shoreline are located between the 5 meter elevation curve and 6 meter depth curve are gloes. For partially separated lagoons, land ([S12], 15 m pixel size) was expanded one pixel and sea-areas that had become separated from the "larger" sea were identified. The identified areas are expanded one pixel back to its original outline. From the results three more selections were made; 1) the lagoons are less than 6 m deep, 2) the lagoons shall not intersect with a shoreline that is exploited and 3) they shall not be larger than 30 hectares. The operation is done on two generations of maps ([S1] dates 1990-1996 and 2006). Lagoons that intersect the recent sea are saved as partially separated lagoons; those not intersecting with the recent sea are merged with the Gloes. All lagoons less than 30 hectares from the Swedish Land Cover were included.

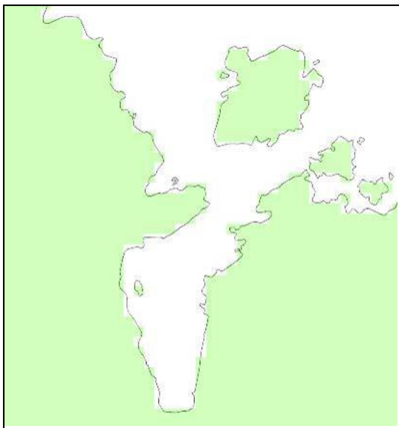


Fig. 5a. *The analysis of lagoons area based on a raster (Land in green).*

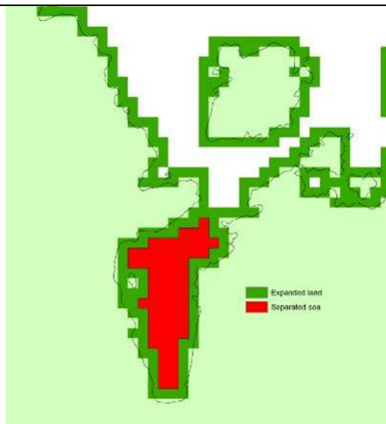


Fig. 5b. *Step 1 and two in the analysis area identical in both countries (se text for details)*

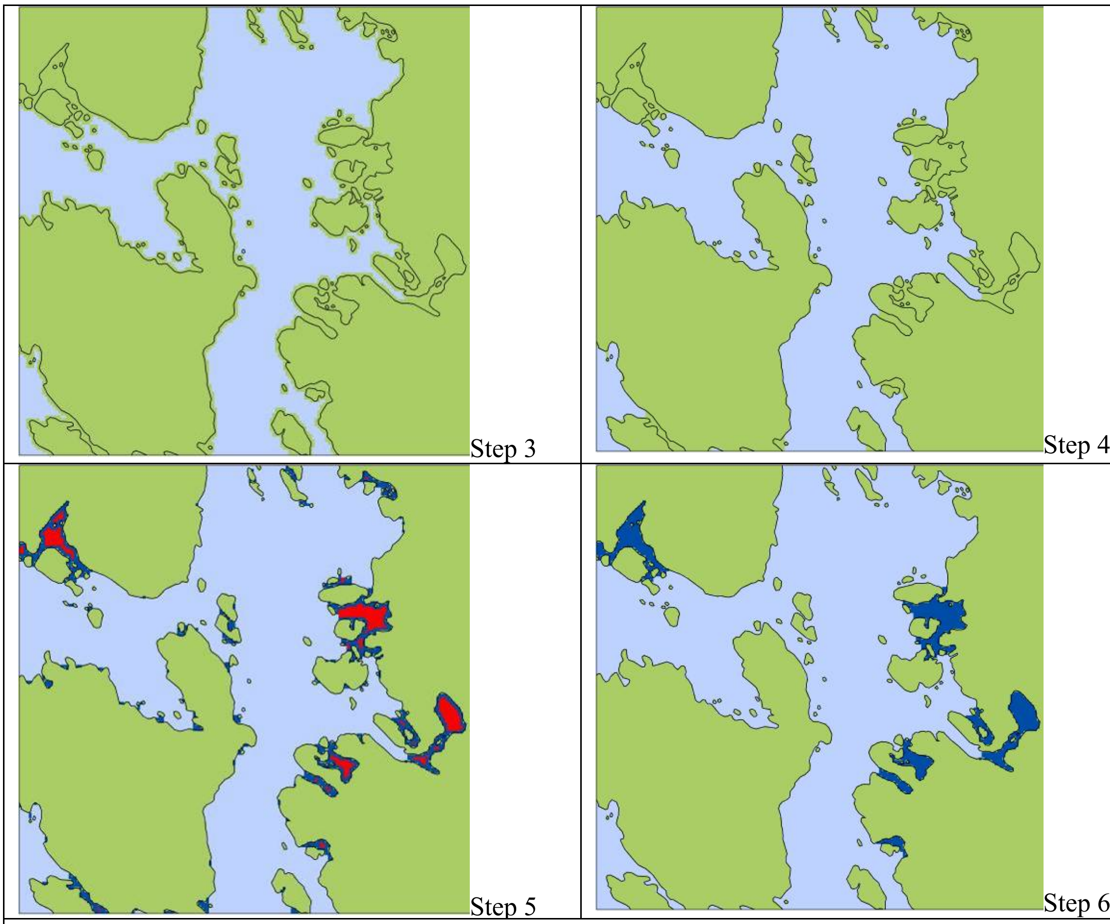


Fig. 6. *Steps 3-6 in the selection of bays almost separated from the surrounding sea in Finland. (See the text for details on the steps.). Steps 1 and 2 as in Sweden*

Approach 2: For gloes, lakes were selected that intersected a 30m buffer of the shoreline [F1] and were in a zone below 5m elevation that intersected the sea. The gloes dataset was rasterised into 10m cells. The analysis of "lagoons" was done for both the 10m and 15m rasters [F12]. *Step 1 and 2* are the same as in the Swedish analysis: land was expanded by one pixel and the separated "lagoons" were extracted. *Step 3:* The "lagoons" and the expanded land were reclassified into the same class. *Step 4*: the land (and "lagoon") was shrunk back by one pixel. *Step 5*: "lagoons" were extracted into a separate raster layer using a sea mask. This results in many small errors caused by the 'expand' and 'shrink' operations. *Step 6*: to remove the errors only those resulting enclosed bays that were more than 3 pixels in size and had a corresponding bay in the results of the original expand analysis were included in the final potential lagoons dataset (Fig. 6).

The final lagoons dataset included the coastal lakes and those "lagoons" that have a maximum depth of six metres and are no larger in size than 30 hectares.

Large shallow inlets and bays (1160)

The number of directions out of eight (N, NE, E, SE, S, SW, W, and NW) in which sea pixels in [F12]/[S12] are within 1 km of land was analysed. In this part of the analysis land areas smaller than 1 hectare are sea as their sheltering effect was considered too small. The output raster only included sea areas that fulfilled the criterion of the number of directions there had to be land. Patches of open water surrounded by "bays" were included in the bays. Areas intersecting with rivers were deselected. The partially enclosed bays and those "lagoons" that are too large or too deep to be lagoons were included. Finally areas intersecting with land, with less than 20 % of the area being deep waters (>15 m) and a total area larger than 20 hectares were selected.

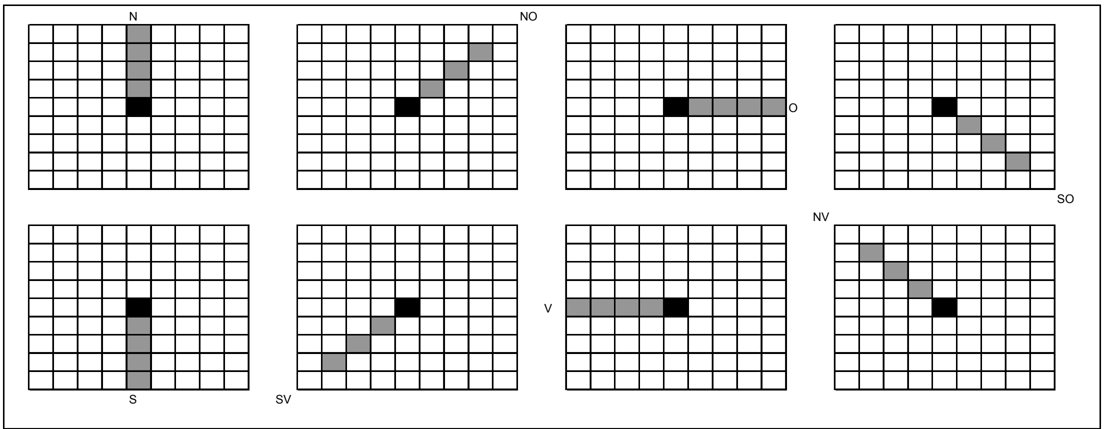


Fig. 7. *The figure shows a sketch of the GIS-model that identifies sheltered sea. The neighbourhood (gray) around each pixel (black) is searched eight times, once for each direction. The analysis returns the maximum value that equals 1 if land is within the neighbourhood and 0 otherwise to the centre pixle. The eight results are then summarized. In the analysis the gray area is 1 km long.*

Approach 1: In Sweden areas with land in 5 of 8 directions were accepted as sheltered.

Approach 2: In Finland the analysis was done using a wedge shaped neighbourhood specified with a one degree difference in the start and end angles to achieve as thin a wedge as possible. Land had to be found in at least 7 directions. Out of the first set of bays a subset was further defined that conformed to the Finnish criteria of larger than 100 hectares and less than 20 percent of area deeper than 6 metres.

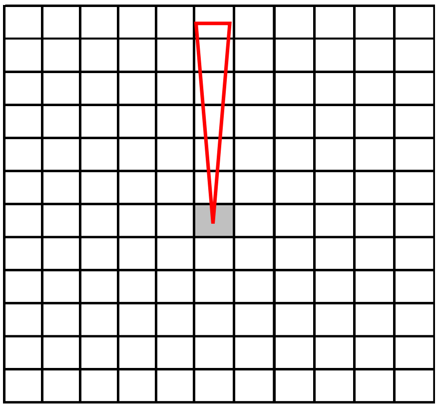


Fig. 8a. *Wedge shaped neighbourhood analysis in ArcGIS*

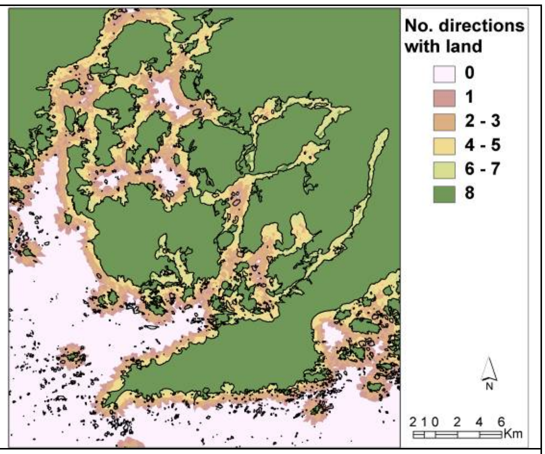


Fig. 8b. *The number of directions with land.*

Lagoons are prioritised above large bays, so an area classified as lagoon cannot be classified as large bay. These two overlap e.g. where a part of a bay is a lagoon.

Fig. 9a. *Large shallow inlets and bays Photo: NN, Affiliation.*

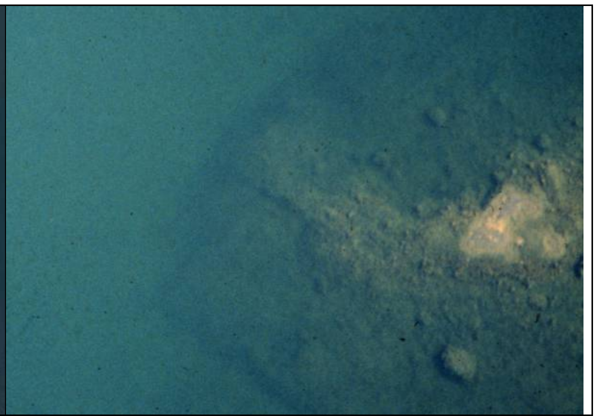


Fig. 9b. Reefs. Photo: NN, Affiliation.

Reefs (1170)

Areas with potentially hard substrate that rise from the surrounding seafloor and have exposures within in a range from very sheltered to exposed (4,000-1,000,000 m^2/s) are selected as reefs.

Approach 1: Potential hard bottoms were selected from two inputs. First depth areas between 0 and 6 meters that intersect with exposure class "sheltered" or higher was selected. Second, surfs and sub surface rocks were analysed in "Neighborhood Statistic" sum for a circle with a 150 radius. Values larger than 1 in the result together with the shallow areas in exposed positions are regarded as potential hard bottom. The layer of potential reefs was created by selecting peak areas ([S13]) overlapping with potential hard bottom areas. Areas deeper than 10 m were excluded as the input depth data were considered too inaccurate. Objects intersecting land are excluded (saved as a separate layer, if criteria will change) and areas mapped as sandbanks are removed. Reefs intersecting with the layer of "Islets and small islands" are removed from the reef-result and included in the "Islet" result. Finally only reefs.

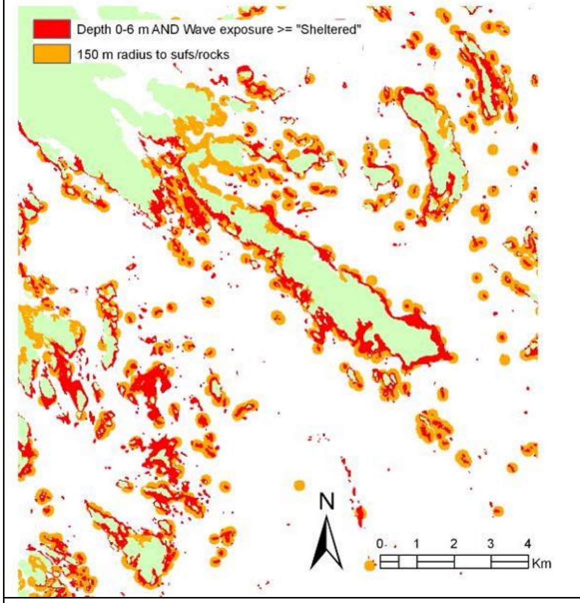


Fig. 10a. The selection of potential hard bottoms are a combination of shallow exposed areas (red) and the neighbourhood of 150 m around surfs and subsurface rocks in nautical charts (orange).

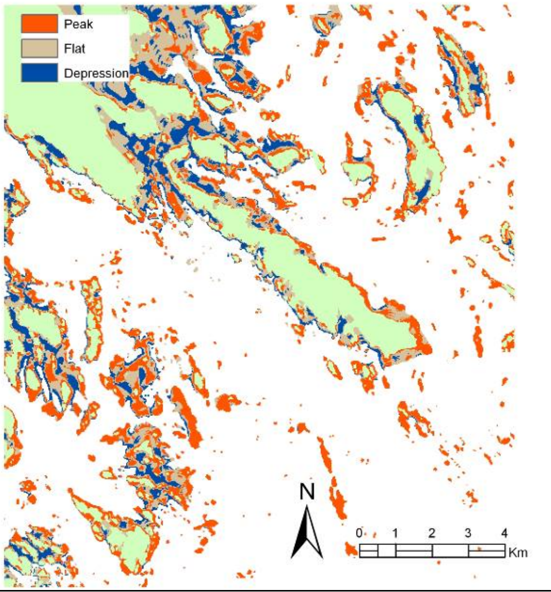


Fig. 10b. Peeks were separated from depressions and flat areas by subtracting the actual depth from an average depth within a 300 m radius. Reefs are peeks that overlap with potential hard bottoms.

Approach 2: A rock density dataset was made using the Point Density tool. The density of rocks ([F6]) within a circular 150m search radius was calculated for a 50m raster dataset. A similar dataset was made for the length of rocky shoreline ([F8]) within a 150m search radius using the Line density tool. Potential reefs were created by extracting "mounds"([F16]), i.e. areas rising above their surroundings, intersecting with areas where the density of rocks or rocky shoreline is more than zero and the depth does not exceed photic depth. Rocky elevations that intersect land are assigned to a separate subclass, to keep the data comparable with the Swedish dataset.

Baltic esker islands with sandy, rocky and shingle beach vegetation and sublittoral vegetation (1610)

Approach 1: Islands ([S1]) intersecting with glacio-fluvial material ([S7]) are selected. A buffer of 200 meter around the island is created and clipped with the depth area 0-6 meter. Finally land areas (that are not esker islands) are removed from the buffer zone.

Approach 2: Islands identified in the Finnish esker survey run in the 70s and 80s ([F13]) were automatically included as were islands digitised by the Nature Division at the Finnish Environment Institute as a part of their investigation into esker islands (Suutari et al. 2002, unpublished report). The soil map was reclassified to contain only what could potentially be "esker deposits" (classes: sand, gravel and till). Each "esker deposit" was assigned to a particular island and the combined area of esker deposits on each island was calculated. Those islands that consisted of a minimum of 50 percent of "esker deposits" were selected as esker islands. The submerged part of the islands was determined by including the "mounds" in [F16] that the island was the top of, limited by photic depth and a buffer of 200 m.

Fig. 11a. *Esker islands Photo: NN, Affiliation.*

Fig. 11b. *Boreal Baltic islets and skerries. Photo: NN, Affiliation.*

Boreal Baltic islets and small islands (1620)

Approach 1: Islands ([S1]) with no forest cover ([S5]) are selected. A majority filter operation is executed on the classified wave exposure data in order to exclude single pixels. Islands intersecting with exposure class “sheltered” or higher are then selected. The submerged part of the habitat is determined as a buffer of 200 meters around the selected islands down to 6 meters depth. Reefs in contact with the buffer zone are included in the habitat.

Approach 2: Islands ([F1]) were chosen if they were located in the outer archipelago zone ([F15] >50 % water) with no forest cover ([F4]) and no forested island within a 200m buffer in their neighbourhood. The submerged part of the island is defined according to it being topographically distinct from the surrounding seafloor, down to the photic depth within a 200m buffer.

6.3.3 Results

The results show potential areas of the NATURA 2000 habitats, one map per habitat. The habitats may overlap each other. The total area mapped as either one of the habitats is 108,000 ha and covers 7 % of the sea area of the Swedish part of PA 3. On the Finnish side the figures are 343,000 ha and 13 %. The larger figure is partly caused by the definition of reefs. If reefs intersecting islands are not included the figures are 295,000 and 11 %. The results of the analyses are set out in the table below.

Table X: Habitats Directive Annex I habitats in pilot area 3

	Number of patches	Size range	Total Area	Notes
1110 (Sandbanks)				
1130 (Estuaries)				
1150 (lagoons)				
1160 (inlets/ bays)				
1170 (Reefs)				
1610 (esker islands)				
1620 (Baltic islets)				

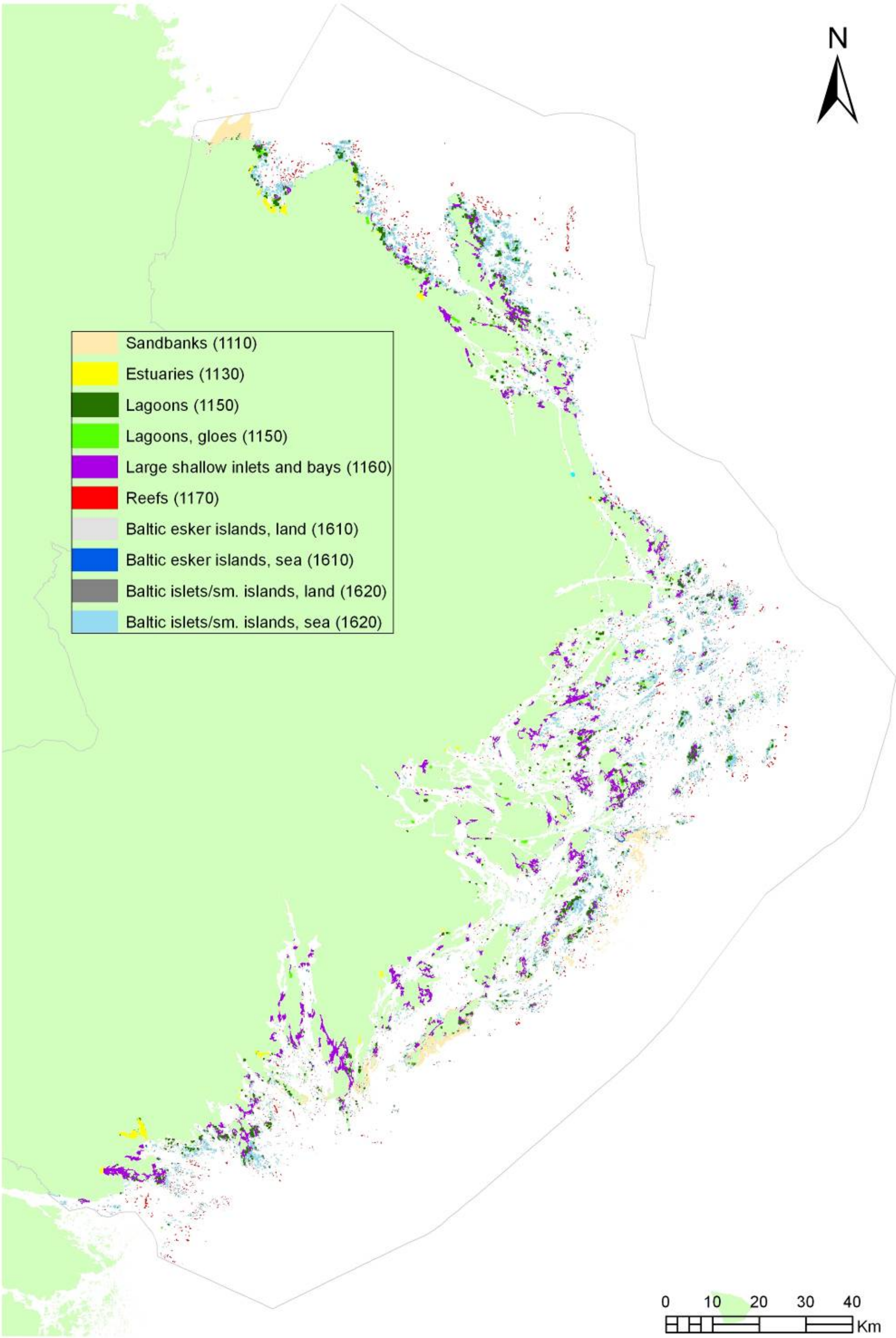


Fig. 12. *The distribution of Natura 2000 habitats according to the habitat directive Annex 1(the Swedish part of Pilot area 3). The habitats may overlap each other, in total the maps covers 7% of the sea (108 000 ha)*

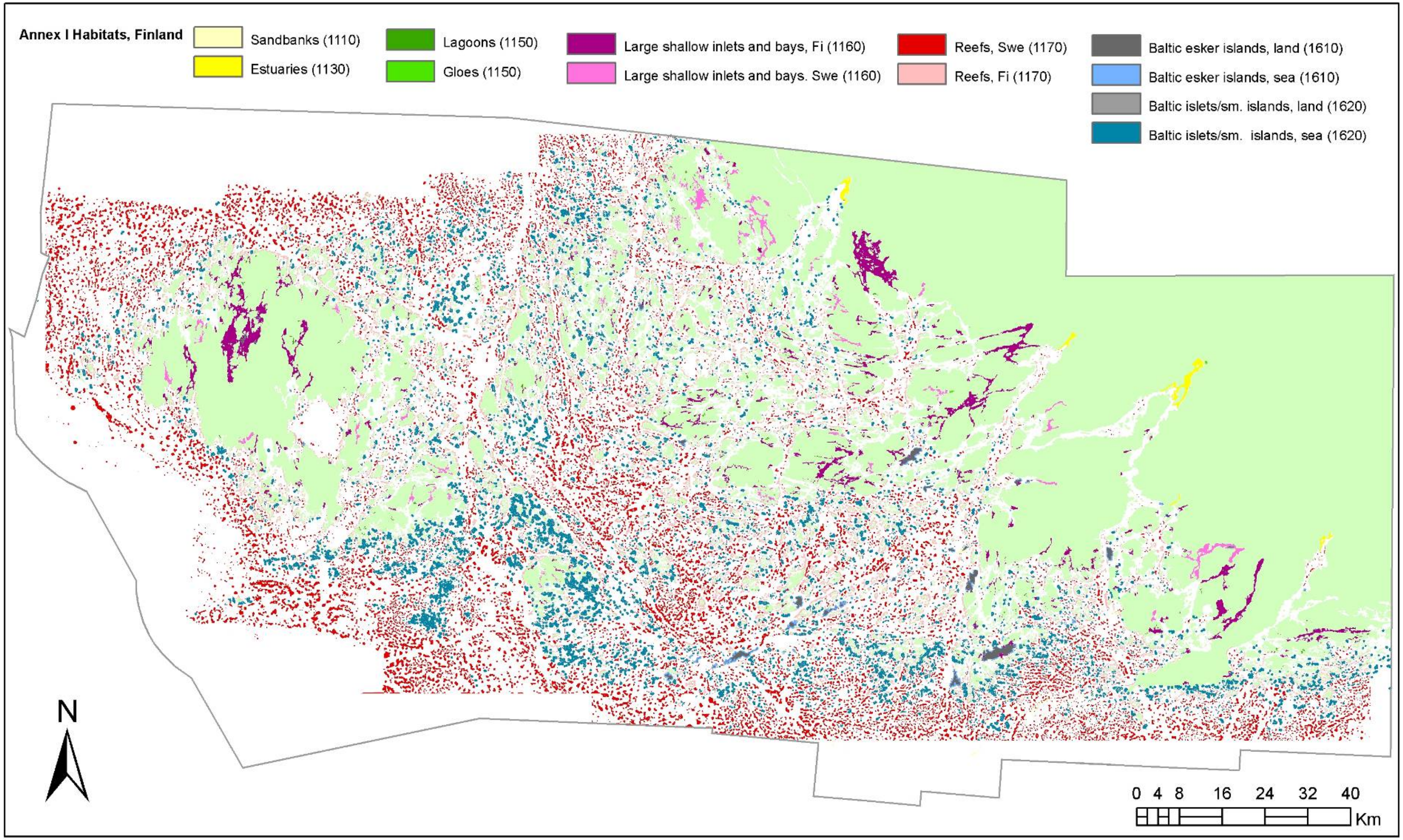


Fig. 13. *The distribution of Natura 2000 habitats according to the habitat directive Annex 1(the Finnish part of Pilot area 3*

6.3.4 Discussion

Quality of data and methods

The methods are direct and very useful for this type of habitat modelling. However, the results are very sensitive to the quality of existing data. Basic data that outlines water and land have enough quality. They are available in very detailed resolution (scale 1:20.000) that is needed to identify small habitats like small islands and lagoons, although maps in scale 1: 50.000 may well suite the purpose for most habitat modelling. The data on wave exposure and land cover have also enough quality to be used in these type of analyses. Better data on water flow from small rivers are wanted, as well as data on water quality.

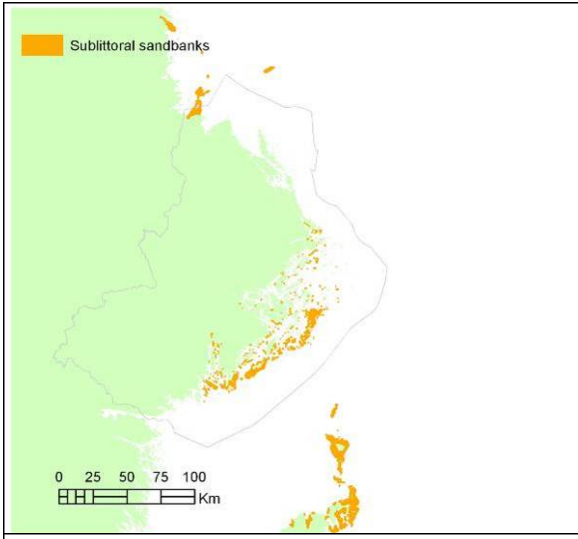


Fig. 14a. *Occurrence of sublittoral sandbanks on the Swedish side of pilot area 3 based on geological maps.*

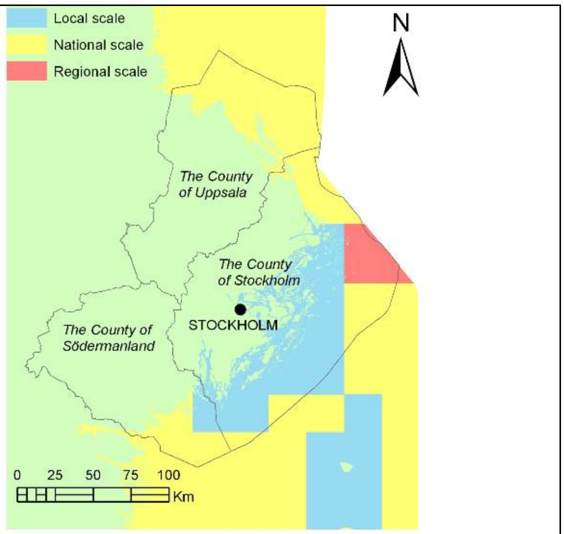


Fig. 14b. *The knowledge of where sandbanks occur is correlated to the scale of the geological survey.*

The main datasets required to produce high-quality detailed habitat maps are a high resolution map of the sediment characteristics and depth (e.g. multi beam surveys). Besides resolution, the current depth and substrate data available has two major problems,. 1) There are quite large areas with very limited or no information available due to military restrictions and 2) the shallow areas of 0-6 meter are not well outlined.

Validity of identified habitats. The validation of the directive habitats was not performed as a confidence assessment due to the lack of the real field data on habitats. Each habitat layer is discussed below.

Sublittoral sandbanks: Ground-truthing data would be required to assess the ability of this method to identify actual sandbanks and to compare the merit of including or excluding depth data in the analysis of potential sandbanks. The results have very varied quality over the pilot area (Fig. 14)

Estuaries: Boundaries are set somewhat arbitrary regarding the actual water mixing and reeds are not always included in the habitat. In Sweden there seems to be an over estimation of the size of the area and the main reason is problem to determine where water mixing ceases. In Finland the basins considered estuaries were defined by deciding on an artificial depth criterion to approximate the potential extent of freshwater influence. The actual extent is affected by both the flow of the river(s) and geographical atrributes. In both cases freshwater inflow may be relatively too small to justify the extent of the estuary, or the outer boundary may be set to far out, compared to where the influence of freshwater ceases.

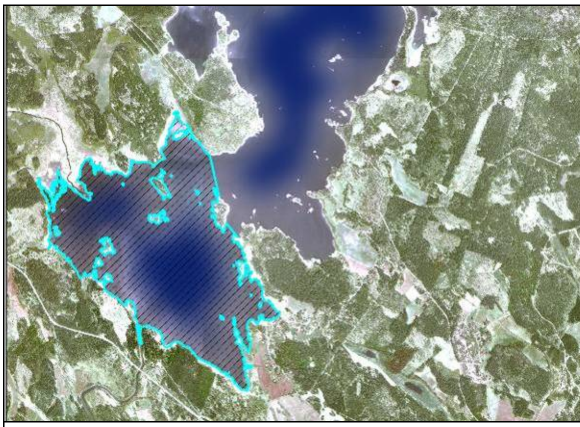


Fig. 15a. *The estuary boundary towards the sea is drawn in line with the 3 m depth curve. This seems to lead to an overestimation rather than an underestimation of the area, especially in the shallow archipelago of Uppland.*

Fig. 15b. *Towards the land the estuary is drawn at mean sea level (according to maps), in many cases this leads to that the reed belts is not included in the area (arrow). One estuary can have more than one river mouths.*

Coastal lagoons: Resulting gloes may include lakes that are not a part of the succession stages in the process of where sea becomes land (at least not in recent time). Some actual gloes identified in the field, were also not picked out by the analysis. This may be due to the uncertainty in the elevation model and its scale. The partially separated lagoon-analysis also leaves out lagoons smaller than 30 m x 30 m and "lagoons" that are narrower than 30m. The analysis may also miss and/or under estimate objects that are outlined by submerged thresholds towards the sea as depth information of an adequate resolution is not available. False lagoons can be crated by the analysis in the outer archipelagos where islets and small islands are close enough together to form a “pool” (fig. 16). These sites are, however, potential lagoons in the future with continuing land uplift. Artificial pools (harbours, piers) are not included in the Swedish results, as the exploitation index is used to only include unexploited areas, on the contrary the analysis may miss areas where the exploitation has no or very little effect on the habitat.

Large shallow inlets and bays: The results are shallow, sheltered water bodies. No information weather the object does have high biodiversity or a well developed zonation exists. The analysis excludes areas smaller than 20 ha and may miss areas that have a freshwater inflow, although of minor effect on the habitat. The analysis also creates rather irregular shaped edges toward the sea, and often includes areas where there are large numbers of islands in close proximity.

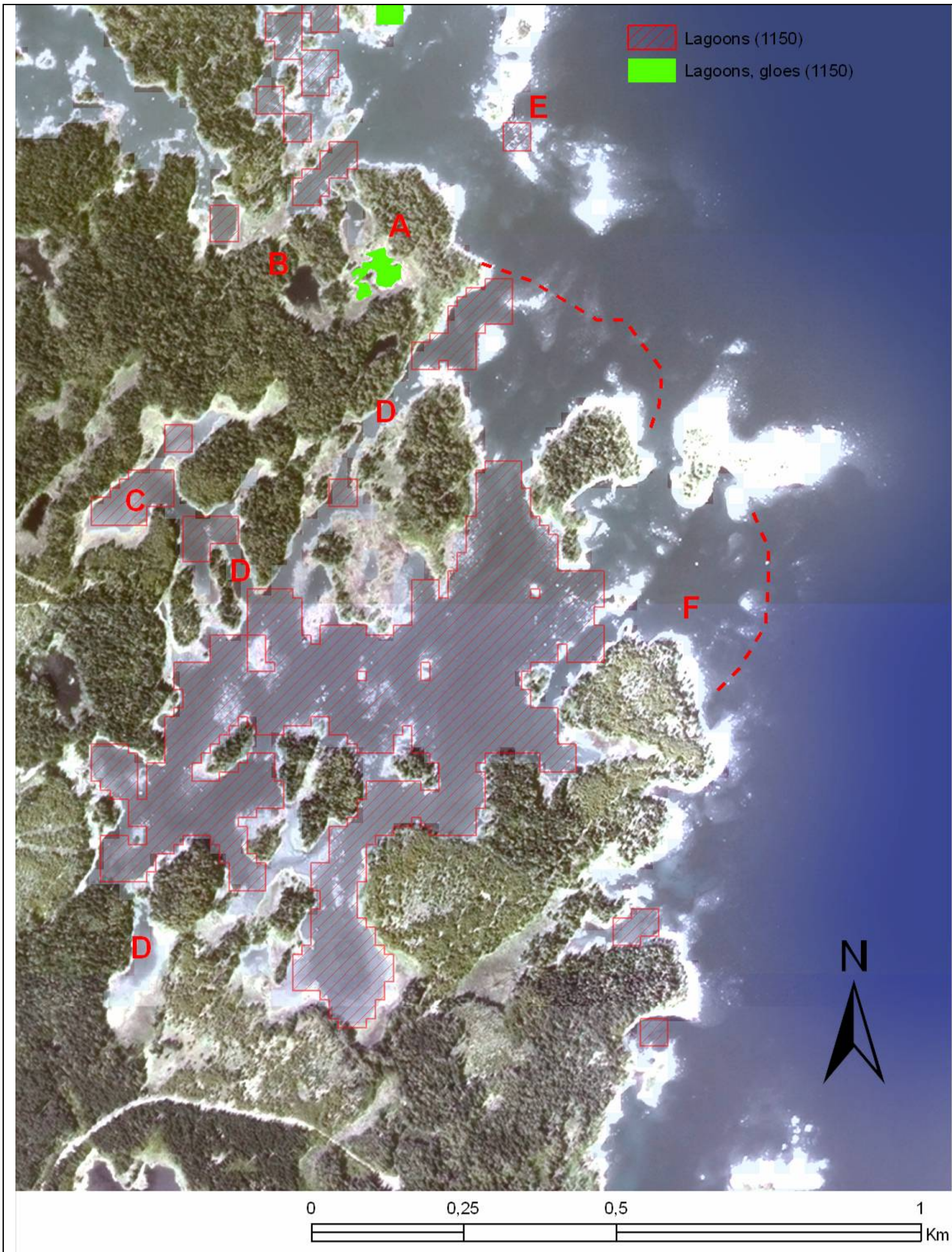


Fig. 16 *Results from the Coastal lagoon analysis. The analysis of gloes identifies lakes (according to maps) that are within 15 meter to the shore line (A). In the flat topography of Uppland, lakes even further from the shore (B) may be recently separated from the sea. The analysis misses these objects. The analysis of partially separated lagoons identifies shallow areas that have an opening towards the sea more narrow than 30 meters (C). The analysis miss areas more narrow than 30 meters (D) and false lagoons can be created when small skerries are close to each other, forming a pool (E). The analysis identifies important locations but probably underestimate the area of lagoons. In the definition of lagoons a threshold towards the sea can outline the habitat. In lack of adequate depth data these areas can not be analysed. In the figure, a larger area of the shallow bay could most probably be included (F) as several thresholds are present (red lines).*

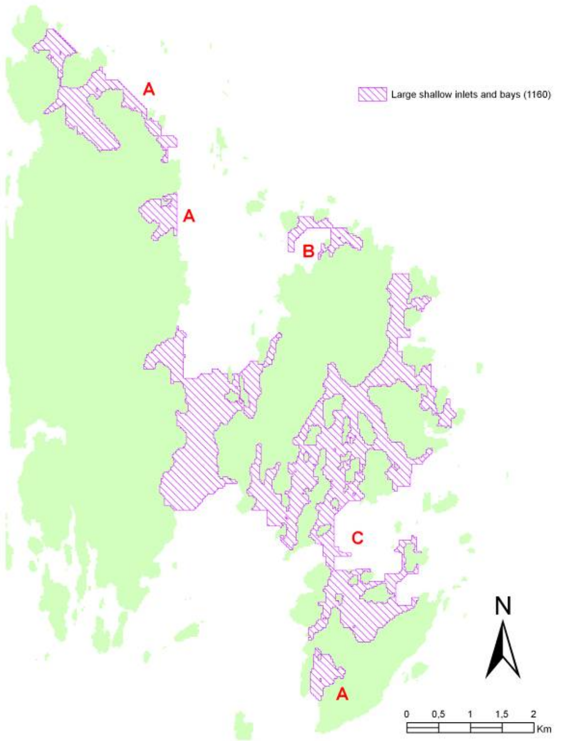


Fig. x *Mapped 1160 in the County of Stockholm. The outline is usually good (A). The analysis is based on each pixels proximity to land in a raster, this can lead to that only areas closest to land are included (B) or to outcrops of the bay into open sea (C).*

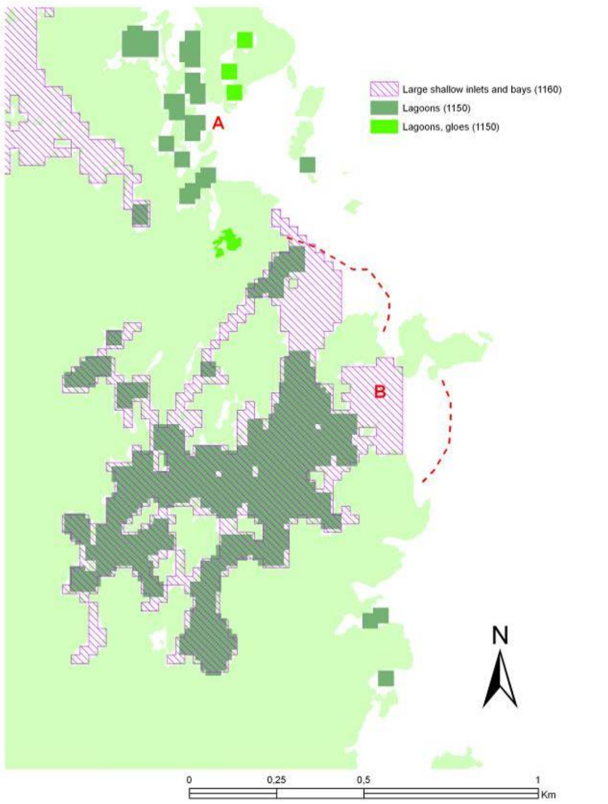


Fig. x *Mapped 1160 on top of 1150. The results overlap to some extent when the objects are between 20-30 ha and more shallow than 6 m. The majority of the mapped lagoons are small separated objects (A) while the objects mapped as Large shallow bays are large and captures the whole sheltered area of the bays (B). Both results are shallow sheltered areas with a high potential of having high nature value.*

Reefs: The method produced potential locations for reefs. However, the presence of a hard substrate is postulated from the presence of rocks marked on nautical charts. This attribute is the generalised to an entire elevated area. Whilst the rock often correspond to submerged bedrock formations, they may also be boulders in a till formation. The accuracy of the selection process would improve with better depth data. With better depth data in deeper areas, more deep reefs could be found. The availability of geological data would also make the selection of reefs easier, by pointing out the rocky outcrops the reefs are made of. The extent of the selected areas may differ some, depending on how certain selection criteria is set, such as the radius of the neighbourhood, when selecting peak areas or extrapolating hard substrate from rocks.

A comparison of the mapped reefs to the EUNIS-classification in the County of Stockholm (Mattisson 2005) gives that 81 % of the objects mapped as reefs are hard bottom substrates according to the detailed geological information. The area cover of the EUNIS-classes within the mapped reefs shows that besides hard bottoms, about 15 % are glacial clays and 4 % are mixed bottoms.

Baltic esker islands: The results have major uncertainties, and better geological information is needed to perform GIS analysis. Mapping by interpretation of aerial photos may be a better approach. In Sweden only islands with glacio-fluvial material are selected, although eskers forming a spit could be included in the habitat, having the same type of environment. The map of soil types does not cover the whole pilot area. In Finland the entire island was selected even if only part of it was esker. No data was available on macrophyte vegetation. An approximation of the photic depth is used as a proxy for the presence of sublittoral vegetation. The extent of the formation below the surface is represented by a 200 m buffer, which may either underestimate or overestimate the actual formation. Often an esker will form a chain of several islands. The presence of glacio-fluvial material in the buffer zone is not included in the analysis as this data was considered too uncertain, and was not available for the whole area. Better data on depth and glacio-fluvial deposits below water would improve the analysis, by allowing the determination of the entire esker formation both above and below the surface.

Boreal Baltic islets and small islands: The analysis selects islands in exposed location with little or no trees. Esker islands are excluded from the results. The selected islands are well correlated to the description of the habitat, the uncertainty lies within the outlining of the surrounding marine environment.

Application of habitat maps in management

In general the modelled layers satisfy the needs of large scale planning of the coastal sea. All maps show the potential occurrence of the habitats and can be used to derive habitat complexity maps, estimate the proportion of protected versus unprotected areas of the habitats and can be used as a first selection of areas of interest for more detailed surveys. The Natura 2000 habitats do not cover deeper habitats or shallow hard bottom habitats that may have high nature value.

6.3.5 Conclusion and perspectives

Conclusions from the mapping:

- The methods are direct and very useful for habitat modelling of Natura 2000-habitats.
- The results are very sensitive to the quality of existing data.
- High resolution map of the sediment characteristics and depth is needed to improve the results.
- Deeper reefs (shallowest part deeper than 6 m) and the deepest part of the reefs identified (> 10 m of depth) and Sublittoral sandbanks outside the geologically surveyed areas are missing
- A separation between large shallow inlets and bays and Long narrow inlets are not done
- Overlap between habitats in the results can be used to find complex areas with a potential of having high nature values.
- The NATURA 2000 habitats covers most of the shallow soft-bottom environments of high nature value in the pilot area, although do not cover hard substrates and deeper habitats.
- The habitats mapped give no information of the vegetation cover in the areas.

To get a more complete overview for management these results could be complemented with models showing:

- Vegetation cover in shallow soft bottoms
- Vegetation cover around islands
- Shallow hard bottoms with zoned vegetation
- Deeper hard bottoms with zoned vegetation
- Deeper soft bottoms separated into sandy/muddy areas without oxygen depletion
- Areas with high values for fish

Some of the part results from the analyses could be used to fulfil the overview for managers:

- The part result from the reef analysis when sloping sea beds are excluded due to the connection to land are most probably hard substrates with zoned vegetation
- The part result of large shallow inlets and bays influenced by fresh water inflow can still have high nature values depending on water exchange.
- The part result of lagoons with exploitation can have high nature values or be possible to restore.

6.4 Application of GIS Analyses and Habitat Modelling for Classified Habitat Maps (Balance Pilot Area 3)

Authors: Anna Nöjd.

6.4.1 Pilot area 3

Pilot area 3 is located in the vast archipelago region that stretches from the counties of Södermanland, Stockholm, and Uppsala in Sweden, over via Åland and the Finnish Archipelago Sea (fig. 1). It is a topologically and geologically very heterogeneous area that consists of numerous islands and smaller islets, and habitat patchiness is thus normally high even on smaller spatial scales. The bottom substrate normally consists of a complex mix of soft and hard substrates. Modelling this patchy distribution of habitats is a challenge, and to succeed in making accurate habitat maps detailed maps of the physical environment is needed.

There are strong salinity gradients in the area, both in a north-south direction as well as from the inner parts of the archipelago to the outer parts. The salinity in the outer archipelago varies from around 5 psu in the northern parts to around 7 in the south. In the innermost bays and flats the salinity may be as low as 3-4 psu, and occasionally even lower, depending on inflow of freshwater.

Water temperature and ice cover also often varies substantially between the inner and outer parts of the archipelago. Maximum summer water temperature range is normally 15-20 ºC in open water and around 25 ºC in sheltered shallow areas. The most dramatic gradients in temperature regime normally occur in spring - early summer when temperature differences between sheltered shallow areas and the open sea may exceed 10 ºC. Shallow and sheltered areas on the other hand cool off more rapidly in autumn than the open sea. Temperature affects growth and reproduction of macrophytes and sedentary and sessile animals. Some mobile invertebrates and most fish are very sensitive to temperature changes.

Temperature variations may have a large influence on fish migrations within an area. Many coastal fish species, such as pike, perch, and pike-perch need warm water for reproduction, and mainly utilise shallow, sheltered areas in the archipelago as nursery areas. Also fish species like turbot, flounder, whitefish, and herring utilize the productive coastal areas for spawning. The adults of these species prefer cold water, and therefore migrate to the outer parts of the archipelago and/or to deeper water layers when the water in the inner parts gets too warm in summer.

Shallow coastal areas are normally characterised by soft sediment bottoms in sheltered areas with a macrophyte community dominated by habitat structuring species such as pondweeds (*Potamogeton* spp.), stoneworts (*Chara* spp.), milfoils (*Myriophyllum* spp.) and emergent reed, (*Phragmites australis* L.). When wave exposure increases, substrates go from soft to a mix of soft and hard. In exposed parts of the archipelago hard substrates dominate at shallow sites. These areas are often dominated by bladderwrack (*Fucus vesiculosus* L.), and in the northern parts of the area by the closely related endemic *Fucus radicans* **, as the main habitat forming species. The maximum depth for vegetation differs depending on light attenuation, ranging from around 2 to 10 m, with the shallowest maximum depth in turbid inner bays. There are large variations in nutrient runoff from land, with the highest loadings in the densely populated parts of the archipelago, e.g. the Stockholm and Turku areas. The variability in nutrient loadings in combination with large differences in water turnover time result in strong turbidity gradients on both large (km) and small (m) scales.

Introduction

Many international, European, and national policies and agreements require the preservation of the diversity of habitats, and assessment of the habitats' conservation status as well as instigating the ecosystem approach and sustainable use of marine areas. The marine environment is incredibly complex, with fuzzy borders between communities making it difficult to pinpoint habitats. Habitat classification schemes have been used to convert the continuum of nature into discreet units suitable for use in management and planning.

Geographical information systems (GIS) offer a number of ways to analyse and split spatial data into classes that are meaningful from the viewpoint of a habitat classification system. GIS enables us to produce full cover maps of our variables of interest, in this case, habitats. The approach presented here would be applicable to any hierarchical classification system where data is available on the factors that determine the classes. At present the Environment Agency's European Nature Information System (EUNIS) classification of marine habitats is the only available classification system. The Baltic Sea environment has been included in the EUNIS classification by integrating the previously existing HELCOM red book classification system (**ref?).

Here, the applicability of GIS methods and hierarchical classification in the Baltic Sea is discussed via a case study near Ormskär in the Archipelago Sea (PA 3), in which an attempt was made to map benthic marine habitats according to the EUNIS classification.

The classes used here for BALANCE purposes follow, as closely as possible, the EUNIS classes on levels two, three, and four.

Archipelago sea habitat maps

The EUNIS system has a hierarchical structure that at level 2 uses purely abiotic characteristics to describe habitats, whilst an increasing amount of biology is included at the higher levels. Level 3 of the classification already includes certain biological factors, namely cover of macrophytes and biogenic reefs. Table 1 shows the EUNIS habitat types that exist in the study area for levels 2, 3 and 4.

Table 1: EUNIS Habitats in the Study Area

Level 2		Level 3		Level 4	
A3	Infralittoral rock and other hard substrata	A3.4	Baltic exposed infralittoral rock		
		A3.5	Baltic moderately exposed infralittoral rock		
		A3.6	Baltic sheltered infralittoral rock		
A4	Circalittoral rock and other hard substrata	A4.4	Baltic exposed circalittoral rock		
		A4.5	Baltic moderately exposed circalittoral rock		
		A4.6	Baltic sheltered circalittoral rock		
A5	Sublittoral sediment	A5.1	Sublittoral coarse sediment	A5.11	Infralittoral coarse sediment in reduced salinity
				A5.13	Circalittoral coarse sediment
				A5.14	Deep circalittoral coarse sediment
		A5.2	Sublittoral sand	A5.21	Sublittoral sand in low or reduced salinity
				A5.27	Deep circalittoral sand
		A5.3	Sublittoral mud	A5.31	Sublittoral mud in low or reduced salinity
				A5.37	Deep circalittoral mud
		A5.4	Sublittoral mixed sediments	A5.41	Sublittoral mixed sediment in low or reduced salinity
				A5.45	Deep mided sediments
		A5.5	Sublittoral macrophyte dominated sediments	A5.52	Kelp and seaweed communities on sublittoral sediment
				A5.54	Angiosperm communitites in reduced salinity
		A5.6	Sublittoral biogenic reefs	A5.62	Sublittoral mussel beds on sediment
		A5.7	Features of sublittoral sediments	A5.72	Organically enriched or anoxic sublittoral habitats

Can be achieved using existing abiotic GIS layers	Requires habitat models made from biological data overlaid with the habitats from abiotic GIS data	Requires a spatial model of organic enrichment and anoxia overlaid with the habitats from abiotic GIS data

The GIS analysis of EUNIS habitats used four sets of data layers:

- Substrate data classified according to the BALANCE substrate classification.

- A photic/aphotic layer dataset derived from a model based on the level to which an area is enclosed (see chapter **).
- Wave exposure index data classified into 3 categories (sheltered, moderately exposed, and exposed), first using the cut-off values derived from an analysis distribution of lichens and algae on shores (Isaeus **) and then combining the resulting seven classes into 3.
- Raster layers (5 m grid cell size) with probability of presence of *Mytilus trossulus*, algae, and angiosperms (see chapter **) in the Ormskär area.

Substrate and photic depth were combined in a GIS overlay analysis to produce maps of EUNIS level 2. Level 3 included a third factor, wave exposure (fig. 1). Level 2 and 3 analyses were run for the whole Archipelago Sea, omitting the biological factors on level 3 (table 1).

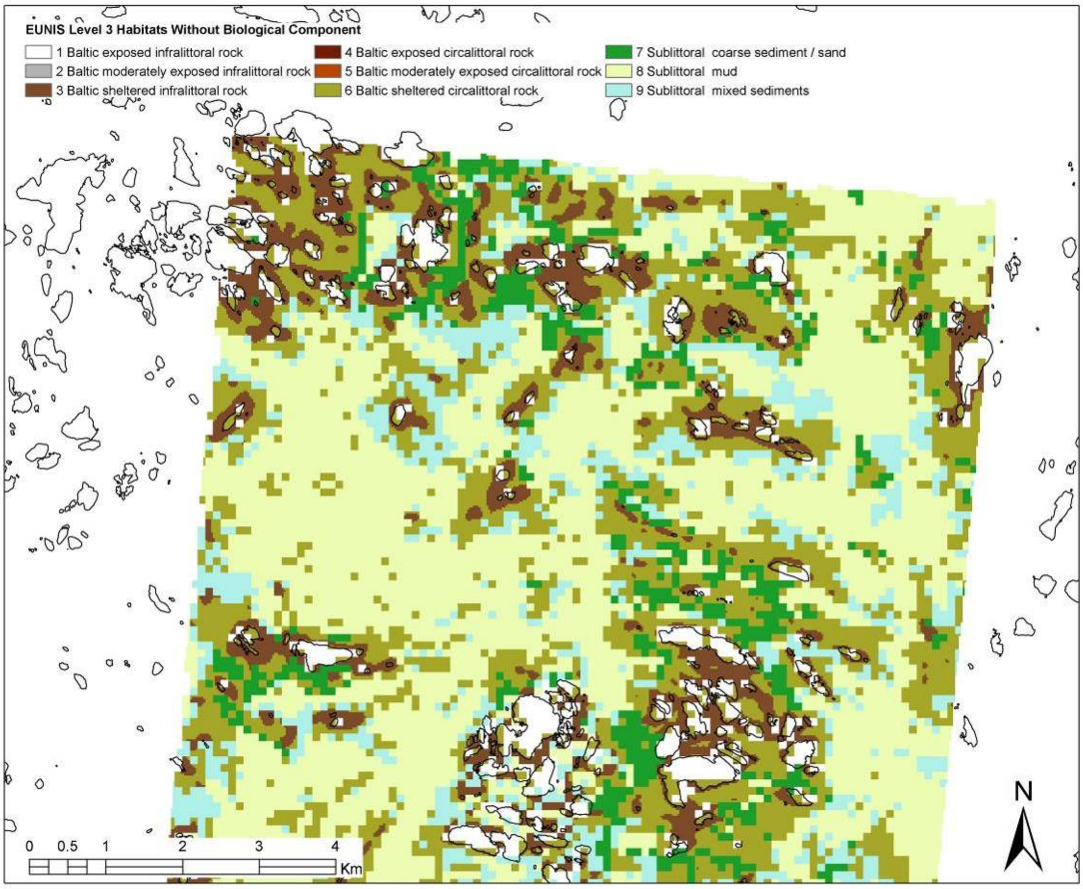


Fig. 1 *An example showing the Ormskär area with the EUNIS level 3 habitat map for the Archipelago Seacreated using only abiotic factors as source data.*

To be able to include the biological factors into habitats at level 4, results from previous modelling exercises of mussels, algae, and angiosperms were used to approximate the distribution of these tree groups of biota. Since no data was available on the extent of anoxia, this feature could not be added to the level 4 map. The Ormskär area, with habitat models for the required elements already created in an earlier exercise (see chapter

**) was used as an example area in an attempt to develop and demonstrate the approach (fig. 2).

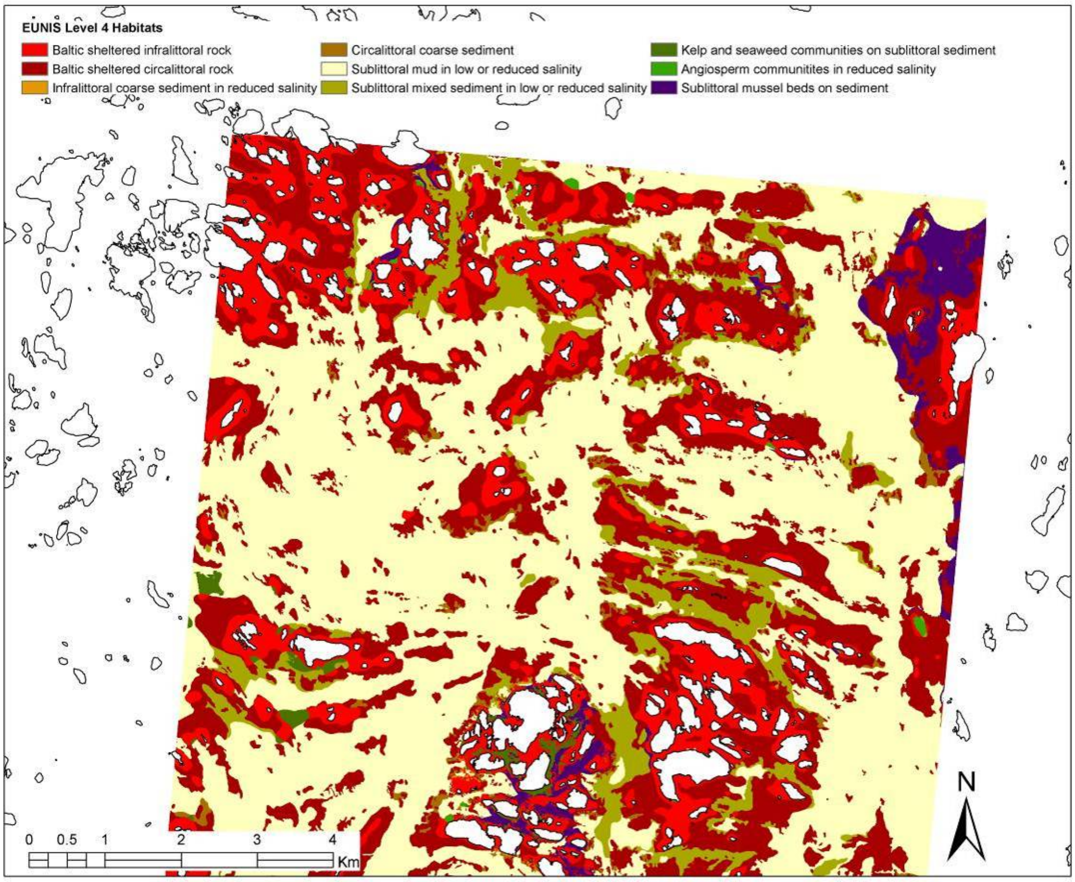


***Fig. 2** EUNIS level 4 habitat map for the Ormskär area in the Archipelago Sea where spatial predictions of the distribution of mussels, algae, and angiosperms have been incorporated in the GIS analysis to include the biotic element of the habitats.*

The applicability of the EUNIS classification in the study area

Combining the methods of GIS overlay and predictive modelling of species distribution it was possible to produce maps of the existing EUNIS habitat classes at level 2, 3, 4 in the case study area. The Balance substrate classification corresponds fairly well to the substrate classes used in the EUNIS classification, and an indication of how to classify the wave exposure index to fit EUNIS classes was available based on the work of Isaeus (2004). In the context of the Archipelago Sea the terms "infralittoral" and "circalittoral" were considered synonymous with photic and aphotic, respectively. The depth of the photic layer has a strong gradient from the inner, more enclosed parts of the archipelago to the outer, more open parts. In these conditions a standard cut-off depth would induce error into the boundary drawing of the habitats. Table 1 describes the data and methods needed to make a map of the habitat classes at each level of the current EUNIS marine habitat classification that are found in the study area.

Analysis of the biological relevance of the level 3 habitat layer using drop-video data from 0 to 25 m depth showed that, in shallow waters, the classification coincided reasonably well with the structure of biological communities. The same analysis on infauna in deeper waters however showed hardly any differences in the communities found within different EUNIS classes in the map. The poor result of the infauna analysis is likely due to the coarse classification of sediment types in the source (original) substrate map, which also showed a relatively poor ability to structure communities. The video data was largely from hard bottoms, so the main structuring factor was photic depth, which was clear from the indicator species analysis. (Reference here to our report/paper). It can be concluded that, in the area tested, the interpretation taken to the EUNIS classes in the GIS analysis bore a resemblance to the real life communities.

However, several problems presented themselves in the process of classifying habitats according to EUNIS. Presently, the Baltic Sea area is poorly represented in the EUNIS system. The Baltic classes do not follow the hierarchical structure of the classification very well and are somewhat inconsistent to what is included on each level (see table 1). Improvements to the classification are sorely needed. The Baltic has several gradients that do not play a significant role in the truly marine environment from where EUNIS originates. The most obvious differences are the lack of tides, the salinity gradient, benthic substrate complexity, and the enclosed nature of the sea.

The lack of tides means there is a narrow or no intertidal zone (<0.5 m). However, some of the species found in the intertidal zone on marine shores, form a similar zonation subtidally in the Baltic (fig. 3). This is currently not laid out in the existing EUNIS hierarchy. In the Archipelago Sea salinity changes from almost freshwater in the innermost archipelago and near river mouths to approximately 5-7 ppt where it joins the Baltic Proper. On the scale of the whole Baltic Sea are the gradient is much larger, from 0 to 34 ppt. The enclosedness limits fetch and consequently wave exposure. Although wave exposure in the Baltic may be small compared to Atlantic shores, the variation within the Baltic plays an important role in structuring communities.

Fig. 3a. *Mytilus trossulus and Balanus improvisatus on hard bottom. Photo: Metsähallitus, 2004.*

Fig. 3b. *Fucus vesiculosus dominated algal community. Photo: Metsähallitus, 2004.*

In the future there is a need to recalibrate wave exposure specifically for the Baltic Sea area. There is also a need to create true classes based on the flora and fauna communities and the special abiotic factors at play in the Baltic Sea area, to be incorporated into the hierarchical structure of EUNIS.

Application of classified habitat maps in management

Although the splitting of nature into discrete categories introduce errors in the representation, it is often necessary for assessment, reporting, and management purposes. The maps produced using such methodology, will provide a basic view of the types of habitats that are found in an area. The ecological considerations related to these habitats can be inferred from existing knowledge. Habitat maps based on environmental data layers are only as accurate as the source data. The approach is very dependent on the quality of existing data. Additionally, in nature some habitats are fairly stable in time lasting for more than 100, 1000 or 10.000 of years, others are highly dynamic. The habitats depicted in the maps may have seasonal or multi-annual cycles (e.g. annual algae, cyclic bottom fauna communities on soft bottoms). However, if these problems are acknowledged and included in decision making, the maps can be a good addition to the sustainable management of marine areas and marine conservation.

A P P E N D I X B
Case studies: Predictive Modelling of species habitats

6.5 *PA1. Spatial predictions of* Laminaria hyperborea *in Norwegian Skagerrak*

Authors: Norderhaug, K. M., K. M., Isæus, M., Bekkby, T., Moy, F., Pedersen. A.

6.5.1 *Introduction*

Background. Results from an ongoing investigation along the Norwegian Skagerrak coast shows that major changes occur in the phytobenthic community along the Norwegian Skagerrak coast. The kelp *Saccharina latissima* (former *Laminaria saccharina*) has disappeared from many sites and been exchanged by a filamentous turf, and has then recolonised again at some sites (Moy et al. 2003, 2007) Changes in the distribution of the kelp *Laminaria hyperborea* has also occurred, but the changes appear not to be as drastic as for *S. latissima*. *L. hyperborea* is a habitat forming species (fig. 1) with a highly diverse community associated with its stands, and the habitat is therefore pointed out as a prioritized nature type by Norwegian authorities (Anon. 2001).

Dive transect data from the national coastal monitoring (Kystovervåkningen or KYO) provides time series data with information about the occurrence of sessile organisms, including *L. hyperborean*, on monitoring stations collected in Skagerrak during the period 1990-2006. These data may be used to model changes in the distribution of *L. hyperborea* if the stations are representative with respect to the distribution of *L. hyperborea* within intervals of the factors that are used in the model.

Aims. The aim of the study was to investigate to what extent the distribution of *L. hyperborea* in Skagerrak has changed over the last 10 years. Available data included yearly registrations (0-30 m depth, 6 stations) from KYO, and registrations from the National program for mapping of prioritised nature types (fig. 2). (KYO stations from which data was not available for all years were excluded, and are not shown in fig. 2). First, we needed to test if the data set from the National monitoring program (KYO) was large enough to be used to predict the spatial distribution of *L. hyperborea*. If so, we use predictive modelling for estimating the spatial changes of *L. hyperborea* over time. If the KYO-data was not sufficient, we use all data for spatial modelling to make the best possible prediction of *L. hyperborea* distribution along the Norwegian Skagerrak coast.

Fig. 1. Laminaria hyperborea *kelp forest at Finnøy on the west coast of Norway. Photo Martin Isæus.*

6.5.2 Material and Methods

The test if the data set from the National monitoring program (KYO) was large enough to be used to predict the spatial distribution of *L. hyperborea* was done by comparing the results of three spatial models; one based on all data from the National program for mapping of prioritised nature types plus KYO data from 2004 (Full model), one based only on KYO data from 2004 (KYO 2004 model), and one based only on KYO data from 1995 (KYO 1995 model). It was assumed that the Full model would be able to predict *L. hyperborea* distribution better than the KYO models. If predictions from the Full model and the KYO model 2004 were approximately similar, we would conclude that KYO data was sufficient to construct models for predicting the distribution of *L. hyperborea* in Skagerrak. KYO models from different years could then be used to analyse changes in the distribution of *L. hyperborea* between years. To use the years 2004-2006 together in the Full model, it was assumed that the change during these three years was not significant.

Field sampling and data sources. The National Mapping Program included approximately 200 drop camera registrations in three areas in the Skagerrak Sea from 2005 and 2006. The KYO datasets included yearly registrations from 1990-2005 at 6 stations that were monitored during the whole period. Registrations of all visible sessile organisms were made by divers along transects from 30 to 0 m depth.

Predictor layers. Predictors in the models are:

I. Wave exposure (SWM in Isæus 2004, 25 m resolution)

II. Depth (digital elevation model, 25 m resolution)

III. Curvature (analysed from the depth model, 500 m resolution)

IV. Slope (analysed from the depth model, 25 m resolution)

V. Light exposure (light exposure in respect to the optimal angle, calculated from slope and aspect)

Response variables. Response variable was presence or absence of *L. hyperborea*.

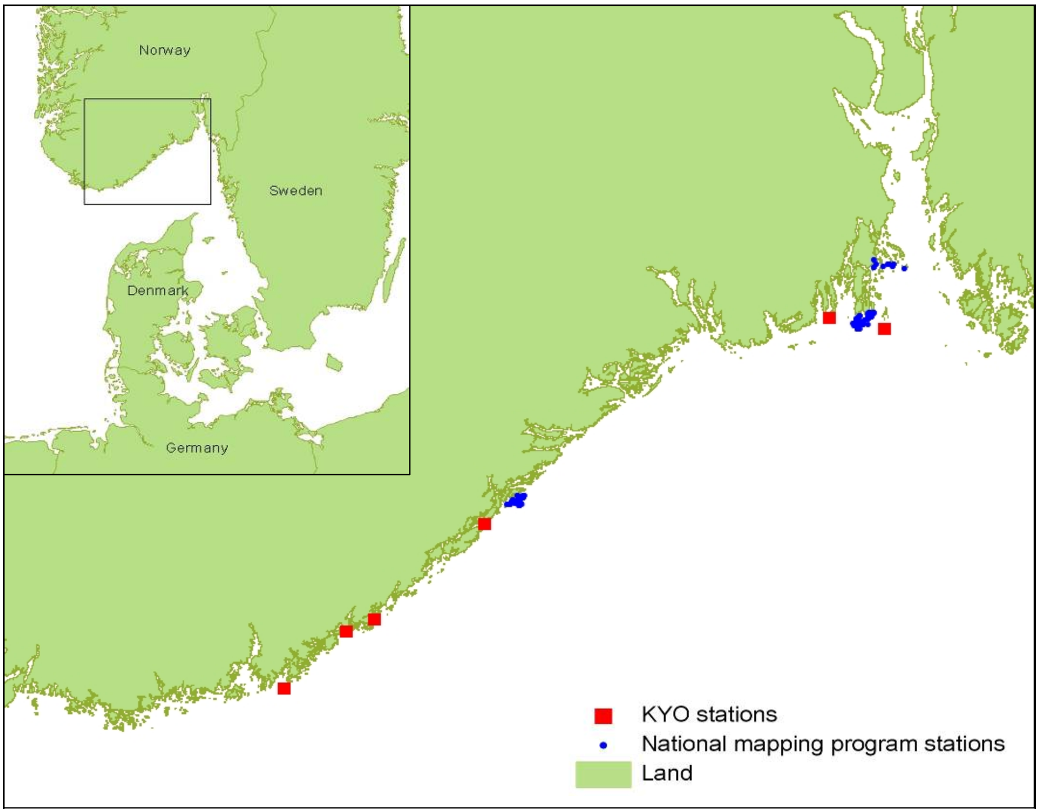


Fig. 2. *Field stations of the national monitoring program (KYO) and national mapping program 2005-2006.*

Model selection (methods, algorithms, software, and routines). Generalized Additive Model (GAM) in the GRASP extension to the S-PLUS software package was used for statistical analyses of the data. The Akaike Information Criteria (AIC) was used for model selection. GRASP (Lehmann 2002) has proved to be a good tool for predictive modelling in both aquatic (Francis 2005, Garza-Pérez 2004, Schmeider 2004) and terrestrial environments (Zaniewski 2002).

Spatial predictions based on the GAM models were made in ArcView. Predictions result in grid-based maps showing the probability of presence of *L. hyperborea* in each grid cell. Predictions were classified in four probability classes, 0-0.25, 0.25-0.5, 0.5-

0.75, and 0.75-1. All grids had a resolution of 25 m. The prediction grids were then compared using the Spatial Analyst extension in ArcView.

To build a model of species distribution that accurately describes the variation of the target species, it is necessary that the collected field data covers the whole gradient of the environmental variables used as predictors. For example, if the model should be valid from 5-50 meters depth, this whole gradient should be present in the field data, even though the target species is only present between 10-15 meters. If part of the gradient is missing from the field data, the model and hence the predictions in this span will likely be inaccurate. In this case, this meant that predictions were restricted to areas where environmental predictors (primarily wave exposure) were inside the span covered in the field data. For the Full model, predictions were made in the exposure interval 2.900-627.000 SWM, while the KYO data predictions were made in the exposure interval 124.000-554.000 SWM.

6.5.3 Results

The Full model. The predicted distribution of *Laminaria hyperborea* (LAMHY) along environmental gradients in the Full model is shown in fig. 3. All available data from the National program for mapping (registrations from 2005-2006) and KYO (National monitoring program, registrations from 2004) were used. As can be seen in fig. 3, field data in this case covered almost the whole gradient of the different environmental predictors.

According to AIC selection, the best model (AIC=181.3) explaining presence of *L. hyperborea* (LAMHY) includes depth, exposure (SWM), light exposure (LYSEKSP) and curvature. Cross validation showed a cvROC (5-fold) of 0.95. The partial response curves for each predictor in the selected model are shown in fig. 4. Spatial prediction of *L. hyperborea* (LAMHY, probability of presence) from the Full model is shown in fig. 5. *L. hyperborea* is found at exposed sites in the sublittoral down to approx. 25 m depth. In a segment of the map (a close-up), the predicted distribution of *L. hyperborea* on the outside (the exposed side) of skerries can be seen (fig. 6).

KYO 2004 model. The distribution of *L. hyperborea* (LAMHY) along environmental gradients in the model based on KYO 2004 data only, is shown in **Fig. 7. As can be seen in this figure, field data in this case did not cover the whole gradients of environmental predictors, perhaps most obvious for SWM and slope. This is due to the fact that the model is based on data from only six sites, and the variation in horizontally varying parameters, such as wave exposure at surface level, is low. On the other hand, the description of the variation of kelp along the depth gradient is well described (**Fig. 7, "DEPTH") since all six stations have registrations at each meter of depth in the phytobenthic zone. According to AIC selection, the best model (AIC: 72.0) to explain presence of *L. hyperborea* (LAMHY) includes depth, wave exposure (SWM), and curvature. Light exposure (LYSEKSP) and slope was excluded as a predictor from the model. Cross validation showed a cvROC (5-fold) of 0.93 for the selected model. In **Fig. 8 partial response curves for each predictor in the selected model are shown. Spatial predictions from the KYO 2004 model of probability of presence of *L. hyperborea* in a larger part of the Skagerrak area is shown in **Fig. 9. In a smaller segment of the map (the same area showed in fig. 6), the distribution of *L. hyperborea* on the outside (the exposed side) of skerries can be seen (**Fig. 10).

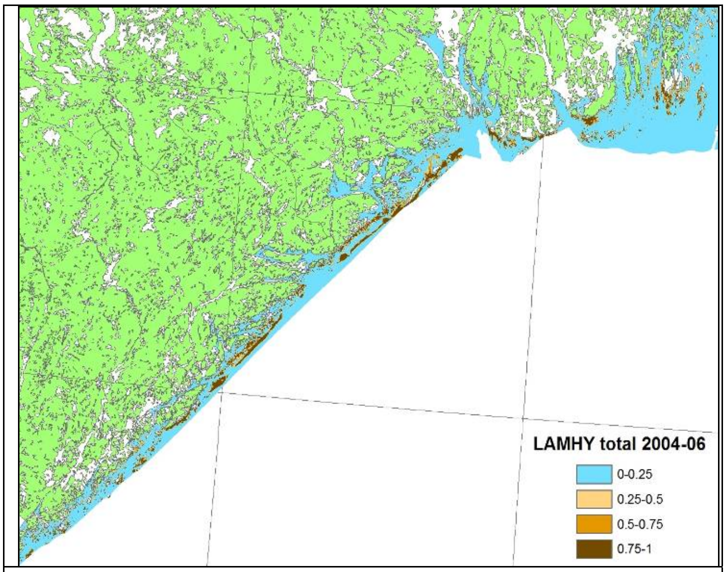


Fig. 5. *Probability of presence of* Laminaria hyperborea *in four classes, as predicted by the full model. White areas have SWM outside the span 2900-627000, and are outside the area in which the model can reliably predict* L. hyperborea *distribution. Green areas are land area.*

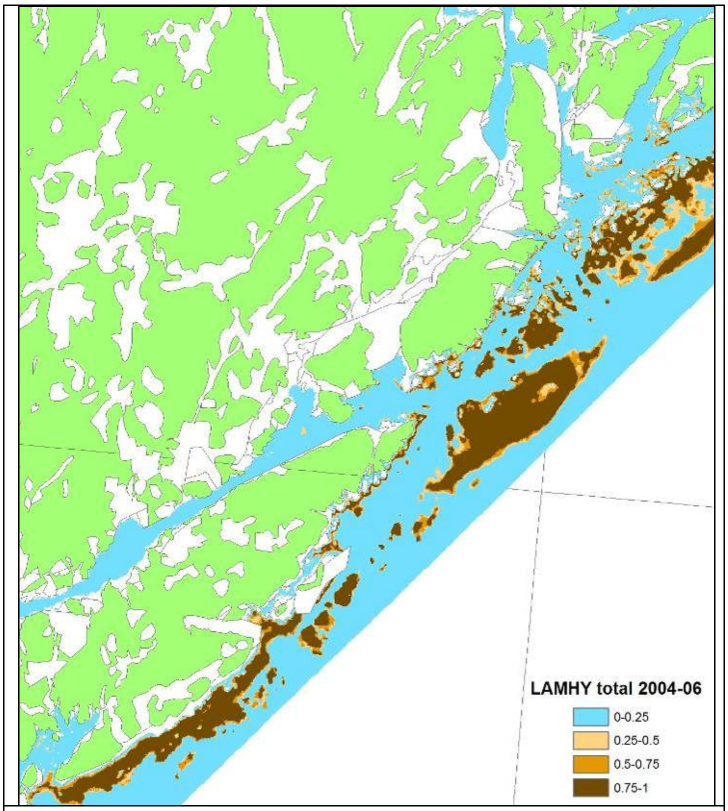


Fig. 6. *Probability of presence of* Laminaria hyperborea, *as predicted by the full model. Map segment from a smaller part of the Skagerrak. White areas have SWM outside the span 2900-627000, and are outside the area in which the model can reliably predict* L. hyperborea *distribution. Green areas are land area.*

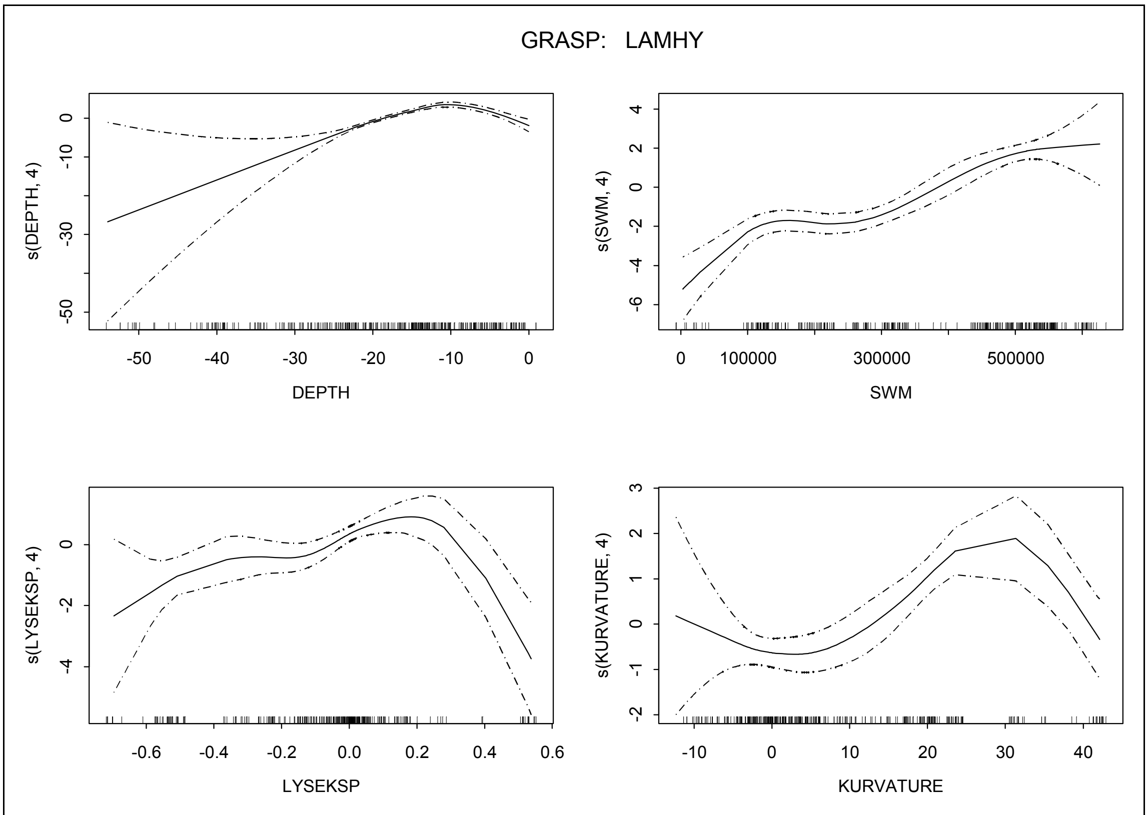


Fig. 4. Partial response curves (±2 x S.E.) of LAMHY presence for each predictor in the selected model.

KYO 1995 model. The distribution of *L. hyperborea* (LAMHY) along environmental gradients in the model based on KYO data from 1995 is shown in **Fig. 11. The figure indicates gaps in the gradients for several of the environmental predictors. According to AIC selection, the best model (AIC=66.0) to explain presence of *L. hyperborea* (LAMHY) includes depth, exposure (SWM) and Curvature. Similarly to the KYO 2004 model, Light exposure (LYSEKSP) and Slope was excluded as a predictor from the KYO 1995 model. Cross validation showed a cvROC (5-fold) of 0.94 for the selected model. In **Fig. 12, partial response curves for each predictor in the selected model is shown. Spatial predictions of *Laminaria hyperborea* from the KYO 1995 model is shown in **Fig. 13. In a smaller segment of the map (the same area showed in fig. 6 **and 10), the distribution of *L. hyperborea* according to the KYO 1995 model can be seen (**Fig. 14).

Comparison of models. There were generally larger differences in the prediction between the full model and the KYO 2004 model, than between the two KYO models. Both KYO models underestimated the distribution of kelp compared to the full model (tab. 1). Differences in predictions between the full model and the model including KYO data from 2004 are shown in fig. 15. The main difference between the models was in areas where the KYO 2004 model underestimated the distribution of kelp (blue areas), compared to the Full model. There were generally small differences between the KYO 1995 model and the KYO 2004 model (**Fig. 16). The models predicted larger distribution of kelp in 2004 than 1995, but the differences were small (and much smaller than differences between the full model and the KYO 2004 model, see the Discussion section).

Table 1. Distribution of data (counts and percentage of cells in the grid) between different probabilities according to the full model and the models including data from KYO in 1995 and 2004.		
Probability of presence of *Laminaria hyperborea*.	**Counts**	**Percentage**
Full model		
0-0.25	3 915 967	0.89
0.25-0.5	136 033	0.03
0.5-0.75	131 757	0.03
0.75-1	204 546	0.05
KYO 2004 model		
0-0.25	1 503 632	0.97
0.25-0.5	7 791	0.005
0.5-0.75	6 097	0.004
0.75-1	25 828	0.02
KYO 1995 model		
0-0.25	1 518 155	0.98
0.25-0.5	3 261	0.002
0.5-0.75	3 085	0.002
0.75-1	18 847	0.01

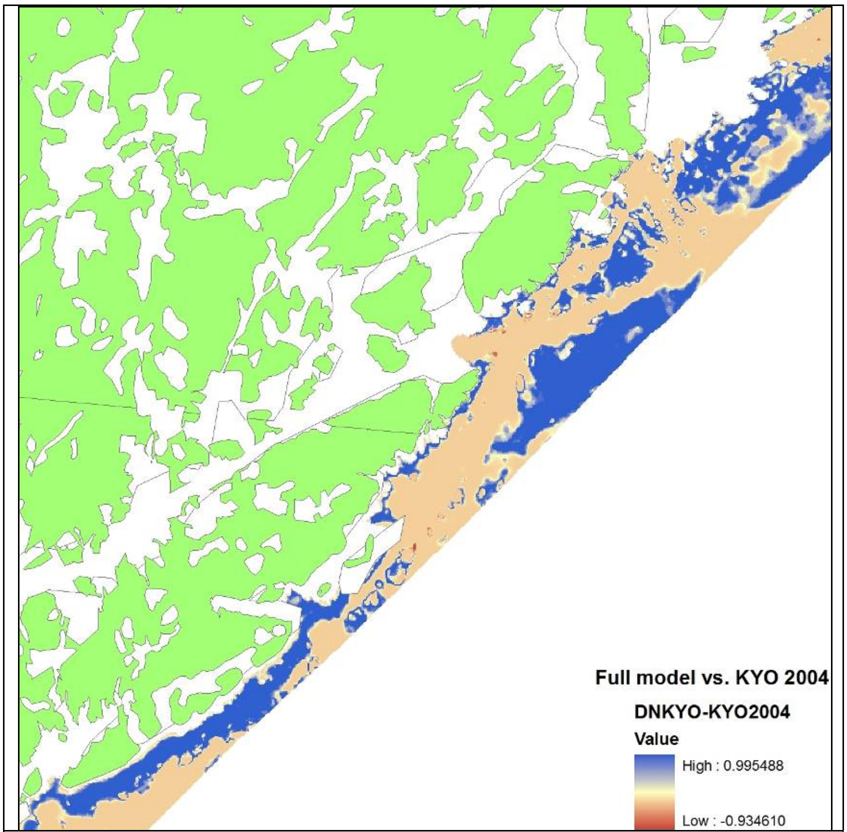


Fig. 15. *Differences in predictions between the full model and the KYO 2004 model (expressed as Prob(Full model) – Prob(KYO 2004 model)). In blue areas the full model predicts a higher probability of finding* Laminaria hyperborea *than the KYO 2004 model. In red areas the KYO 2004 model predicts higher probability. In brown areas the predictions of the two models are equal. White areas are outside the area in which one or both of the models can reliably predict* L. hyperborea*. Green areas are land area.*

6.5.4 Discussion

Quality of data and methods. A comparison between all models showed that there where larger differences between the full model (incl. all available data from 2004-2006) and the KYO 2004 model, than between the 2004 and 1995 KYO models. The KYO models, including KYO data only, underestimated the distribution of *L. hyperborea* when compared to the full model. Because the full model is based on more data covering a larger part of the environmental gradients, it can be assumed that this model is more reliable in predicting the distribution of *L. hyperborea* in the investigated area. The results show that the KYO data alone cannot be used for predicting variation in the spatial distribution of *L. hyperborea* in the Skagerrak between years.

Significance of the predictor curvature in the model may reflect the importance of substrate, because curvature indicates where rocky bottom vs. level bottom (potentially of all bottom types) is found. There is a general need for substrate information which is expected to increase the precision of benthic models. The lack of importance of slope as a factor in the model (excluded by AIC selection in all the GRASP models) probably reflects that there were few steep stations in the datasets (fig. 3). It is known that kelp do not attach to very steep surfaces. The factor slope is also strongly dependent on scale and it may be that the resolution of 25 m grid cell size does not sufficiently describe slope variation for this modelling purpose.

The prediction is limited to the range of wave exposure in which kelp data are available. Within this range there is a positive effect of exposure on kelp distribution, but this effect is expected to drop in areas with very high exposure (outside the range of this study).

The time series data from KYO represent a very important tool for monitoring community changes in the Skagerrak. Such time series data will be crucial in the future for analysing possible biological effects of large scale changes, e.g. climatic changes. The monitoring program is however not designed to analyse spatial changes in the distribution of species. To do this, specific monitoring programs are needed.

Quality and validation of models. Validation of the modelling results was done as part of the process in GRASP. Validation is given as a cross-validation Receiver Operating Characteristic Curve (cvROC). If the prediction is no better than random, the ROC gives a value close to 0.5, and a perfect prediction gives a ROC value of 1.0. Values around 0.8 can be considered good, and above 0.9 is very good. None of the models used were validated with external data, but the Full model represents the status of the knowledge concerning *L. hyperborea* distribution in the Skagerrak.

Application of habitat maps in management. A great advantage of GIS based predictions such as this is that they are suitable to use in management of marine environments since the relevant authorities also uses GIS. They thereby provide a direct link between research and management. The results of this specific project may be used in the management of the kelp forests in Skagerrak for information on where important kelp sites can be found and could be monitored.

6.5.5 Conclusion and perspectives

The design for collecting field data is crucial for modelling and making spatial predictions. Dive transects are cost-effective since they describe the whole depth gradient in detail. To gather this information by using point inventories instead of transects takes much more effort. However, sampling sites must also cover gradients of other ecologically important parameters.

Wave exposure is one of the most important factors structuring the shore community (Lewis 1964) and field data for a coastal model should therefore include the full range of wave exposure variation that occur in the model area. In the present study, KYO stations were not covering the whole exposure gradient, which is thought to be the main reason for the less accurate predictions based on this dataset.

6.6 PA1. Spatial prediction of Nephrops norvegicus on the Swedish Skagerrak

Authors: Isaeus, M., Carlén, I., Nilsson, H., Sköld, M.

6.6.1 Introduction

The goal of this study was to examine the habitat distribution of the Norwegian lobster, *Nephrops norvegicus* in Pilot Area 1. This was done by a new approach, using field data of frequency of potential lobster burrows from Sediment Profile Images (SPI) as the response variable in spatial modelling.

Background

Spatial modelling is a method that complements field surveys since the spatial distribution of most marine species are not possible to map in a cost-effective way and therefore often only point data is available. Modelling is the only efficient way to create surface covering maps using point data.

Sediment Profile Images is a cost-effective way of acquiring point data on structures in the bottom sediment.

The species. The Norwegian lobster (fig. 1) is common in the East Atlantic and adjacent seas, as well as in the Mediterranean. It lives on muddy bottoms where it digs its burrows. The Norwegian lobster is ecologically important as a habitat structuring species causing bioturbation and oxygenation of the sediment. The species is mainly nocturnal and feeds on detritus, other crustaceans, and polychaetes. It is important commercially throughout it's range, with annual landings of approx. 60 000 t (Holthuis 1991). The species is fished using trawls, nets, and stationary cages.

Aims

The aims of this study were to investigate:

1. Spatial modelling of the Norwegian lobster habitat by using presence of potential lobster burrows.

2. Suitability of Sediment Profile Images (SPI) as model input.

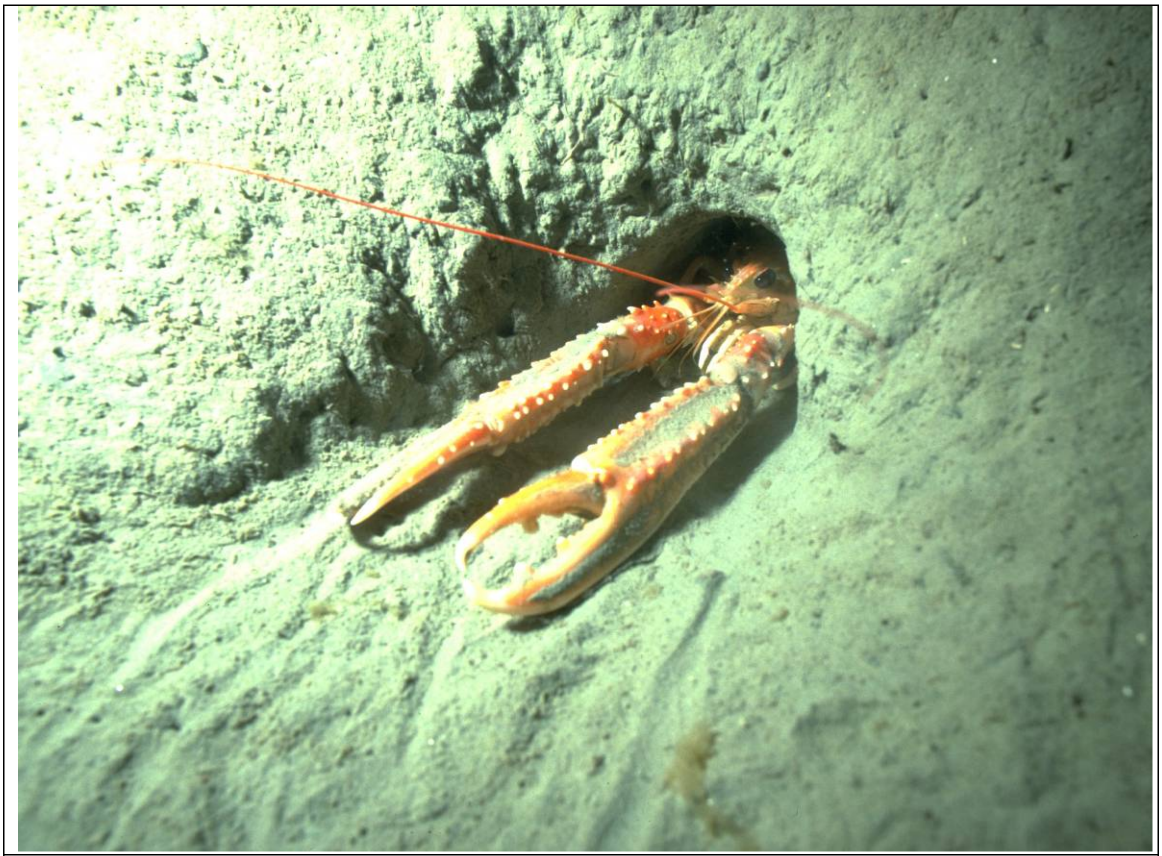


Fig. 1. *Norwegian lobster (Nephrops norvegicus) in its burrow. Photo: Mattias Sköld*

6.6.2 Material and Methods

Field work was carried out in the Swedish Skagerrak within an area investigated by multi-beam, providing bathymetry and backscatter data on bottom sediments (fig. 2).

Bathymetry and bottom sediments. During 2003, Marin Mätteknik AB was commissioned by the Swedish Board of Fisheries to carry out sea measurements in the Swedish Skagerrak (Marin Mätteknik AB 2003).

Multi-beam echo sounder was used to collect data on bathymetry and bottom sediment type. Bottom sediment type is calculated as the degree of hardness of the seabed based on loss of reflection in the echo sounder data; the greater the loss in reflection, the softer the sediment. Bottom sediments are divided into five classes (tab. 1). For the purposes of this study, the two last classes (hard clay and soft mud) were combined into one class of clay and mud.

Table 1. Classes for bottom sediments used in backscatter calculations.			
Bottom index	**Reflection**	**Typical sediments**	**Intervals for reflection loss (dB)**
Hard	Very strong re-flection	Rock, moraine, gravel	> -6.9
Coarse	Strong reflection	Sand and gravel	-6.9 to -10.2
Fine	Medium reflec-tion	Sand and silt	-10.2 to 12.7
Hard clay	Weak reflection	Clay and loose sediments	-12.7 to -20.0
Soft mud	Very weak re-flection	Soft mud and loose sediments	< -20.0

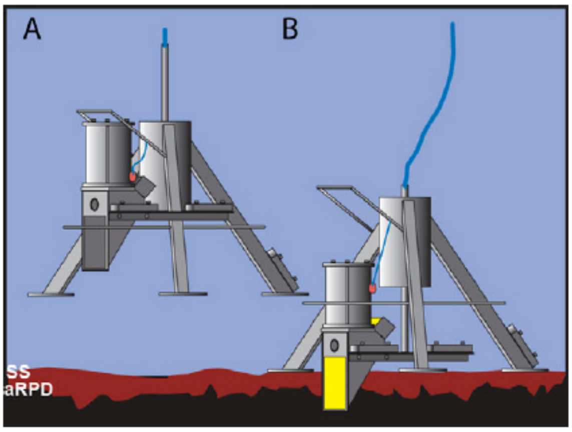


Fig. 2. *Principal sketch of sediment profile camera. To the right it is shown how the prism sinks into the sediment before taking the picture.*

Presence of potential lobster burrows. Sediment Profile Images (SPI) is a method that emerged in the 1970's. Since the 1990's, it has expanded as use of digital cameras and image analysis software made it more efficient. A majority of the use have focused on areas well known to be temporally affected by low oxygen level. However, SPI have also been used in EIA surveys on mussel and fish farms, trawling experiments, extraction and dumping of material, for mapping of drill cuttings around oil platforms, and in benthic monitoring programs incl. habitat mapping (Nilsson & Rosenberg 2006 and references therein). A benthic habitat quality (BHQ) index may be calculated, which is related to ecological status in the Water Framework Directive. It is concluded that use of SPI technique is cheap, rapid and a powerful tool for benthic mapping (Nilsson & Rosenberg 1997). Parameterization of sediment and animal features in the images is quickly and accurately made by digital image analysis.

The sediment profile camera works as an upside-down periscope that penetrates the sediment surface and takes pictures horizontally into the sediment (fig. 2). The image is

17.3 cm wide and 26 cm high, with a typical penetration depth of about 15 cm. The image shows surface and subsurface features of the sediment, such as fecal pellets, tubes, pits, mounds, infaunal structures, burrows, and oxic voids. It is also possible to measure the apparent redox potential discontinuity (RPD).

In this study, 35 stations were investigated by 5 SPI replicates at each station. A density measure of potential Norwegian lobster burrows was determined at each station by number out of the 5 SPI replicate that showed burrows. This results in a frequency of potential lobster burrow presence at each site of 0, 0.2, 0.4, 0.6, 0.8, or 1.0, which was used as the response variable in spatial modelling.

A burrow viewed in a Sediment Profile Image may be defined as a subsurface structure recognized by a distinct halo, often vertical, of more oxic sediment than the surroundings. In this case, not all such burrows have been classified as lobster burrows, but only those which have some characteristics of *Nephrops* burrows as described in Tuck et al. (1994) and Marrs et al. (1996). The burrows are described as potential lobster burrows because it cannot be completely certain that they have been made exclusively by Norwegian lobster; some may have been created by other digging crustaceans such as *Calocaris macandreae* or *Callianassa subterranea*.

Predictor layers. The quality of predictor layers often limits the accuracy of the predictions in spatial modelling. However, in this project we had the opportunity to use high quality layers describing bathymetry and bottom sediment hardness (backscatter data). These high resolution grids were collected in an earlier project (Marin Mätteknik AB 2003). Predictor variables used in modelling included:

V. Bottom sediments (fig. 3).

VI. Detailed bathymetry (fig. 4).

VII. Slope and aspect derived from the bathymetry grid using ArcGIS v9.1.

All predictor variables were available as GIS raster layers in the geographic projection RT90 with a grid cell size of 10 x10 m.

Response variables. As response variable in the modelling, 35 points of frequency of potential lobster burrow presence was used (data also in RT90).

Model selection (methods, algorithms, software, and routines). Modelling was done in GRASP (Generalized Regression Analysis and Spatial Predictions), a set of S-PLUS/R functions developed for modelling and analysis of the spatial distribution of species (Lehmann et al. 2002). GRASP communicates with ArcView, and the resulting predicted distribution maps are in ArcView format.

The model is built upon the relationship between the response parameter, the presence of potential Norwegian lobster burrows in this case, and the environmental variables that are used as predictors. To make spatial predictions, or distribution maps, for GIS, spatial descriptions of all predictors are used as input layers.

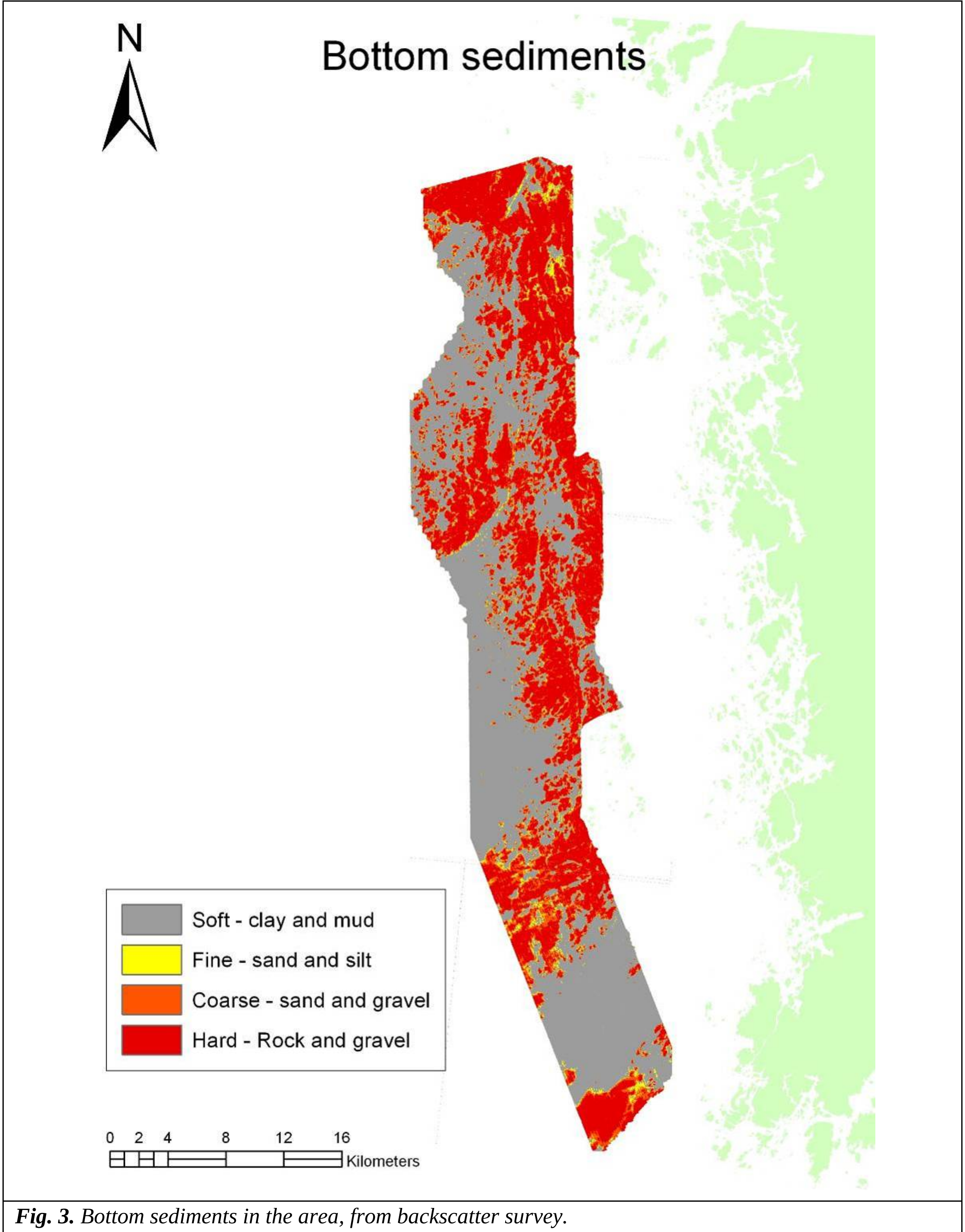


***Fig. 3.** Bottom sediments in the area, from backscatter survey.*

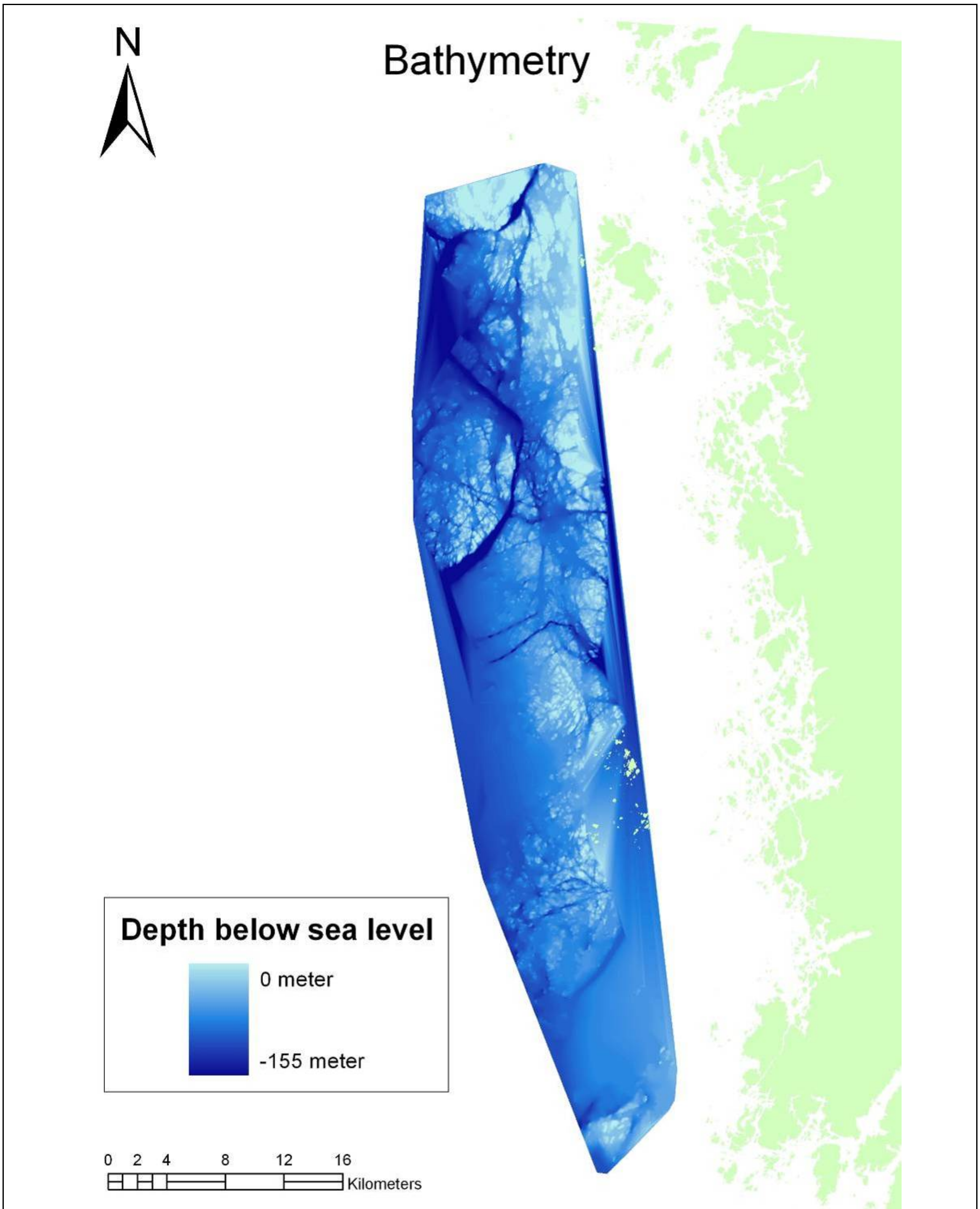


Fig. 4. Bathymetry in the study area.

Habitat models used

GRASP uses GAM, generalized additive models (Hastie & Tibshirani 1986) to fit predictor variables independently by non-parametric smooth functions. The best model is selected through a stepwise procedure where successively simpler models are compared with a measure such as the Akaike Information Criteria (AIC). GAM has proved particularly robust in modelling species presence/absence data (Yee et al. 1991, Franklin

1998, Bio 1998). In this case, a binomial model with two degrees of freedom to fit the spline function was created in GRASP, using the frequency of burrow presence and all available predictors. After modelling of the predicted distribution the projection was transformed to UTM34N.

6.6.3 Results

The resulting map shows the predicted probability (between 0 and 1) of presence of Norwegian lobster burrows (fig. 5).

The predictor with the strongest influence on the model was the bottom substrate (BS in fig. 6). This can also be seen when comparing the predicted distribution map in fig. 5 to the bottom sediment grid in fig. 3. However, depth also contributes significantly to the model.

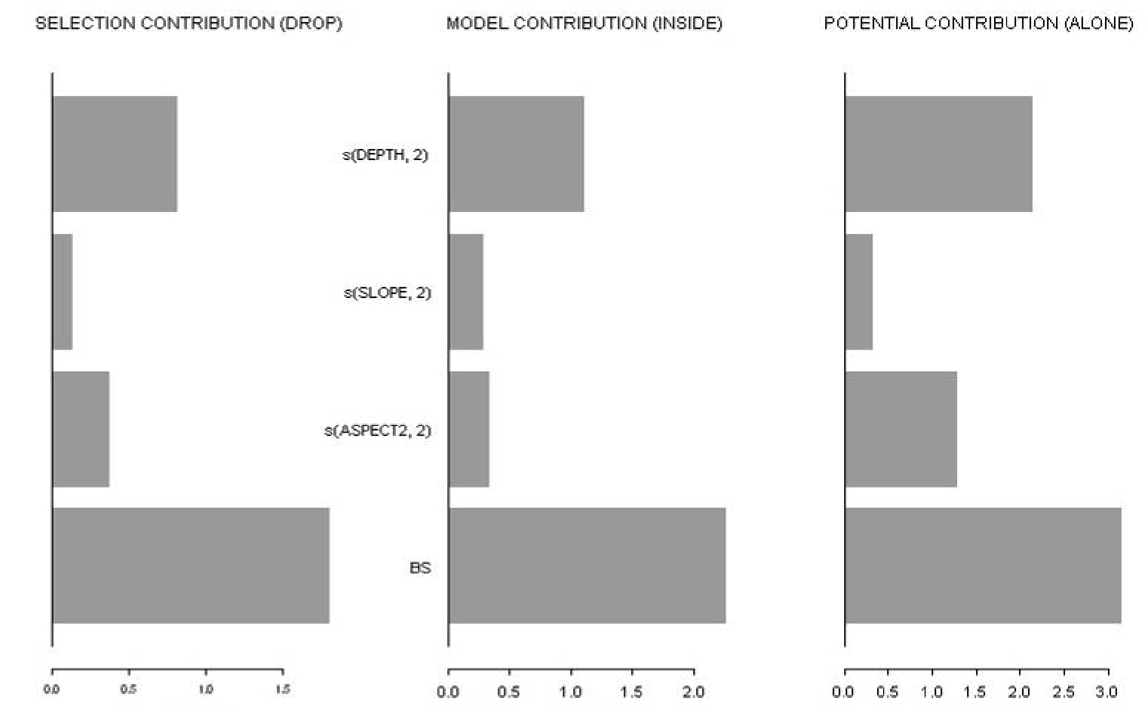


Fig. 6. *Relative contribution of predictor variables to the model. The left bars represent the drop in explained deviance when the variable in question is dropped from the model. The middle bars represent the range on the linear predictor scale, and the right bars show how much of the deviance is explained by the variable if it's used alone in the model.*

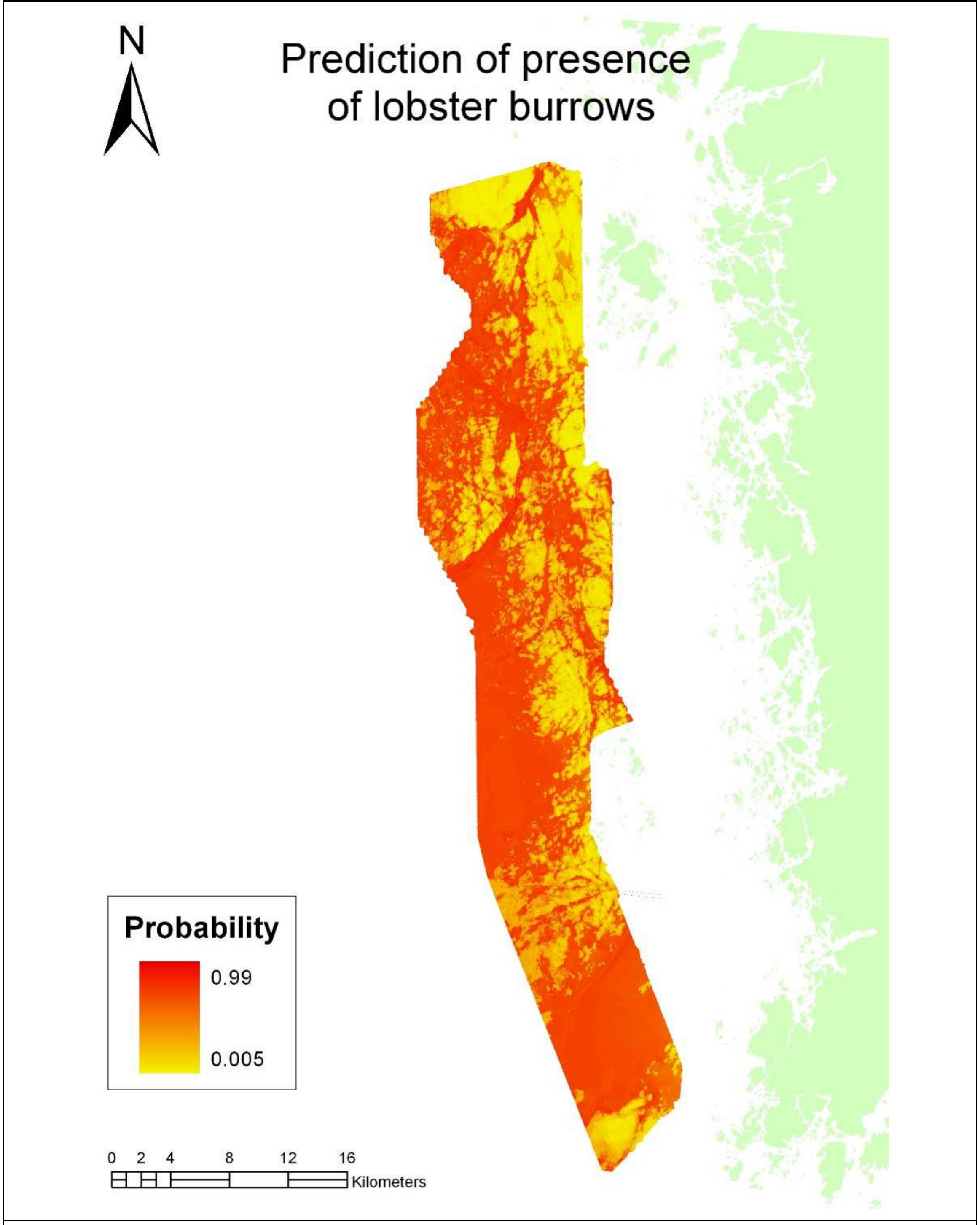


Fig. 5. *Predicted probability of presence of lobster burrows.*

6.6.4 Discussion

Quality of data and methods

The SPI method of identifying lobster burrows will need to be evaluated before being used in larger scale. The first step would be to validate SPI using video recordings, to verify whether the method distinguish properly between lobster burrows and excavations made by other organisms. In this study, burrows made by other burrowing crustaceans, such as *Calocaris macandreae* or *Callianassa subterranean*, may have been classified as *Nephrops norvegicus* burrows.

The high resolution data on bathymetry and bottom substrate are an excellent foundation for predictive modelling of the distribution of marine habitats. Multibeam and backscatter surveys are relatively inexpensive. The coverage of such data is steadily increasing in Finnish, Norwegian, and Swedish waters. The Norwegian lobster is known to prefer soft bottoms, where it digs its burrows. It has been shown by Tuck et al. (1997A) and Chapman & Bailey (1987) that density of *Nephrops* is correlated to particle size composition of the sediment. Hence, backscatter data on bottom substrates is a useful predictor variable when modelling this species, as it describes the bottom characteristic that is most important to the lobster distribution.

Other predictor layers could be useful in a study like this. For example currents may have an influence on the distribution of lobsters, affecting both the sediments they live in and their food supply. While the present model was clearly satisfying, perhaps the addition of current data could make it even stronger.

Quality and validation of models. Validation of the modelling results was done as part of the process in GRASP. The validation is given as a simple Spearman correlation coefficient between observed and predicted *Nephrops* burrow presence with $r_s = 0.659$. External validation was not performed, as no separate validation data set was available.

The sample size of 35 SPI points in this study is rather small. The fact that bottom substrate and depth are the most important predictors suggests that the model is realistic but may be improved further by using more SPI data in building the model.

Application of habitat maps in management. A great advantage of the GIS based predictions is their suitability for planning and management, including of the marine environment. It provides a direct link between research and management and can be used as a tool in developing of nature conservation and sustainable fishery. The results of this specific project may be used in the management of the Norway lobster stock in Skagerrak for information on where important lobster habitat can be found.

6.6.5 Conclusion and perspectives

Estimating the density of *Nephrops norvegicus*. The size and complexity of lobster burrows depend on trawling intensity, with larger "gallerias" in non-trawled areas and single tube shaped burrows in intensively trawled areas (M. Ulmestrand, pers. com.). This affects the detection rate of the SPI sampling. A script has been developed to analyse the detection rate of different burrow-types by simulating random SPI sampling in GIS (fig. 12). Different shapes of burrows and corresponding occupation rate of Norwegian lobsters are known (Tuck et al. 1994, Marrs et al. 1996, 1998, Tuck et al. 1997B and references therein). However, to use this knowledge in modelling of the Norwegian lobster density, trawling intensity has to be mapped as well. This can be done by using VMS data. The results of the methods above should be compared to results using other methods for estimating burrow and *Nephrops* density, particularly those using underwater video recording (Bailey et al. 1993, Marrs et al. 1998, Smith et al. 2003).

Modelling of the marine environment. When modelling species distribution in the marine environment, it is important to design the field work to fit the model building requirements. For the model to accurately describe the variation of the target species, it is necessary to cover the whole range of the environmental variables. For example, if the model should be valid from 5-50 meters depth, this whole gradient should be present in the field data, even though the target species is only present between 10-15 meters. If part of the gradient is missing from the field data, the model and hence the predictions in this span will likely be inaccurate.

6.7 PA1. Spatial prediction of nursery grounds for juvenile flatfish in the Danish Kattegat

Author: Claus R. Sparrevohn.

6.7.1 Introduction

In recent years the need for an integrated coastal zone management that involves stakeholders and takes biologic aspects into account has received increasing attention. One way of approaching this need has been to implement spatial based management tools such as Marine Protected Areas. In order to set up such management practice the need for more knowledge on the spatial distribution of important areas and nature types is obvious. The coastal zone is an area characterized by a high degree of overlap between various stakeholders and several important nature types and habitats. Particularly in the coastal areas there are many stakeholder interests, such as tourism, recreational, leisure, sports and commercial fishing, bird watching, infra structure such as harbours etc. At the same time, the coastal zone is an important part of the marine ecosystem. It serve as a high productive area in terms of primary production, and 75 % of the fish species of commercial or recreational interest have one or more life stages associated with the coastal zone. Coastal water serves as a transition zone for diadromous species in their migration between the marine and fresh waters. An equally important biological feature of the coastal zone is that it constitutes nursery habitats for many fish species, there among flatfish.

For flatfish species a clear relationship between the size of the nursery area and the size of the populations is found and has been formulated as the “Nursery size” hypothesis (Rijnsdorp et al. 1992, van der Veer 2000). Thus, the abundance of these stocks is directly linked to the quantity of the coastal areas that serves as nursery habitat for their juveniles. This is highly important to bear in mind in the planning of use and management of such areas.

In the present study connection between various hydrographic parameters (predictor variables) and abundance of juveniles of three flatfish species are established. From this prediction of abundance maps are visualised in a GIS.

Background
Number of juvenile flatfish was monitored as trawl catches sampled in the area between Skagen (57°43’06N, 10°41’28E) and Djursland (56°24’30N, 10°59’28E) along the Kattegat coast of Jutland, Denmark (fig. 1). Samples collected during 1985-2005 were used

in the modelling. The survey was carried in July and August, but the exact timing differs from year to year. Approximately 60 stations were visited each year, with a variation between years from 8 Stn. in 1987 to 90 Stn. in 1994. The gear used was a 4.5 m wide young fish trawl (Støttrup et al. 2002). Only depths between 1 m and 3 m were trawled, and only where possible, i.e. in areas without stones and larger patches of vegetation. In general the same stations were visited each year, but since no exact geoposition is available for the first period sampled the accuracy is unknown. The towing speed was kept steady around 1 knot and all trawling was taken parallel to the coast line.

All caught fish were length measured to lowest mm and divided into species. Estimating the numbers of juvenile fish in a given area as described above is a time consuming and costly affair. Therefore a statistic correlation between the numbers of individuals at a given locations and specific characteristics at that station are needed. If e.g. a correlation between depth and abundance is established, this knowledge can be used to predict the theoretical abundance within the depth range from where data exists. The results provide the information needed to create distribution maps. The modelling was performed as described below.

The species. Models were made for the three flatfish species: Plaice (*Pleuronectes platessa*), Flounder (*Platichthys flesus*) and Sole (*Solea solea*). For Plaice and Flounder both young of the year (YOY or 0-group) and 1 year old (1-group) was analyzed, but since Sole catches of the 0-group was very limited only the 1-group was analyzed for this species.

Aims. The aim was to identify predictors that have a significant correlation with the distribution of juvenile flatfish in a Danish shallow Bay (incl. NATURA 2000 site No. **). From these statistical correlation maps were created showing the abundance of fish within the area (resolution: grid cell size of 100 x 100 m).

6.7.2 Material and Methods

The study area. The study area is characterized by a predominantly North-South oriented long stretch of straight coastline and a smaller path of East-West oriented coastline (fig. 1). Although sheltered from the westerly winds, the fetch is large if the wind is easterly. The bottom sediment consists primarily of sand with stones and scattered patches of vegetation, but no detailed sediment map is available for the area. Since the Kattegat serves as a transition zone between the low saline Baltic proper and the high saline North Sea, the range of salinity and water temperature is wide and varies with climatic changes (tab. 1).

Table 1: Hydrographical characteristics for the area model in present report. Summer is the months June, July, August and winter January and February. T is temperature is in ºC and S is salinity in ‰. Numbers in brackets are the range. All data are from The National Environmental Research Institute homepage: http://www.dmu.dk/International/Water/Monitoring+of+the+Marine+Environment/MADS/. Only data gathered after 1990 are included. The station is st. 4410 was where the depth was 11 m.				
	Summer		Winter	
Depth	T	S	T	S
0-1 m	17.2 (11.4-22.4)	22.2 (15.3-29.8)	2.8 (-0.7;5.1)	25.4 (19.5-29)
4-5 m	16.2 (9.5-21.3)	23.1 (19.2-26.8)	2.9 (-0.6-5.8)	26.1 (21.8-29.3)

Predictor layers
To use an abiotic variable as predictor of abundance or habitat distribution outside the area from where the biological information (biotic or response variable) has been collected, requires a range of the predictor variable equal to or larger than the range found within the entire area. For example, if depth is used as a predictor then it is not sensible to predict a habitat distribution outside the depths range sampled. This is important to take into account before setting up the model. In the present study, depth was not included as a predictor variable as it could not be used for areas of depth <1 m or >3 m. Unfortunately an obvious predictor variable was not been used in the modelling, sediment composition. This parameter was not included simply because it was not available at a scale fine enough for the purpose. Type of substrate at a resolution of ** x ** (or finer) could prove to be one of the best predictor variables as close correlation between flatfish and sediment composition is known from **(review in Gibson 1994).

I. Wave-exposure/energy loss (By Doris Mühlestein, DHI) (Exposure). In shallow waters (from 0 to10 m depth) wave exposure is believed to be one of the most important variables determining abundance or organisms. Wave conditions at a given location can be determined by:

- Fetch (stretch of sea within which the waves are generated upwind from the location),
- or duration of the wind blowing with a given wind speed,
- and, in both cases, also by wind speed and depth of water along the fetch line.

For all but very high wind speeds, the fetch is the limiting factor in the Kattegat area. Therefore, wave simulations assuming stationary wind conditions and calculating wave states corresponding to fully developed sea for the given wind conditions have been carried out. The East coast of Jutland is exposed to wind waves from directions N through E to S. In order to cover both normal wave conditions and extreme wave conditions, the following 35 wind conditions were simulated:

- Directions (5): N, NE, E, SE, and S.

- Wind speed (7): 2.5, 5, 7.5, 10, 15, 20, and 25 m s^{-1}.

Besides the main model input, which are the wind conditions and bathymetry, the model uses different variables to describe various phenomena, such as wave breaking and bottom friction.

The energy loss (E_5), or rate of energy dissipation due to breaking, is defined by

$$E_5 = \int_0^{\omega} S_{breaking}(\theta) d\theta$$

where $S_{breaking}$ is the rate at which the directional integrated energy density is dissipated due to wave breaking.

II. Slope in percent (Slope). The Slope was calculated from the corresponding depths in percentage using a resolution with a grid size of 100 m x 100 m.

III. Distance from shore to the 5 m depth curve (Distance). Distance from the shore to the 5 m depth curves measured perpendicular to the orientation of the coastline was estimated.

IV. Distance from sampling place to the nearest coast (Place). Distance from the sampling position to the nearest coast in meters. This predictor was the only ones that were included in the model as a linear effect.

V. Year (Year). Year was included in the model to account for any inter-annual variation. The predictor variable Year showed significance for most age-groups and species. Thus, a specific year, 1995, was chosen arbitrarily as basis for the final GIS habitats maps.

VI. Number of Sand Banks (Banks). The number of Sand Banks along the coastline was counted from airplane photos for each 100 m of coastline.

Response variables. In the modelling of flatfish abundance in BALANCE the success criteria of the model was to be able to predict the abundance of a given species at a given location within the pilot area 1. This raises the problem, how to transform our catch data into real abundance data. The information that we get directly during the data sampling, when fishing standardized is the catch per unit of effort (CPUE). This is proportional to the abundance but exactly how CPUE and abundance is linked is not known, i.e. we do not know the fraction of individuals that has been in contact with our fishing gear that is actually caught. Sparrevohn & Støttrup (submitted) conducted a series of experiments with turbot (*Psetta maxima*). Based on a method identical to the one used in catching of juvenile flatfish in this study, they estimated that for turbot the fraction of fish retained in the trawls was depending on the fish size. For small individuals (length = 4.5 cm) the fraction was 40 % and decreased to 26 % and 11 % for fish of 11 cm and 17 cm length, respectively. Although species specific variation should be taken into account, the best that could be done in the present study was to use the figures found for turbot and using these to calculate the absolute abundances of flatfish from the catches. It is assumed that 40 % of the age group 0 was caught and 26 % of age group 1.

Model selection (methods, algorithms, software, and routines). For modelling of the flatfish catches a negative binomial distribution model was used. A Poisson model was not used here as the variance in general turned out to be much higher than the mean. The models were evaluated in three different ways. A normal stepwise elimination of non-significant effect was chosen, where the significance level was set to be 0.05. Next the models were compared using the Akaike Information Criteria (AIC). Last the percentage of the deviance explained by the models were also evaluated and compared between models as they were reduced.

For all flatfish the first model evaluated was:

X ~ s(log(Exposure+1), k)+s(Slope, k)+s(Distance, k)+s(Year, k)+s(Banks, k)+Place

where s is the spline term and k is the dimension of the basis used to represent the smooth term in the spline estimation.

6.7.3 Results

For two out of the five species models it was not possible to reduce the full model, i.e. all six predictors were contributed significantly to the model (tab. 2 and 3).

Table 2: The Akaike Information Criteria (AIC) for the full model and for models where one predictor has been removed. The smaller the AIC the better the model is assumed to describe the data. The asterisk indicates the model that the elimination procedure chose as being the best to describe the data and the † indicates the model chosen for map creation.

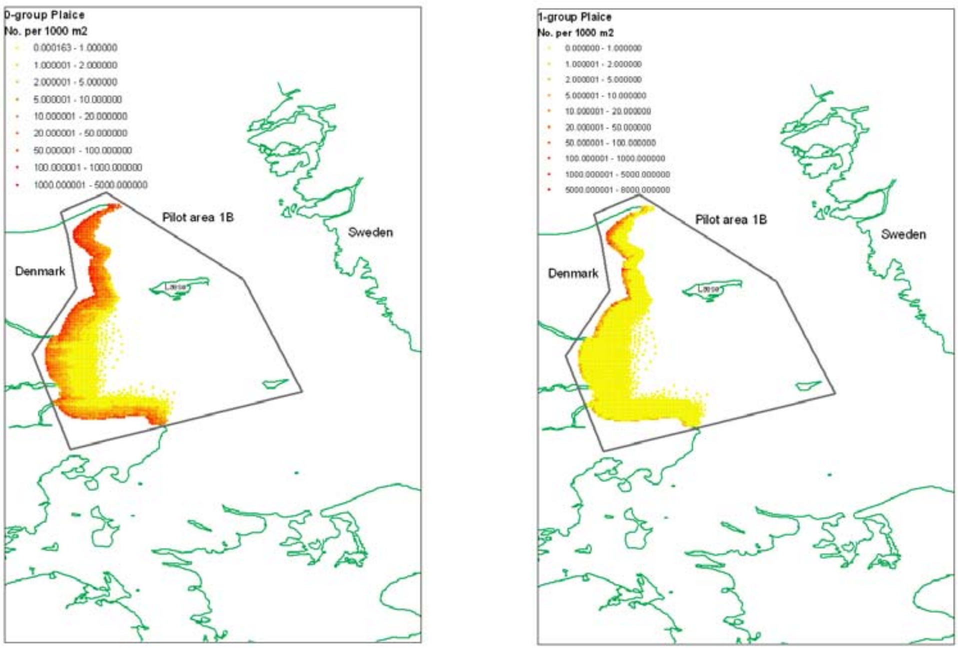
	Plaice		Flounder		Sole
Model	0	1	0	1	1
Full model	9148	3340*†	NA	4618	6755*†
Reduced model:					
-Year	9148*†	3496	3782	4790	6768
-Exposure	9197	3387	NA	4618	6786
-Slope	9218	3404	NA	4616	6802
-Place	9149	3375	NA	4620	NA
-Sand Banks	9174	3397	NA	4608†	6824
-Year, -Place, -Distance			3783*†		
-Place, Slope and Sand Banks				4610*	

Table 3: Deviance explained in percent for the full model and for models where one predictor has been removed. The asterisk indicates the model that the elimination procedure chose as being the best to describe the data and † indicates the model chosen for map creation

	Plaice		Flounder		Sole
Model	0	1	0	1	1
Full model	20.6	54.1*†	NA	19.4	23.9*†
Reduced model:					
-Year	20.4*†	42.9	28	6.8	22.9
-Exposure	17.4	50.2	NA	18.9	21.5
-Slope	16	48.9	NA	19.1	20.4
-Place	20.3	52.2	NA	19.3	NA
-Sand Banks	18.8	49.5	NA	19.9†	19.4
-Year, -Place, -Distance			27.8*†		
-Place, Slope and Sand Banks				19.3*	

For each of the species and age-groups examined one model was chosen to be the best given the predictors at hand and maps was produced. The abundance was predicted in the area from the coastline and until 10 meters depth.

Spatial predictions of 0-group Plaice. The abundance map shoved that the highest concentrations of 0-group Plaice are found along the most northern coastline (**Map 1). Especially the area north of Læsø showed a higher concentration than the rest of the investigated area. Whether these fish originates from the spawning areas North of Læsø, or the spawning areas North of Zealand, is not known.



Map. 1. The abundance of 0-group Plaice to the left and 1-group Plaice to the right. The abundance is estimated using the model shown in Table 1 and 2. The additive effect from the different predictors included in the model and hence used to produce the map is shown in Figure 1 and 2 for 0-group and 1-group place respectively.

Spatial prediction of 1-group Plaice*. Similar to what was observed for the 0-group Plaice, the highest concentration of 1-group Plaice was found in the northern part of Pilot area 1B. The trend was even more explicit than for the 0-group Plaice, as the number of individuals estimated per 1000 m^2 for the 1-group Plaice in the southern part of Pilot area 1B was very low.

Spatial prediction of 0-group Flounder*. For this particular species and age group the model had problems converging, as described previously. This indicates that the predictors included in the model did not fit the data very well. The only three predictors included in the model were number of Sand Banks, Exposure, and Slope and none of these seemed to have any major effect (fig. 4). As a result the estimated abundance seems to be rather dubious.

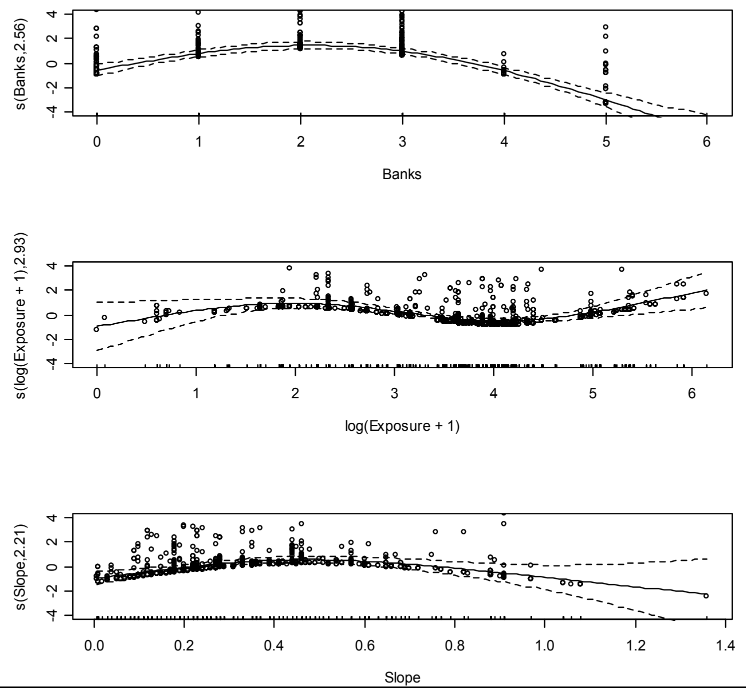


Fig. 4. *On the figure the predicted additive effect (solid line) on the abundance estimate are shown for the 3 significant predictors of 0-group Flounder abundance. The dashed line is the 95 percent confidence interval and the circles the residuals. The number in brackets on the y-axis is the degree of freedoms used to spline the line. The closer the number is to 1 the closer the spline line is to a straight line.*

Quality of data and methods. The gear is appropriate when sampling juvenile flatfish and the variance induced by the gear is not considered to be a problem. This means that on a sandy bottom the trawl will catch a certain numbers of individuals with a good precision. The problem in using trawling for sampling is that the gear is not usable on several types of substrates, e.g. it does not sample efficiently in sea weed or eel grass beds, areas with patches of stones, or in muddy areas. Hence, such areas are not included in the final map, although they are often essential fish habitats.

The two predictors slope and exposure could not be excluded in any of the models set up to describe the juvenile flatfish distribution. The effect of exposure on abundance showed substantial species and age specific differences. For both the 0-group and 1-group Plaice the abundance increased with increasing degree of exposure. For the 0-group Flounder the pattern was not clear but a small tendency for decreasing abundance estimate with increasing degree of exposure was observed for 1-group Flounder, a trend also seen for the 1-group Sole. The predictor slope was also kept in all of the models and in general the effect from slope was u-shaped, meaning that for low and high slopes an increase in abundance was observed. For those models where year was included as a predictor it seemed to have a rather large effect on the estimated abundance. Large fluctuations in 1-group Flounder and 1-group place abundances with year were observed but only minor for 1-group Sole. Place, i.e. the distance from the coastline to the point, was only excluded in the model predicting the abundance of 0-group Flounder. In the remaining models place had a negative effect meaning that the farther away from the coastline the lower the abundance, independent on age and species. The number of sand banks was an important predictor for both 0-group and 1-group Plaice. The lower the number of sand banks observed along a coastline the lower the abundance of this species. The opposite trend was observed for Flounder where higher abundances were observed with fewer numbers of sand banks. For Sole the effect from sand banks was the same as seen for Plaice, but not as noteworthy. The last predictor was the distance, i.e. the distance from the coast and to the 5 meter bathymetry line. For Sole a doomed shaped relationship was found indicating highest abundance of 1-group Sole a medium distances. For Plaice the abundances was highest in the areas where the distance to from the coastline to the 5 meter bathymetry line was low opposite to what was observed for the Flounder.

In general, the prediction of the 1-group plaice was by far the one where the highest signal was observed, but also the 0-group Plaice performed well. It appeared as if they both had a preference for the same areas, located to the north of pilot area 1B, which is an area with high degree of exposure and a coast line that is characterize by numerous sand banks.

Quality and validation of models. The model was not validated, but ideally the model should be validated against an independent dataset. This was not done for the present model since effort was put into identifying which predictors that appeared to be of most important. This was believed to be the first important step. Thereafter, dealing with the actual performance and validating the model should be done. Variation in the temporal scale should also be dealt with in more details in future studies. Question as how the habitat utilization and hence distribution are affected by the abundance of recruits should be investigated.

Validity of identified habitats (confidence assessment). In the present modelling specific habitats are not identified but only the spatial distribution of juvenile flatfish along a coastline.

Application of habitat maps in management. Spatial based marine management tools will most probably play a central role in an integrated coastal zone management in the future. These methods have the potential to take all stakeholders and biological aspects into account and evaluate the various interests in the costal zone on a spatial basis. In order to set up such management practice, maps on the spatial distribution of biological important species are needed, there among the distribution of juvenile flatfish. The maps created in the present study can help to provide the information needed in order to take the right decisions. But cautions should be taken. The process of creating the maps reviled that at least reliable sediment maps are missing. Further, the historical depth data available was not stratified in a manor that made it possible to include in the model. Thus, accurate and fine scale data on both substrate and topography would considerably improve the reliability of the habitat maps of juvenile flat fish.

6.7.4 Conclusion and perspectives

In general, sampling biological data is a very time consuming and resource demanding task. This is also the case in sampling juvenile flatfish and therefore a statistic correlation between abundance and the more easily measured predictors is needed in order to provide a spatial coverage appropriate for mapping. These types of analyses have been carried out with success in many branches of biology (Lehmann et al. 2002a, b) and in the present study the potential was also reviled. It was found that the two predictors slope (i.e. of the seabed) and exposure was the two most important predictors. Exposure was a measurement of the amount of energy conveyed from the waves to the bottom. In addition it was also found that at least two important predictors were lacking, most importantly fine scale measures of substrate and of depth.

6.8 PA2. Modelling of distribution patterns and 3-D pelagic habitats of cod and sprat spawned eggs and adults in the Danish Bornholm Basin

Author: Gerd Kraus.

6.8.1 Pilot Area 2

Pilot area 2 is the Bornholm Bassin, located in the Baltic Proper, East of Bornholm. This area is the main spawning ground for the Baltic cod. Thus the area has been selected for modelling of pelagic habitats of selected fish species.

6.8.2 Introduction

The pelagic ecosystem of the central Baltic Sea has been studied for long time and a considerable amount of data on biology and distribution of important fish species in relation to the hydrographical conditions exists. During the most recent decade the Bornholm Basin has been in focus as it is presently the only spawning ground where cod, the commercially most important fish species in the Baltic Sea, is able to reproduce successfully due to adverse environmental conditions in the eastern basins. Furthermore, the Bornholm Basin represents an important spawning ground for sprat gaining commercial importance due to a strong increase in stock size. Environmentally defined thresholds and preferences for occurrence determine the distributional overlap volume of the two species, which are strongly linked by trophic interactions, i.e. adult cod prey on sprat and sprat prey on cod eggs (Koester et al. 2001).

Detailed knowledge on the spatial extent of essential habitats for different life stages of these fishes and their food organisms is available for the Bornholm Basin and can be used to evaluate the effects of habitat availability and quality on population sizes and structures. Spatial predictive modelling, using key habitat characteristics to identify spatio-temporal regions of particular interest could provide a tool in this respect. Using environmental predictors in GIS models may promote understanding of pelagic ecosystems.

The species. In BALANCE we have focused on the two ecologically and commercially most important pelagic fish species of the central Baltic Sea. The eastern Baltic cod stock and sprat interact in many ways, i.e., they mutually prey on each other, compete for food in specific life stages, reproduce in the same locations, and their spawning periods largely overlap. However, their reproductive strategies differ considerably and this in combination with heavy differential fishing pressure and the complex species interactions lead to diametric patterns in population dynamics of both stocks.

Cod. In the Baltic Sea, two distinct cod stocks exist, the western stock or "Belt Sea cod" (*Gadus morhua morhua* L.) and the eastern stock or "true" Baltic cod (*Gadus morhua callarias* L.). Meristic (Poulsen 1931, Kändler 1944) and genetic studies (Jamieson & Otterlind 1971, Schmidt 2000) as well as tagging experiments (review by Aro 1989) indicated that the two stocks are located West and East of a borderline at 14°30' longitude near Bornholm Island with some overlap in the Arkona Sea (fig. 1). Historically, the Eastern Baltic cod stock is one of the largest in the North Atlantic region (Dickson & Brander 1993) with a long term average of SSB of 400.000 to 500.000 tonnes, whereas the stock level of the Belt Sea cod was approximately one order of magnitude lower. However, due to a combination of increasing fishing pressure and low reproduction caused by unfavourable environmental conditions, SSB and recruitment of Eastern Baltic cod showed distinct time trends with the SSB declining from over 700.000t in the early 1980´s to ~70.000t in 2005 (ICES 2006). At present, the Bornholm Basin is the only cod spawning ground in the central Baltic Sea that allows for successful egg development.

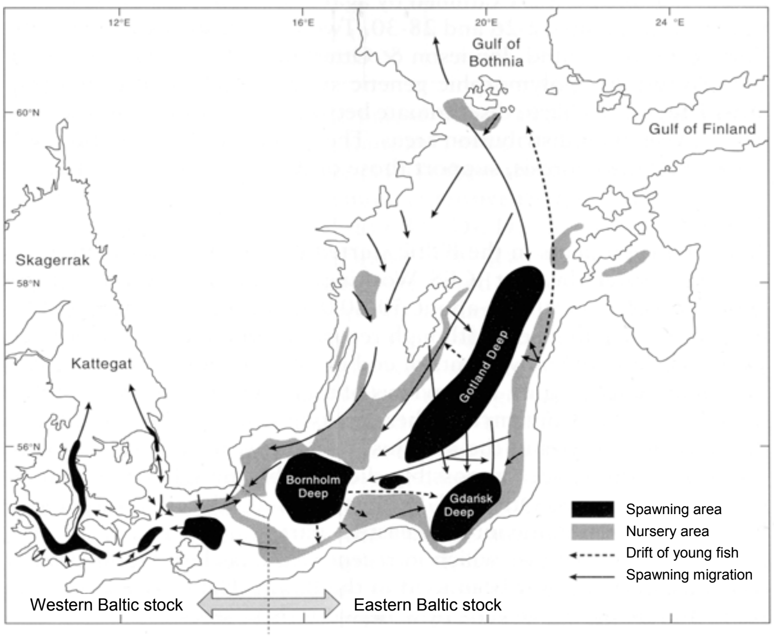


Fig. 1. *Distribution of spawning and nursery areas of cod in the Baltic Sea (redrawn after Bagge et al. 1994).*

Sprat. Sprat is distributed mainly in the open Baltic and the western and central Gulf of Finland. The year class abundance of sprat and sprat predation by cod (*Gadus morhua callarias* L.) are regarded as the chief variables influencing fluctuations in sprat biomass (Aps 1989, Grauman & Yula 1989, Köster et al. 2003, Alheit et al. 2005). In periods of warm winters and good oxygen conditions in the deep layers, the volume of water habitable by sprat increases. This facilitates reproduction, feeding, and normal wintering of sprat and supports its domination in the pelagic layers of the sea.

During the last two decades, in the upper trophic level of the central Baltic Sea ecosystem a shift from a cod-dominated (*Gadus morhua callarias* L.) to a sprat-dominated (*Sprattus sprattus balticus* S.) system was observed (Köster et al. 2003). The corresponding decrease in predation pressure on sprat, combined with low fishing mortality and high reproduction success of this species, resulted in a pronounced increase of the sprat stock (Parmanne et al. 1994).

Aims. In the BALANCE Pilat Area 2, mapping and analysis of a set of hydrographical and biological features governing the spatial and temporal distribution of different cod life-stages in e.g. spawning grounds, nursery areas, and feeding grounds will be carried out, and the application of these findings in area-based management measures, such as zoning of MPAs, is explored. Maps describing the 3-dimensional distribution of the studied species as well as their ambient environmental conditions would allow characterising essential and preferred habitats for predictive modelling.

6.8.3 *Material and Methods*

The study area. The Bornholm basin of the Baltic Sea is the western most area in a series of three deep basins of the central Baltic Sea (fig. 1). Historically, the Baltic cod has aggregated in all three deep ocean basins during spawning, i.e. Gdansk Deep, Gotland Deep, and the Bornholm Deep (PA 2). For successful spawning, the Baltic cod is dependent on sufficient oxygen and salinity levels, and to a lesser degree temperature, in the water column at specific time of the year. However, due to eutrophication and other environmental drivers, the oxygen conditions have in recent years become increasingly unfavourable for cod spawning in the Gdansk and Gotland basins, and the Bornholm Deep has therefore become the only active spawning ground for the Baltic cod in the Baltic Sea (Bagge et al. 1994).

To protect the stock of mature spawning cod against excessive fishery, temporal closure of fishery has been used since 1995. Temporal fishing closure is one of many tools of marine spatial management. Focus of the case study in PA2 is to provide knowledge for use in future spatial planning to protect pelagic habitats of importance for survival of different life-stages of Baltic cod and sprat, i.e. their essential fish habitats (review of concept in **).

Field sampling and data sources

Hydrographic data. Regular hydrographical measurements in the Baltic Sea have been carried out since the beginning of the last century. ICES maintain the largest bank of oceanographic data supplied by Member Countries, dating back to the early 1900s covering the entire Northeast Atlantic. Submission to the databank is subject to intense quality control, thus providing some measure of validation. For the reconstruction of realistic hydrographic environmental variables in the Baltic Sea, a comprehensive database containing the spatial and temporal development of the relevant hydrographic conditions was created from the ICES database and complemented by data from national German and Danish surveys in the Bornholm basin.

Biological field observations. Field studies directed towards the larval stage of cod and sprat have been conducted since 1987 during several cruises each year by the Institute of Marine Sciences in Kiel. In total approximately 125 sampling dates were covered during that period. Less detailed information (not resolved to egg stage) is also available from 1971-1986 onwards (approx. 40 cruises). The horizontal distribution and abundances of cod and sprat eggs and larvae were obtained with a Bongo net equipped with 335 and 500 µm mesh size and sampled with double oblique hauls covering the entire water column. The Bongo (60 cm diameter) was equipped with flow meters in each of the nets. Fish eggs and larvae were sorted from the samples and staged. The counts were finally standardized to 1m² by the volume of water filtered and the maximum depth of the tow (~2 m above the ground). Egg staging was performed according to a 5 stage system based on morphological criteria (Westernhagen 1970, Thompson & Riley 1981), which was adopted for the Baltic (Wieland 1988, Wieland & Köster 1996). Cod larvae were staged following a 10 stage system for Norwegian cod (Fossum 1986). To check the applicability of this staging system, cod larvae from laboratory studies with known hatching date were sub-sampled daily for a period of two weeks to be able to compare the stage/age relationship. No significant deviations in development times were detected. The station grid in use comprised 30 stations in 1987-90, 36 stations in 1991-93 and finally 45 standard stations from 1994 on. The station grid was always sampled around the clock, i.e. catches were obtained both during day and night.

Catch rates, age and sex composition of cod as well as the proportion of mature females in the Bornholm Basin (ICES Subdivision 25, 1995-2003) was obtained from a national German-Danish trawl-survey database including approximately 80.000 records.

Model selection (methods, algorithms, software, and routines)

Hydrodynamic model. In order to obtain temperature, salinity and oxygen conditions at temporal and spatial scales much finer than possible from field observations a hydrodynamic model, based on the free surface Bryan-Cox-Semtner model (Killworth et al. 1991) which is a special version of the Cox numerical ocean general circulation model (Bryan 1969, Semtner 1974, Cox 1984) was used. A detailed description of the equations and modifications made, necessary to adapt the model to the Baltic Sea can be found in Lehmann (1995) and Lehmann & Hinrichsen (2000a). Physical properties simulated by the hydrodynamic model agree well with known circulation features and observed physical conditions in the Baltic (for further description see Lehmann 1995, Hinrichsen et al. 1997, Lehmann & Hinrichsen 2000a). The model domain comprises the entire Baltic Sea including the Gulf of Bothnia, Gulf of Finland, Gulf of Riga, as well as the Belt Sea, Kattegat, and Skagerrak. The horizontal resolution is 5 km, with 60 vertical levels specified. The thickness of the different levels is chosen to best account for the different sill depths in the Baltic Sea region.

The Baltic Sea model is driven by atmospheric data provided by the Swedish Meteorological and Hydrological Institute (SMHI: Norrköping, Sweden) and river runoff taken from a mean runoff database (Bergström & Carlsson 1994). The meteorological database covers the whole Baltic Sea drainage basin with a grid of 1° x 1° squares . Meteorological parameters, such as geostrophic wind, 2-m air temperature, 2-m relative humidity, surface pressure, cloudiness, and precipitation are stored with a temporal increment of 3 hours.

Prognostic variables of the hydrodynamic model are the baroclinic current field, the three-dimensional temperature, salinity and oxygen distributions, the two-dimensional surface elevations, and barotropic transport. These prognostic variables were extracted from the model every 24 hours, and formed the geolocation database for the subsequent analysis.

During the preparation of the maps for the geographical distribution of ichthyoplankton and fish data, the point data was interpolated for visualization purposes. The interpolation method used was nearest neighbour and the grid size was 500 x 500 meters. The same interpolation method and grid size were used interpolating the proportion between female and male cod.

Habitat models (incl. input data and GIS selection criteria). Threshold levels of environmental variables (here temperature, salinity, and oxygen concentration) forming the physiological preferences or boundaries for the distribution of adult cod and sprat as well as their early life stages are readily available. These physiological thresholds have been applied to 3D-hydrographic model output considering temperature salinity and oxygen conditions, thus, yielding for each time step of the hydrodynamic model a three dimensional pelagic habitat. The following physiological threshold values are available at present and have been considered in the pelagic habitat models (tab. 2).´

Habitat preferences of adult cod

(EU-Project "CODYSSEY", Final Report to the European Commission, Q5RS-2002-00813)

Oxygen:	>60 % saturation	no impact	(green light)
	35-60 % saturation	reduced vertical activity	(yellow light)
	< 34 % saturation	spend limited time there	(red light)
Salinity:	>7 psu	no impact	
Temperature:		no impact	

Viable cod egg habitat

(reproduction volume in MacKenzie et al. 2000)

Oxygen: >2 ml/l

Salinity: 11 psu

Temperature: >1.5 °C

Oxgen related cod egg survival function

(Köster et al. 2003)

y = 1.0808 * (1 – e –(0.5833*x)) 8.0099

y = Relative viable hatch

x= oxygen content in ml/l

Sprat egg bouyancy

(Nissling et al. 2003)

Bornholm Basin April:	1.01092 g/cm -3	SD: 0.00084
Bornholm Basin May/June:	1.00880 g/cm -3	SD: 0.00151
Gdansk Deep May/June:	1.00865 g/cm -3	SD: 0.00118
Gotland Basin May/June:	1.00826 g/cm -3	SD: 0.00084

Adult sprat habitat preferences

(Stepputtis 2006, PhD thesis, Univ. Kiel)

Oxygen:	<1.0 ml/l	lower distribution limit (red light)
Temperature	<5.1 °C	upper distribution limit (red light) (cold intermediate winter water layer)

6.8.4 Results

Patterns in temperature, salinity, and oxygen in the Bornholm Basin.

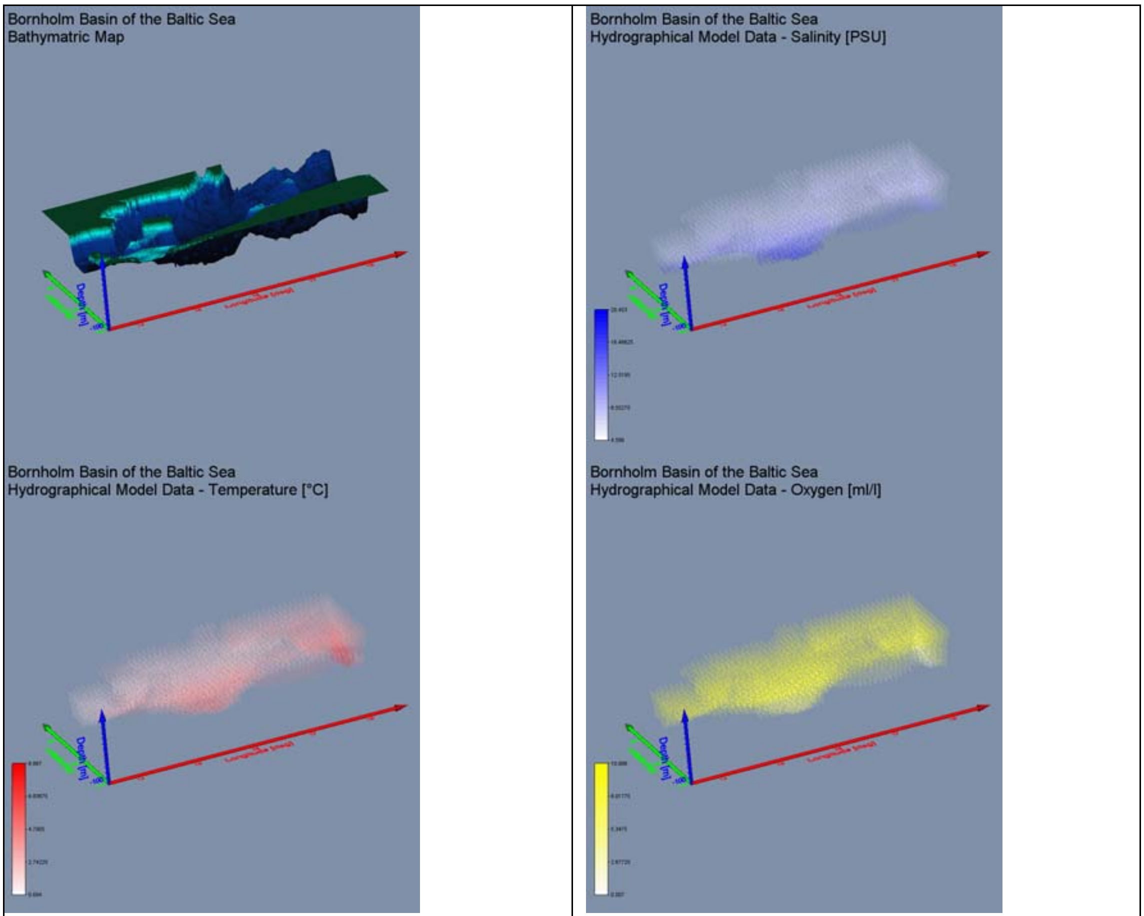


Fig. x. *3D-model of the Bornholm Basin with examples for T, S and O2 distribution clouds from hydrodynamic model output.*

Intra- and inter-annual patterns in horizontal distribution of cod eggs and larvae.

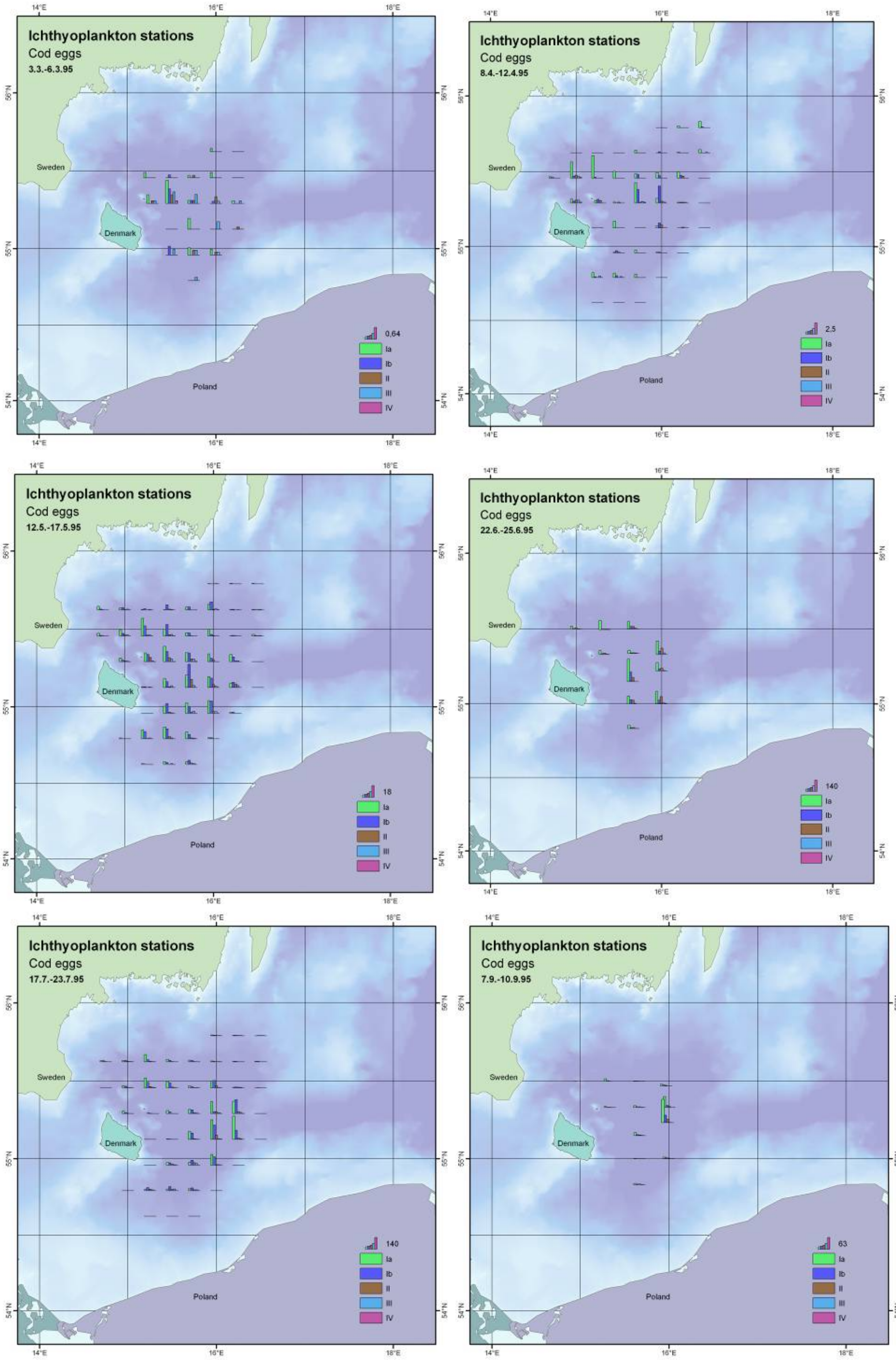


Fig. **. Seasonal development of cod egg and larval abundance in the Bornholm Basin. Bars indicate different egg stages and larvae. An example from 2005.

Distribution patterns of adult cod in relation to season and sex.

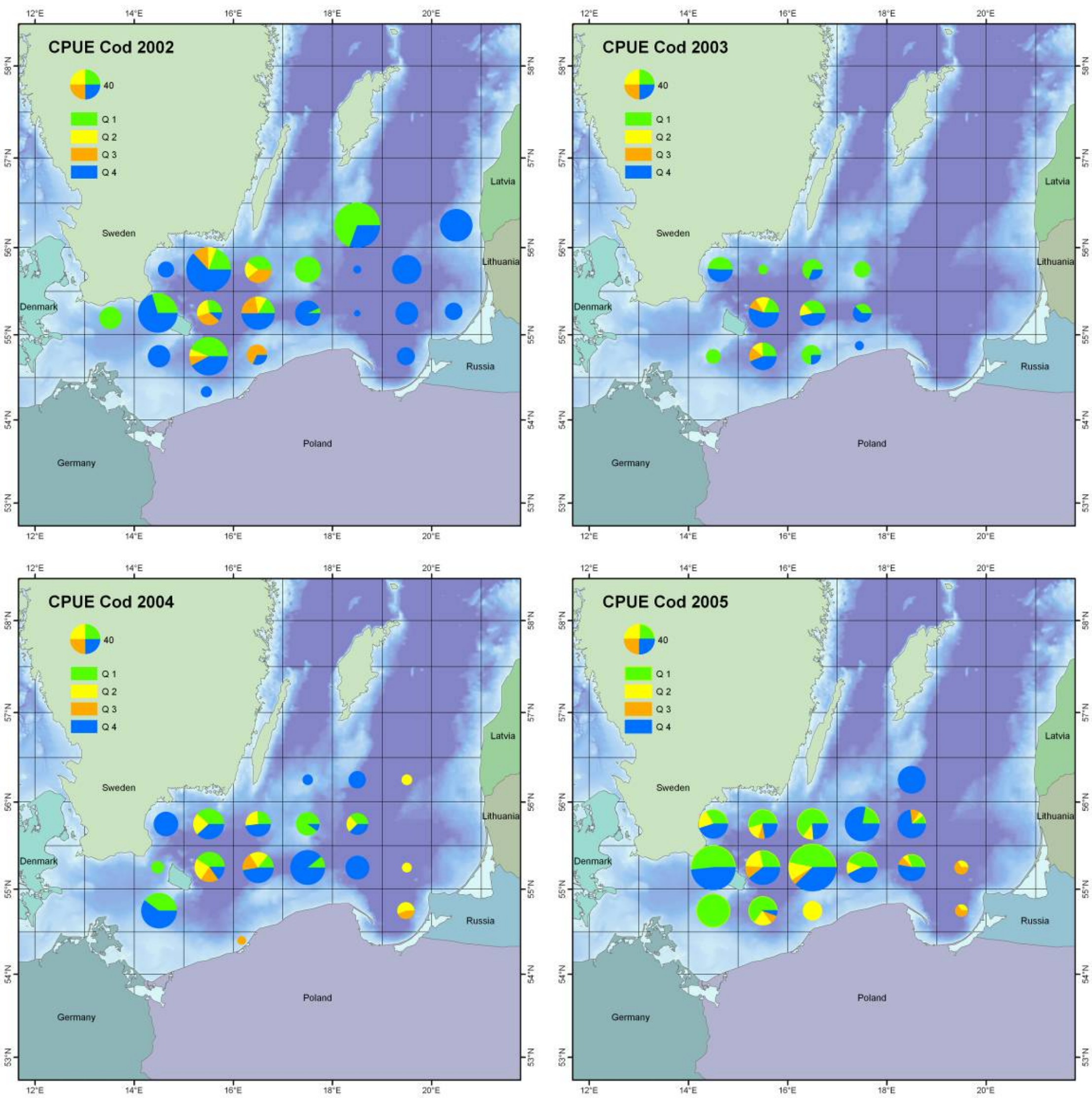


Fig. x: Relative quarterly distribution cod CPUE values from trawl surveys in ICES SD's 25-26. Examples for 2002-2005.

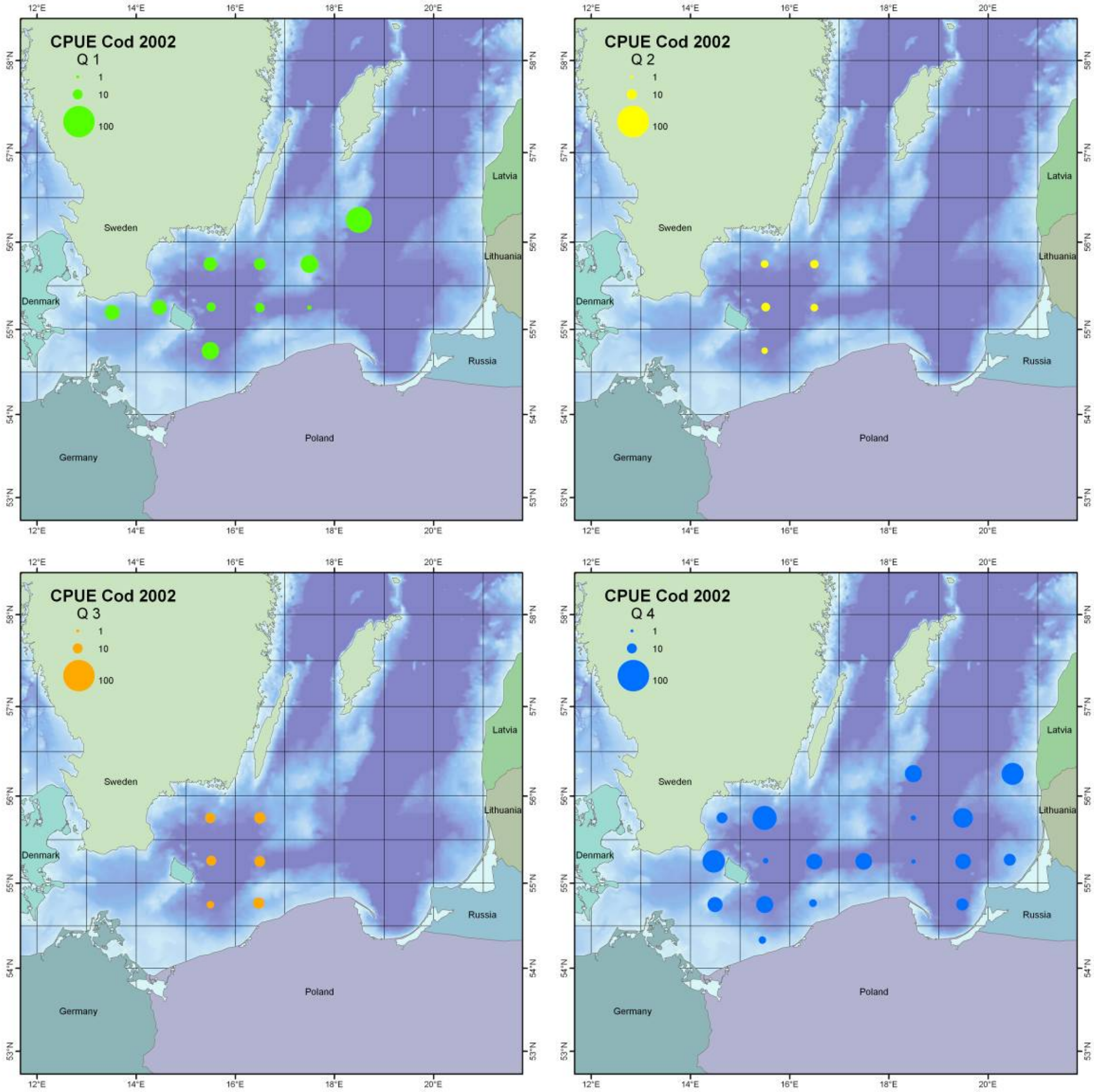


Fig. x: Seasonal changes in cod distribution from trawl surveys in ICES SD's 25-26. During 2nd and 3rd quarter a concentration of the stock in the deep Bornholm basins can bee seen as an indication of spawning activity. An example for 2002.

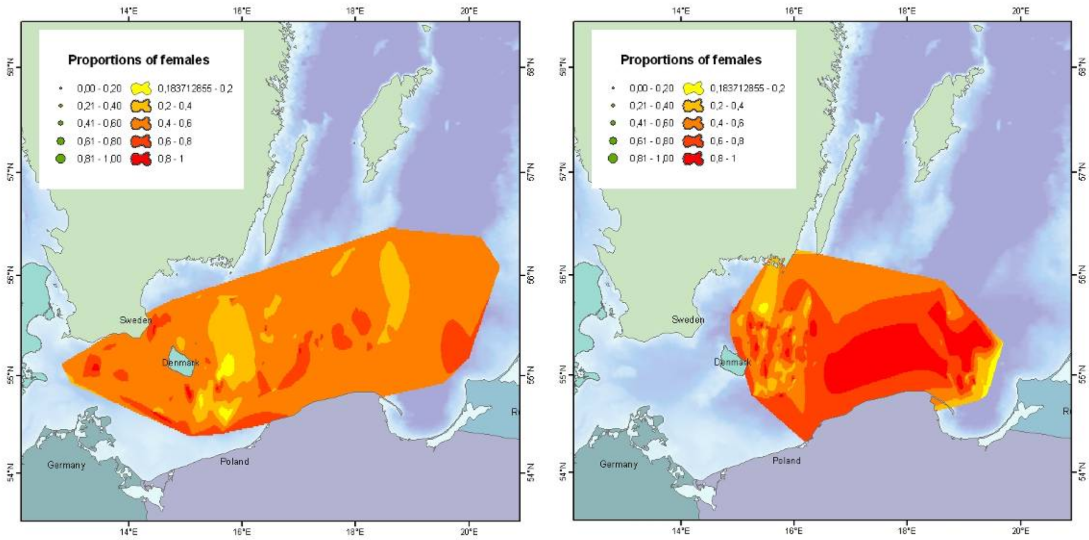


Fig. x: Changes in female distribution between November (left) and July (rihght) derived from trawl surveys in ICES SD 25. During peak spawning time in July the proportion of females in the deep basin is higher than in November. An example for 2002.

3-D habitat maps based on observed distribution patterns and modelled hydrography.

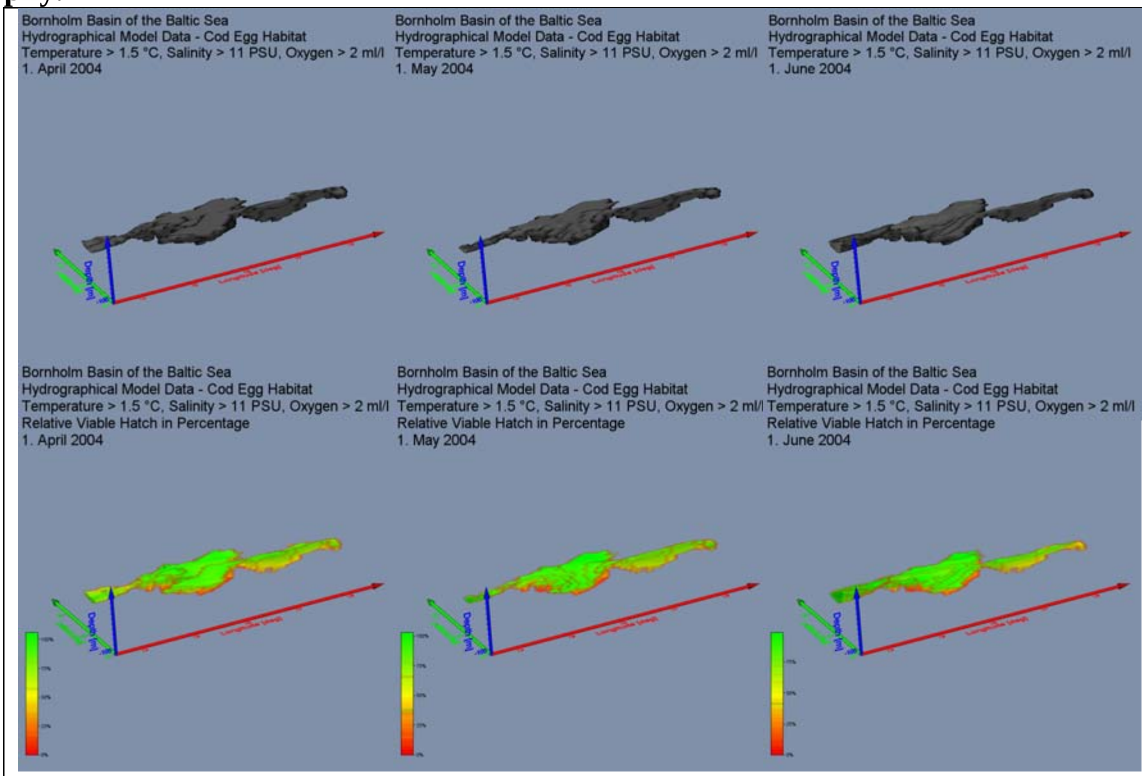


Fig. x: 3-D visualization of the seasonal development of cod reproduction volume and relative viable hatch (reproduction volume corrected for oxygen related egg survival) of cod eggs in the Bornholm basin. An example for 2004.

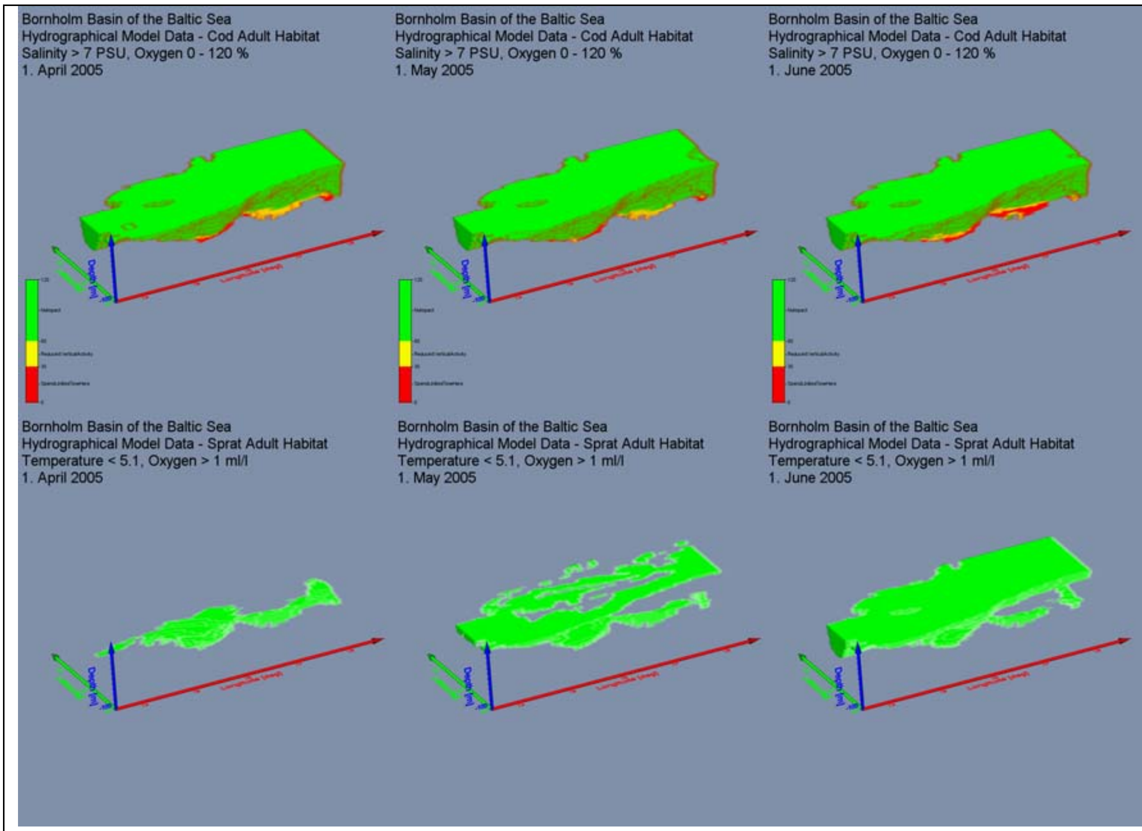


Fig. x: 3-D visualization of the seasonal development of adult cod (top panel) and sprat (lower panel) habitats. Whereas the cod habitat declines from April to June due to oxygen depletion in the bottom water, the sprat habitat increases during the same period of time due to increasing water temperatures in the intermediate and surface water. An example for 2005.

6.8.5 Conclusion and perspectives

Information available at ICES databases can be used to quantify the spatial heterogeneity of the environmental conditions associated with successful spawning of cod and sprat in the Bornholm Basin. Recruitment of Baltic cod critically depends on egg survival (e.g. Köster et al. 2001). It has been recognized earlier that hydrographic conditions in the central and eastern Baltic are critical for successful reproduction of cod and that the inflow of saline and oxygenated water from the North Sea is a prerequisite for the formation of strong year classes (e.g. Kosior & Netzel 1989, Bagge et al. 1994). In order to evaluate this hypothesis, the different data sets presented here offer the opportunity to quantify the water volume suitable for successful development of eggs, which represents a measure of suitable habitat size (Plikshs et al. 1993, MacKenzie et al. 2000). Direct determination of spawning habitats is often limited by low numbers of observations. However, data sets, which contain a considerable number of spatial cod and sprat egg distributions with correspondingly measured environmental data are provided in addition to evaluate the potential spawning habitats. With help of statistical methods it can be evaluated if environmental variables are randomly related to the spawning locations or if they present significant habitat choices. Such characterization of spawning habitat of Baltic cod would allow the identification of processes which are likely to alter the size and/or quality of the reproductive volume of Baltic cod. Furthermore, spawning habitat size and properties may also be predicted in absence of biological data (egg or fish abundance) through physical field observations, i.e., the method might be useful to allow regular monitoring of space-time variability of size and location of the spawning habitat.

Analyses of the databases have the potential to improve the understanding of horizontal movements of Baltic cod and sprat in relation to environmental factors in order to provide information on their spatial availability, accessibility, and individual vulnerability of cod to fishing activities during spawning. A specific objective to be tested is the hypothesis that patterns of horizontal distribution of cod vary systematically, and that the variation is the consequence of behavioural responses to environmental factors. A direct application can be seen in a characterization of spatial and temporal variability of eastern Baltic cod spawning habitats in the light of implemented closed areas to ensure undisturbed spawning.

6.8.6 Acknowledgement

Our sub-contracting parties IFM-Geomar (Kiel), H3 (Lindau), and SFI (Gdynia) are gratefully acknowledged for the contribution of models and datasets utilized in the mapping exercises presented in this report.

6.9 *PA3. Spatial prediction of* Fucus vesciculosus, Mytilus trossulus, and Zostera marina *in the Archipelago Sea, Finland*

Author: Anna Nöjd.

6.9.1 Introduction

No maps are yet available of the potential distribution of the main habitat building communities in coastal waters of Finland. In this study a first attempt has made to demonstrate and develop one approach to modelling the probable distribution of mussel bottoms, algal communities and angiosperms in a part of the Archipelago Sea. Of the many modelling methods available, generalised additive models (GAMs) were selected as the preferred method. The main aim was to come up with working models to predict the probability of presence for Mytilus trossulus, all algae together and all angiosperms together.

The spatial extent for First paragraph, text text text (Author et al. yyyy), text text text (Author & Authour yyyy). Text text text (http://text/text).

. Text text ** m (table x) and (fig. x).

6.9.2 Material and Methods

The study area

The availability of data limited case study area in the Archipelago Sea to a very small area, covering a 10 km x 10 km map square known as Ormskär (fig. 1). The area is located in the outer part of the archipelago and has varying topography and substrate characteristics.

I will put the images in afterwards…
Fig. 1 *The study area for habitat modelling in the archipelago sea.*

Field sampling and data sources

The biological data used as response variables in the models are sourced from the 2005 and 2006 field campaigns of the Natural Heritage Service of Finland and Alleco Oy. The Natural Heritage Service's data are underwater video surveys and Alleco's data are dive transects. In both datasets observations of the species visible to the eye have been recorded as percent cover from the field of view.

Predictor layers

Several predictor layers were available for the analyses:

I. A depth model based on nautical chart data and elevation data re-sampled to 5m (from 25m) resolution using nearest neighbour interpolation, as well as slope and aspect derived from this dataset.

II. A 5m raster of shoreline density within a 500m radius – as a proxy for enclosedness of the archipelago.

III. Wave exposure re-sampled from 25m raster to 5m raster using nearest neighbour interpolation.

IV. Euclidian distance measures to sandy shores (25m raster), rocky shores (5m raster) as well as submerged and emergent rocks in nautical charts (25m raster).

V. Turbidity derived from satellite images interpolated to a 5m raster.

Response variables

As the presence records for other species than mussels (*Mytilus trossulus*) were very limited in the dataset, all algae and all angiosperms were pooled together into two group variables. The main species of algae forming the algae group were *Fucus vesiculosus*, *Chorda filum*, *Pilayella littoralis, Ceramium* spp. and various filamentous algae. The main angiosperms in the area were *Potamogeton* spp. and *Zannichellia*. Although both methods used to gather the biological data use observations of percent cover in the field of view, the response variables were converted to presence / absence to minimise the impact of different methods.

Three response variables were consequently used:

- presence/absence of *Mytilus trossulus*
- presence/absence of any algae
- presence/bbsence of any angiosperms

Model selection (methods, algorithms, software, and routines)

Generalised additive models (GAMs) were calculated for the response variables. All models and predictions were produced using the open source statistical software R (available from http:\\www.r-project.org). The statistical package used for GAMs in R was 'mgcv' (Wood, 2006), which uses a penalized regression spline approach. The models were built using a logit link function and a quasibinomial family. Smooth terms were constructed using thin plate regression splines with the degree of smoothness of model terms initially estimated as part of fitting using Generalized Cross Validation (GCV). In the final models, however, degrees of freedom for some variables were set manually to avoid overfitting. The final models are shortly described in table 1.

Table 1: Description of final models including predictor variables, their degrees of freedom (d.f.), the deviance explained by the model, the models Generalized Cross Validation (GCV) score and the area under the receiving operator curve (AUC).

	Predictor variables	d.f.	Deviance explained	GCV-score	AUC
Mytilus trossulus	Depth Distance to rocks Exposure Distance to sandy shore	2 1 2.5 1	51.1%	0.69	0.68
Algae	Depth Density of shoreline Aspect Distance to sandy shore	1 2 4 2	37.7%	0.91	**0.84***
Vascular plants	Depth Distance to sandy shore Exposure	1 3 1	27.3%	1.04	**0.94***

* AUC significantly exceeds the critical AUC level 0.7 (Hosmer and Leveshow 2000).

6.9.3 Results

The initial check of predictor data showed that exposure, density of shoreline and turbidity were highly correlated. As exposure and the density of shoreline worked better in the models, turbidity was left out at an early stage. One contributing factor to this was that the satellite image based turbidity is tied to the moment of acquisition, where as exposure and density of shoreline are more permanent measures. Slope did not really reduce the deviance in any of the models, so was also left outside the model development.

Depth became the most important factor in all models. Other important factors turned out to be the distance to sandy shores and either exposure or the density of shoreline. Only one of the latter was used in a model. With *Mytilus* the distance to submerged and emergent rocks also became significant, whereas aspect explained a large amount of deviance in the algae model.

The *Mytilus* model got the highest score for explaining variance in the response variable as well as the lowest Generalized Cross Validation (GCV) score (table 1), whilst the deviance explained by the angiosperm model was fairly low. However, validation using the area under the receiver operator curve (AUC) shows that the models with best predictive capability are the algae and angiosperm models, falling into the category 'excellent and 'outstanding', respectively. The *Mytilus* model on the other hand does not significantly exceed the critcal AUC level set at 0.7(Hosmer and Leveshow 2000).

*Spatial predictions of Species 1 ***

Text text text.

Text text text.

Results of modeling. Text text text.

Prediction maps. Text text text.

Spatial prediction of Species 2 **

Text text text.

Text text text.

Results of modeling. Text text text.

Abundance prediction maps. Text text text.

6.9.4 Discussion

Quality of data and methods

Text text text.

Quality and validation of models

Text text text.

Validity of identified habitats (confidence assessment)

Text text text.

Application of habitat maps in management

Text text text.

Text text text.

1. Text
2. Text
3. Text

Text text text.

Text text text.

6.9.5 Conclusion and perspectives

Text text text.

Text text text.

- Text
- Text

- Text

6.9.6 Acknowledgement

Text text text.

6.10 PA3. Spatial prediction of fish habitats in the Swedish Archipelago

Authors: U. Bergström, A. Sandström, and G. Sundblad.

6.10.1 Introduction

Young fishes are often dependent on certain habitats for their survival, and protecting these habitats may be crucial for maintaining strong adult stocks. An important step in providing sufficient protection for such essential fish habitats is reliable large-scale habitat mapping. One promising approach is that of spatial, predictive modelling, where statistical models relating species occurrence to environmental variables are coupled to geographic information systems (GIS). This approach was used for mapping spawning and nursery areas of a number of common coastal fishes in the large, complex, Swedish-Finnish archipelago area in the Northern Baltic Sea.

The species. Within BALANCE WP2, we have concentrated on developing models for four of the most ecologically as well as economically important fish species in the coastal ecosystem of the Baltic Sea. These are the freshwater fish species Eurasian perch (*Perca fluviatilis*), northern pike (*Esox lucius*), pike-perch (*Sander lucioperca*) and roach (*Rutilus rutilus*). Pike-perch and northern pike can be considered as obligate piscivores, perch as a facultative piscivore, and roach as a generalist and omnivore and an important prey fish for the other species. They all depend to a varying extent on shallow near-shore areas during their early life stages, mainly since such areas are heated more rapidly early in spring and high water temperature is needed for juvenile development and survival (Karås & Hudd 1993, Sandström et al. 1997). All four species have been subject to numerous studies, and their biology and ecology is well known.

Aims. The aim of this study was to produce maps describing the distribution of nursery habitats for perch, pike, pike-perch and roach, and of spawning habitats for perch in the BALANCE pilot area 3, in the northern Baltic proper. Generalized additive models (GAM) were fitted to describe the relationship between fish occurrence and habitat variables. Maps describing the distribution of recruitment habitats of the studied species are currently lacking, why protection of these essential fish habitats through marine spatial planning is difficult. Besides being used directly in marine spatial planning by national and regional authorities in Sweden, Åland and Finland, these maps will be used for analyses of the coherence of the Natura 2000 network (WP3) and for development of GIS tools for marine spatial planning (WP4).

6.10.2 Material and Methods

The study area. The BALANCE pilot area 3 is located in the vast archipelago region that stretches from the counties of Södermanland, Stockholm, and Uppsala in Sweden, over via Åland and the Finnish Archipelago Sea (fig 1).

Field sampling and data sources. Sampling of juvenile fishes was conducted in late July-August 2005-2006. Juvenile fish were monitored by point abundance sampling with small detonations that stun small fish within an area of ca. 60 m^2 (evaluated in Snickars et al. 2007). This method allows quantitative sampling of fish (15-150 mm) with well-developed swim bladders in all shallow habitats, including dense vegetation. All stunned individuals (floating and sinking) were collected via snorkelling for later counting of the number of individuals, determination of species and length measurements. The distribution of sampling sites was stratified along wave exposure and archipelago zonation gradients, in order to cover the whole ranges of distribution of the studied species (fig. 1).

Surveys of perch egg strands were conducted three times during a period from late April to mid June, with intervals of 14-20 days in 2003. The survey was conducted by snorkelling along parallel transect lines (length 20-480 m, 4-8 lines per site) drawn perpendicular to the length axis of each site from one shore to the opposite shore until the entire site was covered. All visible egg strands within one metre on both sides of the transect lines were registered. Totally 22 000 m^2 were surveyed for eggs covering 22 studied sites. The majority of the sites were shallow inlets and selected from a previous large-scale survey of inlets. Selection criteria included a minimum level of anthropogenic disturbance and a considerate but not extreme variation in geomorphometry and wave-exposure. Sites were spread over a relatively large geographical area in order to study general trends within the Baltic Sea (fig. 1)

In both the surveys of juvenile fish and perch egg strands, representative information on environmental variables such as depth, temperature, turbidity, substrate and vegetation were collected for all sampling points. The surveys were conducted by the Swedish Board of Fisheries in collaboration with the Foundation for Uppland, the Finnish Game and Fisheries Research Institute, Metsähallitus in Finland, and Åbo Akademi University.

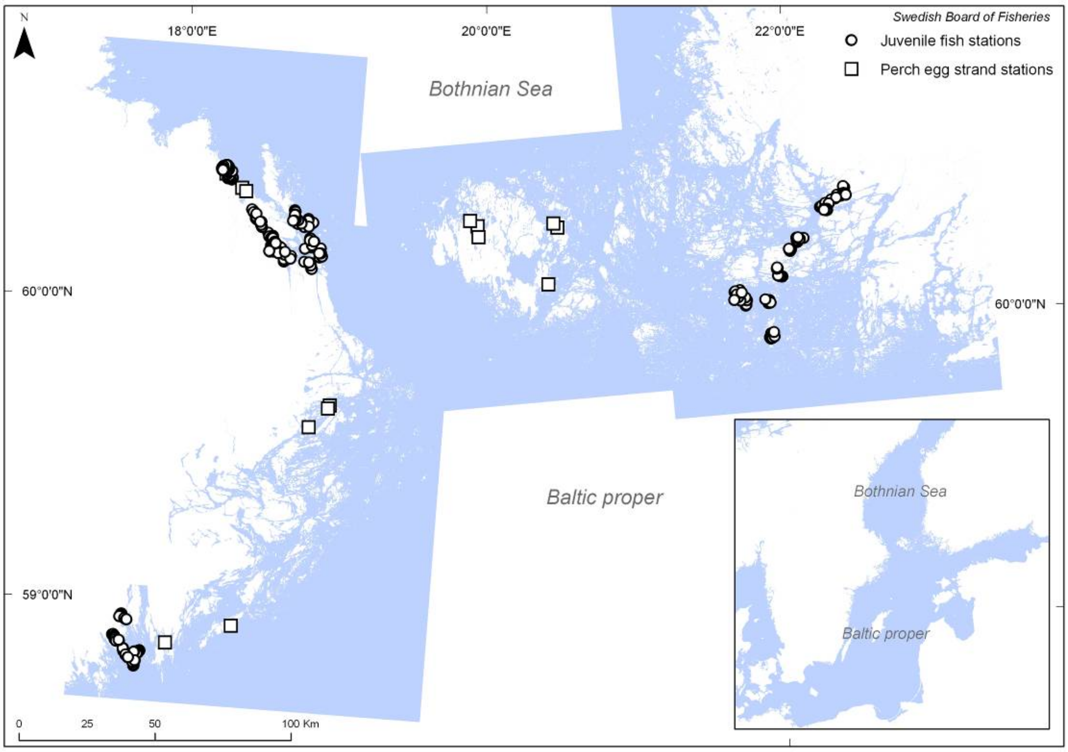


Figure 1. Sampling stations within pilot area 3.

Predictor layers. Few environmental predictor variables were available as continuous maps for the whole BALANCE pilot area 3. Only wave exposure (Isæus 2004) and depth from sea charts were considered to have a spatial coverage, resolution, and accuracy suitable for the fine-scale fish habitat modelling. In some areas the depth information is very coarse, mainly due to military restrictions affecting the access to the existing data, which in these areas substantially limits the usefulness of this variable for these fine-scale modelling purposes.

Wave exposure and depth are important predictors for fish distribution, but further variables are needed to obtain high quality habitat maps. One important predictor variable for fish distribution that was not available at a sufficient resolution was water clarity. To cover this gap, a GIS model of Secchi depth was developed. The index was based on distance from the base line and wave exposure. Evaluation of the index was made against 293 in-situ measurements of Secchi depth (range 0.3-10 m), from different years but the same season. There was a significant positive correlation between the index and field measurements of Secchi depth, but the index only explained a minor part of the variation in the level of Secchi depth (r^2=0.2911). The input data was from different years, thus the model cannot be expected to be very accurate, as water turbidity is highly variable even at short time scales. The proxy, however, still captured the gradients in water clarity that are found at small spatial scales from small sheltered bays to open areas, which are important for the distribution of juvenile fish, very well, and was therefore considered to be valuable as a predictor layer.

All layers were in ESRI raster format with 25 meter cell size and in UTM34N projection. Depth was limited to maximum 6 metres depth, as the data from the fish surveys was available down to this depth. The Wave exposure index used was a log10-transformation of data produced using WaveImpact (Isæus 2004). The proxy on water clarity was calculated using water distance from the base line (connecting the outmost islands and thereby defining the archipelago zone) and the wave exposure index, using the equation: water clarity proxy = logwaveexposure/logdistance^0.5.

Response variables. Separate models were constructed for young-of-the-year of perch, pike, pike-perch and roach, as well as for spawning of perch. For each model, presence/absence of the species was used as response variable.

Model selection (methods, algorithms, software, and routines). Using Hawth's tool in ArcGIS, data from each predictor layer was extracted for the fish sampling positions. GAMs were used to model the probability of occurrence for the different species/life stages. Modelling was conducted in S-PLUS using the GRASP work package. All predictor variables were forced into binomial models with 3 degrees of freedom to fit the spline function. No weights on prevalence were applied since this has been shown to overestimate the probability of presence when making spatial predictions (Maggini et al. 2006).

The models were used for producing spatial predictions in GIS using script provided from the GRASP work package. The extent of the predictions was the whole PA3, limited to areas shallower than 6 m depth. All predictor raster layers were in 25 m resolution, as was the resulting maps, showing probability of presence. ESRI Spatial Analyst was used to reclassify the continuous probability of presence predictions into dichotomized maps of suitable and unsuitable habitat for each response variable. The threshold for habitat suitability was determined using the true-skill statistic TSS (Allouche et al. 2006), where the sum of specificity and sensitivity is maximised. There are many methods for determining thresholds of presence, and TSS has the advantage of not being affected by the prevalence or the size of the validation set (Allouche et al. 2006). Thus, two different sets of grids were made. The first showing continuous probability of presence for each species/life stage, and the second set showing categorized habitat in suitable/unsuitable habitat.

Additional explanatory Y-O-Y models were built using the same data set with the addition of total vegetation cover (%) in order to show differences in modelling potential with alternative predictor variables.

Model performance was evaluated by an analysis of deviance, which is equivalent to variance analysis in general linear models. The overall test statistic is called D2 and is a measure of goodness-of-fit for the overall model. The increased flexibility of these models can however lead to overdispersed errors. Over dispersion is characterised by largely inflated residual deviance, which was examined in all models. A general recommendation is that one should rely more on empirical evaluation rather than D2, due to a tendency of overfitting the calibration data (Guisan et al. 1999). Model evaluation was therefore also based on receiver-operating-characteristic (ROC) plots. ROC-plots are obtained by plotting all sensitivity values on the Y axis and (1-specificity) on the X axis for all available thresholds on the X axis (DeLeo 1993, Fielding & Bell 1997). ROC-plots give an area-under-curve (AUC) value that range between 0.5 and 1. An AUC-value of 1 indicates no overlap between the two group distributions, i.e. true positives and false positives. A value of 0.75 shows that 75% of the time a random selection from the positive group will have a score greater than a random selection from the negative group (DeLeo 1993, Fielding & Bell 1997). ROC (AUC) has been recommended as a measure of accuracy since it is insensitive to the response variables prevalence (McPherson et al. 2004). To aid model evaluation in comparative studies it has been recommended that ROC and sampling prevalence should be reported (McPherson et al. 2004).

The built-in validation procedure with ROC-plots in GRASP produces both a ROC-plot for the entire data set as well as a cross-validation with subsets of the data to compare predicted versus observed values. All models were validated using 5 groups in the cross-validation.

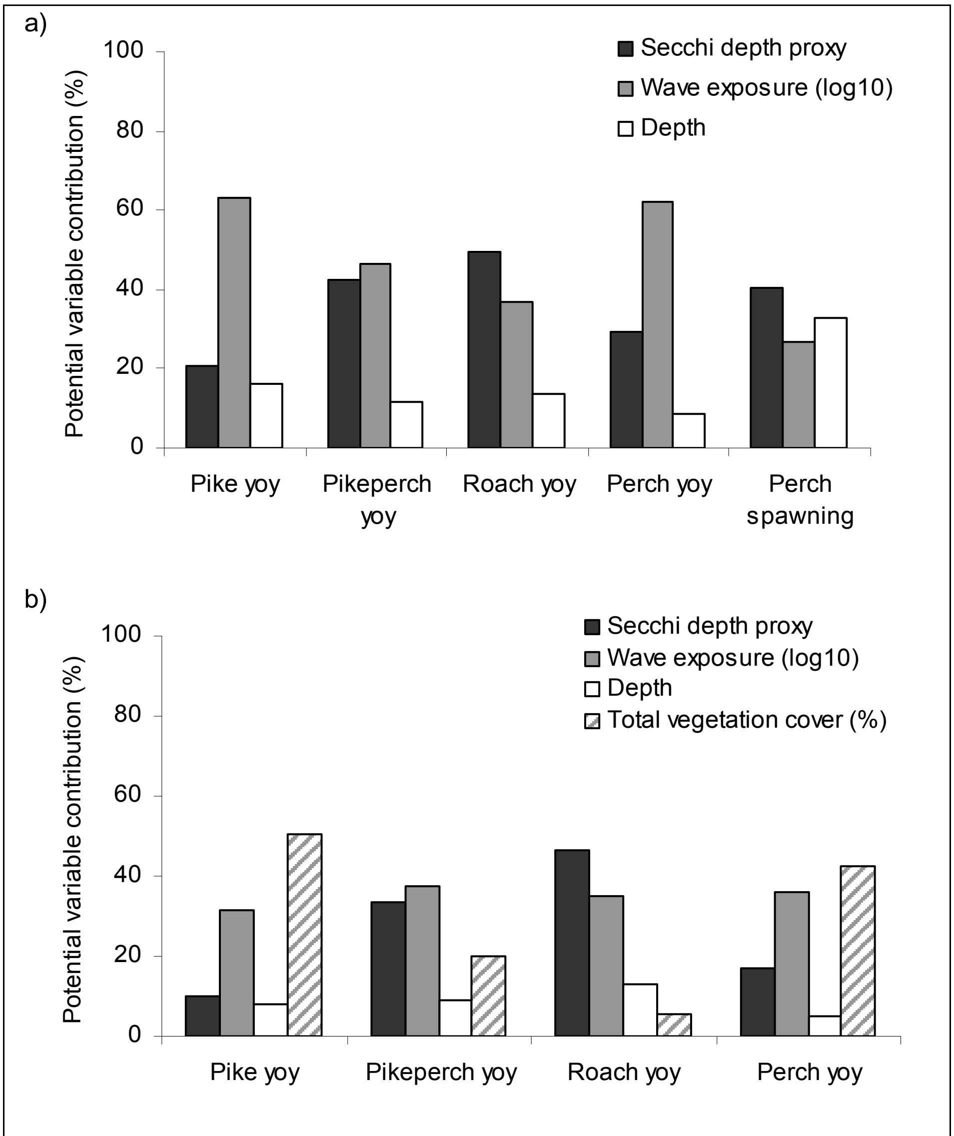


Figure 2a-b. Potential contribution of each predictor variable alone, a) for the environmental factors used in the map predictions and b) when adding an additional environmental parameter, total vegetation coverage, to the statistical models.

6.10.3 Results

The model for pike-perch was the most accurate (fig. 4, ROC = 0.90), and the models for perch spawning, pike Y-O-Y and roach Y-O-Y also performed reasonably well (fig. 5, 6, and 8, ROC = 0.75-0.81). The model for perch Y-O-Y proved less accurate (fig. 7, ROC = 0.66). This model also showed a tendency for overdispersion (Df = 296 total, Residual deviance = 378.9644), indicating that there is some unexplained spatial heterogeneity in the data. All other models were well within limits.

The potential contribution from each predictor variable, calculated by creating new models with only one predictor, showed that wave exposure and Secchi depth contributed most to the performance of the models while depth did not contribute as much to model strength (fig. 2a). Adding vegetation coverage would probably improve the spatial predictions substantially, especially when modelling the distribution of perch and pike Y-O-Y (fig. 2b). However, as vegetation was not available as a continuous map, no fish habitat maps using vegetation coverage as a predictor could be produced.

The partial response curves of each GAM illustrate how each explanatory variable affects the distributions of the species and life stages modelled. For pike Y-O-Y, there was a positive effect of increased water clarity, a negative effect of increased wave exposure and a slight negative effect of depths >3 m (fig. 3a). For pike-perch Y-O-Y, there was a negative effect from increased water clarity and wave exposure, while depth had little effect (fig. 3b). In the roach Y-O-Y model, water clarity had the highest contribution, with a positive effect of low clarity. There was a negative effect of increased wave exposure and a tendency of preferred depth around 2 m depth. For the perch Y-O-Y model, there was a clear positive effect of increased water clarity and a negative effect of increased wave exposure, while the effect of depth was not as evident. In the perch spawning model, water clarity had a positive effect, and wave exposure a negative effect. Depth had a negative impact from an optimum depth around 1 m.

Validation. Model evaluation statistics for the probability of presence predictions, i.e. D^2, AUC for both cvROC and ROC, as well as prevalence in tab. 1. These statistics show that both accuracy and level of generalisation was highest for the pike-perch model, intermediate for the perch spawning and roach models and lowest for the pike and perch Y-O-Y models.

The evaluation of the categorised presence-absence maps, i.e. the TSS scoring based on an error matrix, gives a similar picture. The highest scores were found for the pike-perch and roach Y-O-Y and perch spawning habitat maps, while pike and perch Y-O-Y maps were less accurate. The misclassification rates were 30-38 % in all models.

Table 1. Model evaluation statistics for the five recruitment habitat models.

Species / Statistic	D^2	cvROC	ROC	Prevalence indata	TSS, categorised maps
Pike Y-O-Y	0.11	0.64	0.75	0.08	28.6
Pike-perch Y-O-Y	0.34	0.85	0.90	0.09	34.6
Roach Y-O-Y	0.19	0.74	0.81	0.13	39.6
Perch Y-O-Y	0.06	0.61	0.66	0.43	23.9
Perch spawning	0.12	0.77	0.78	0.08	35.0

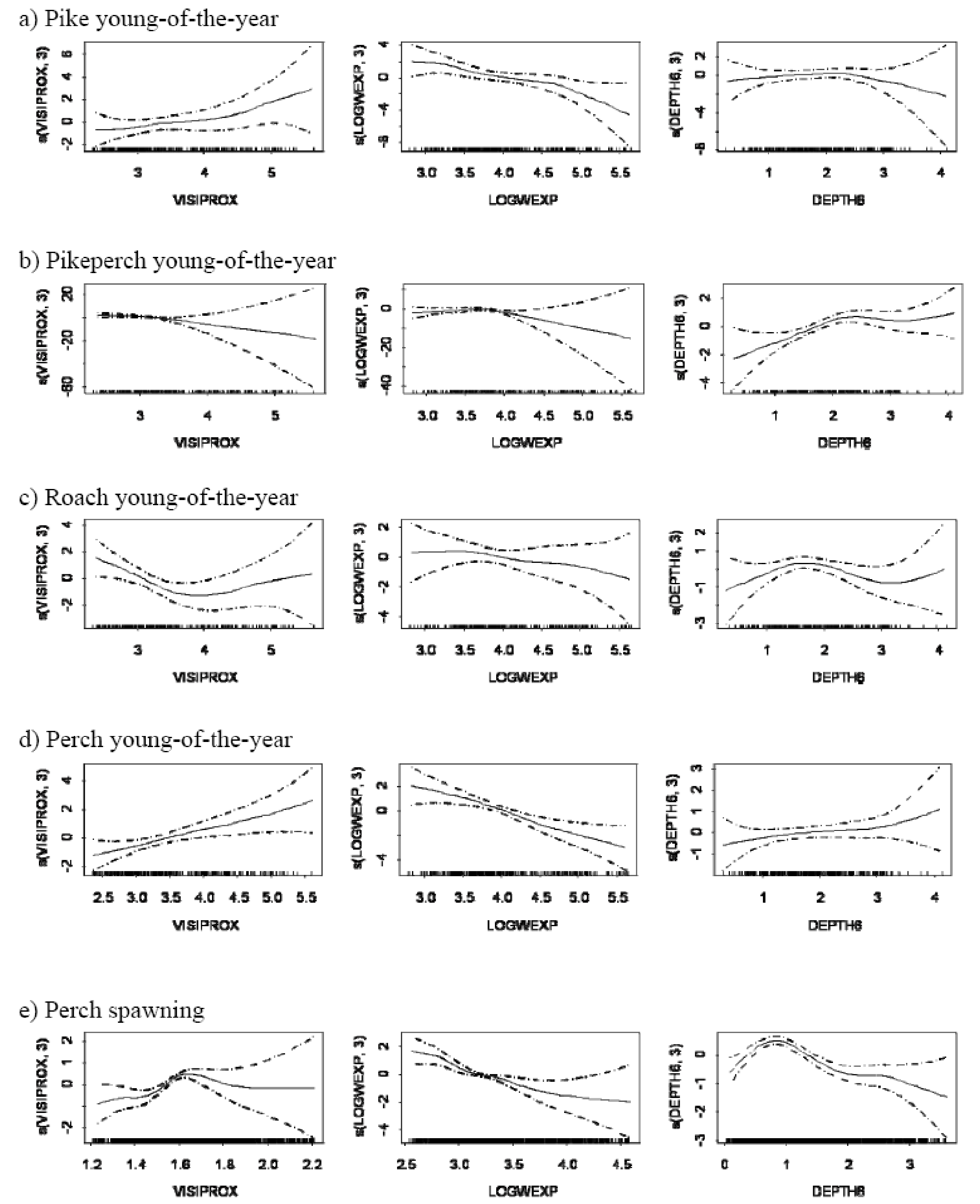


Figure 3 a-e. Partial response curves for the GAMs of the species and life stages modelled. The y-axis represents the response variable in the linear predictor scale. Visiprox denotes the Secchi depth proxy, logwexp the logarithm of the wave exposure index, and depth6 water depth. Dotted lines indicate twice point standard errors and the dots on the x-axis represent the samples along each predictor variable gradient.

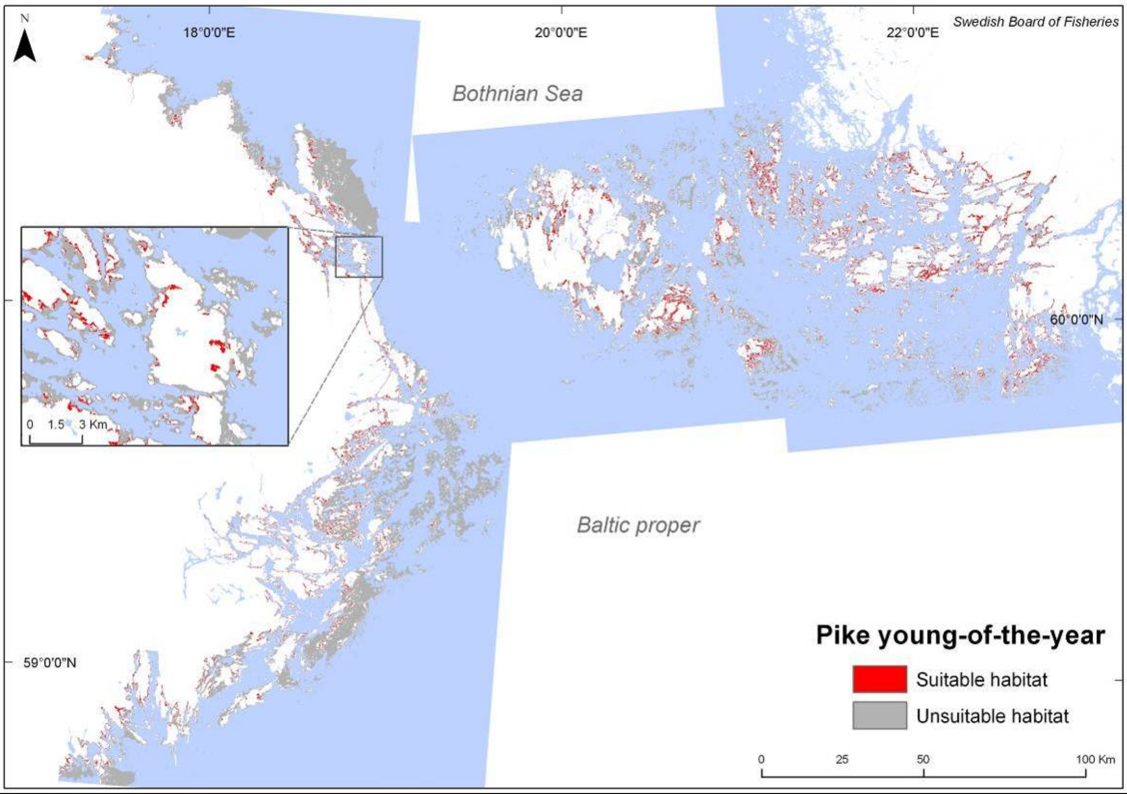


Figure 5. Pike young-of-the-year habitats.

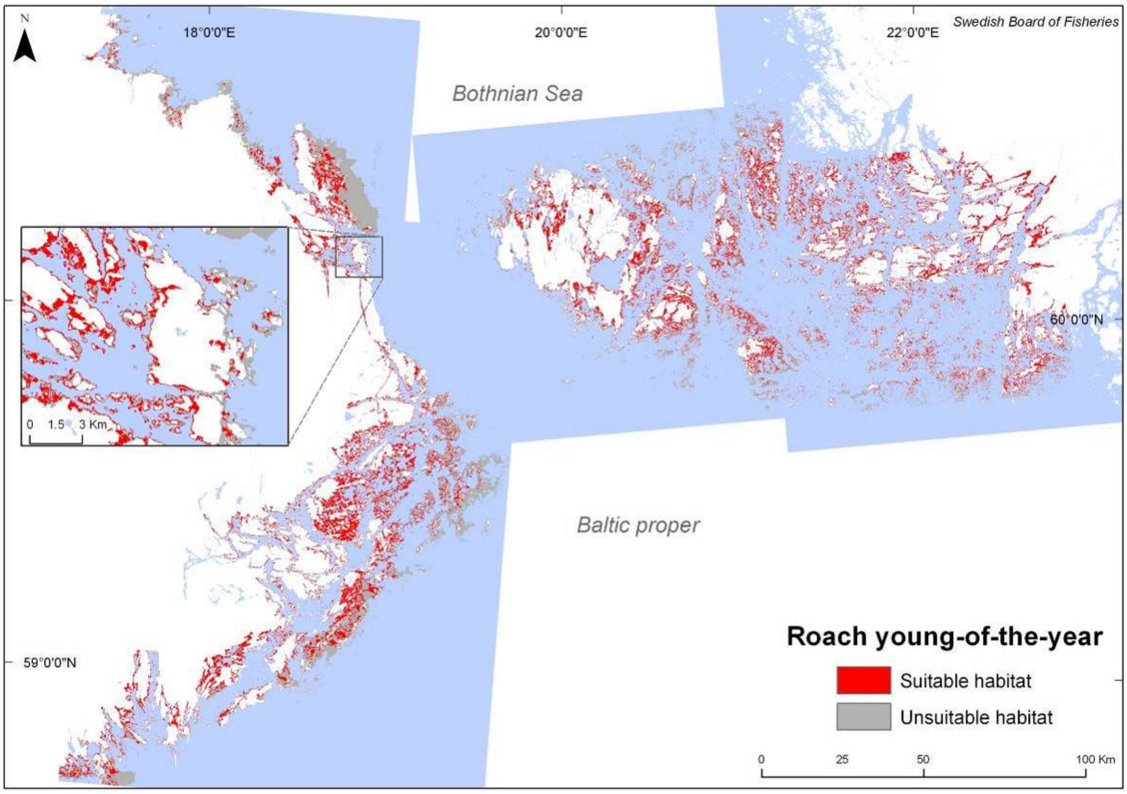


Figure 6. Roach young-of-the-year habitats.

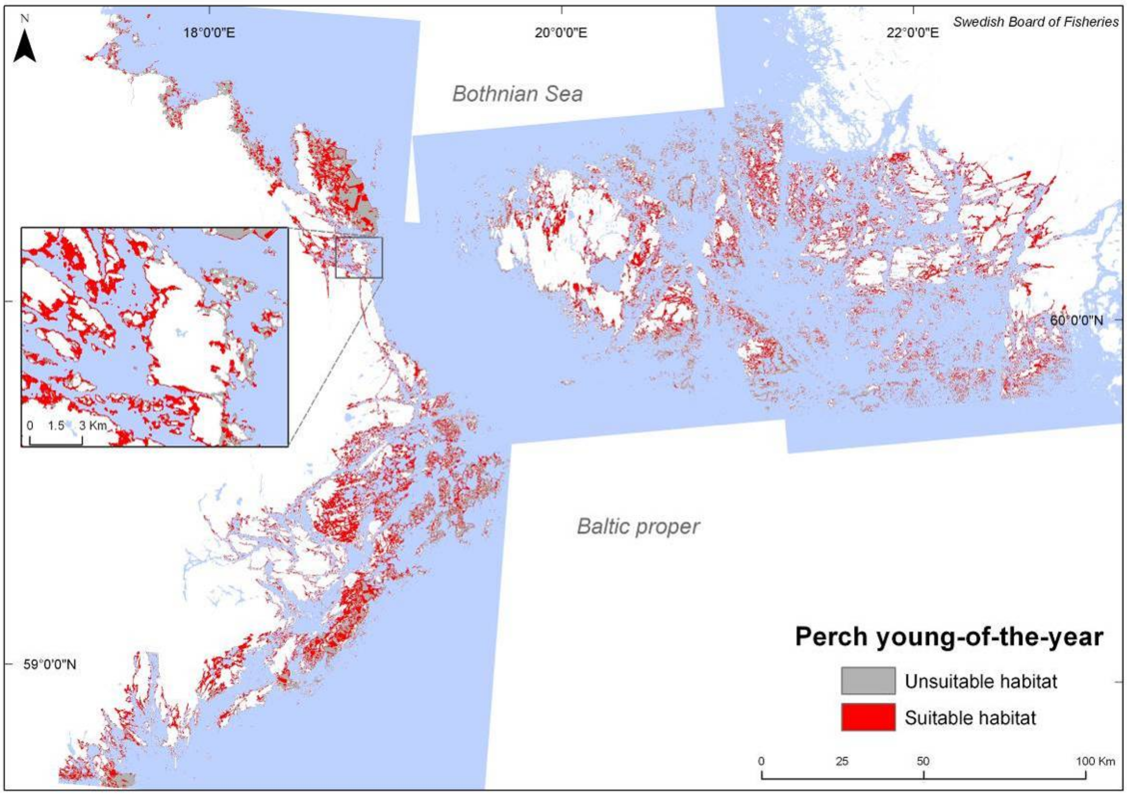


Figure 7. Perch young-of-the-year habitats.

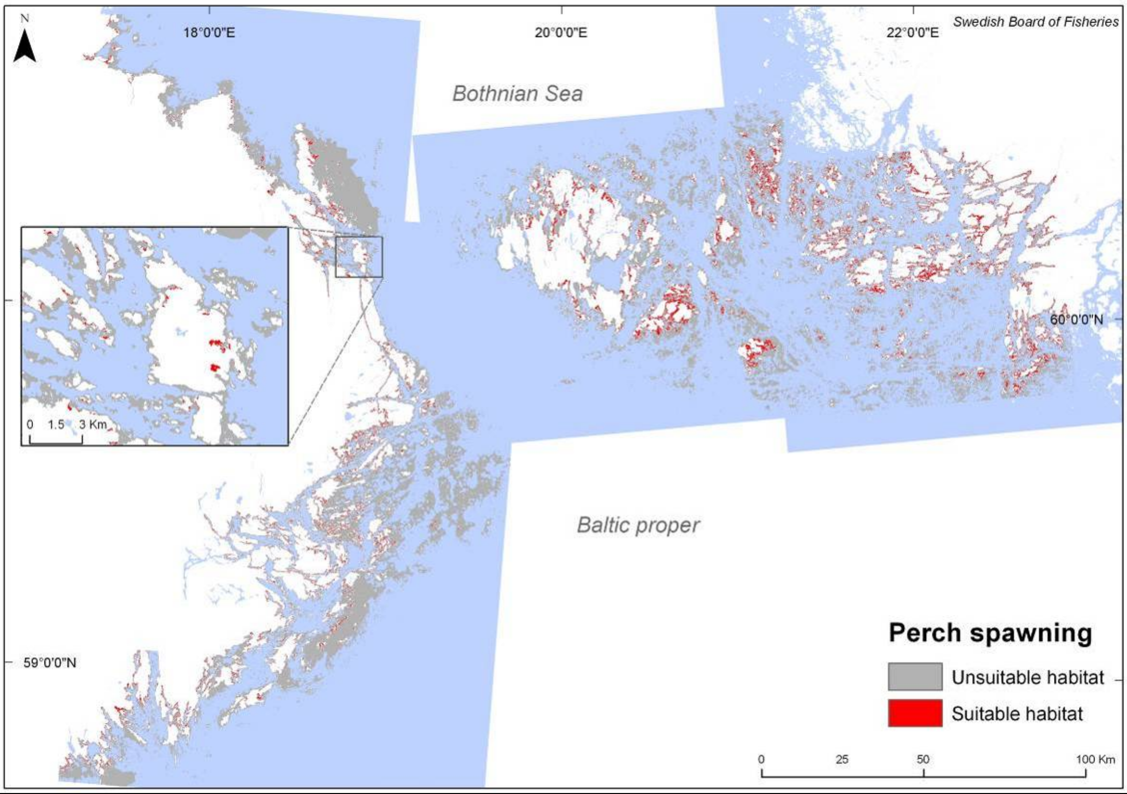


Figure 8. Perch spawning habitats.

6.10.4 Discussion

We have used three easily obtainable GIS-layers to produce large scale maps of predicted habitat of four species and two life stages in PA 3. For all the investigated species/life stages, except for perch Y-O-Y, the resulting statistical models were relatively strong, especially considering the high resolution of the predictions in relation to the large extent of the study. Our results show that the predictor variables used are important components when determining the habitats of these species, and that GIS-modelling could develop into an indispensable tool in large-scale mapping of essential fish habitats.

Two of the predictor variables, wave exposure and the visibility proxy, are completely GIS-derived and can be considered as indirect variables (Austin 2002). Still, in all Y-O-Y-models they contribute the most in explaining species distribution of the variables tested. Depth can also mainly be characterised as an indirect variable encapsulating other more direct variables such as light (attenuated with increased depth), temperature, water movement (decreasing with depth) and vegetation. Depth has the least influence on the models, not because it is unimportant for determining species distributions, but rather because no predictions were made deeper than 6 m. The low impact of depth in the models only shows that there are no large differences in habitat quality between 0 and 6 m depth – deeper areas are certainly less suitable to these young life stages, but the field data covered areas only down to 6 m.

Information on water turbidity for the archipelago area between Sweden and Finland is also available only at a resolution too coarse for the kind of habitat modelling undertaken in this study. The GIS-derived visibility proxy was reasonably successful in detecting small-scale gradients in Secchi depth, and proved to be an important predictor variable in all models. For future modelling work, an alternative approach to attaining large-scale maps of turbidity could be to use satellite imagery. A separate study in PA 3 showed that turbidity can be accurately interpreted from SPOT 5 images at a resolution of only 10 m (Bergström et al. 2007).

Comparing explanatory models based on the three variables used to produce the predictions with additional data on total vegetation coverage, a more direct predictor variable, showed that both Y-O-Y pike and perch distributions also are strongly governed by vegetation coverage (fig. 2b). Vegetation adds habitat complexity and can be important both as a refuge against predators as well as a host to many prey animals (Persson & Eklöv 1995). The selection of spawning sites by perch is also known to depend largely on vegetation type (Thorpe 1977, Treasurer 1983). Thus, including vegetation coverage as a predictor variable would most likely increase the predictive power of the models. Producing high-resolution, large scale GIS-layers of vegetation coverage is therefore an important step towards increasing the precision of many fish habitat models.

Statistical modelling using GAM and the GRASP work package proved to be a flexible and accessible technique for describing species-habitat relationships. A potential disadvantage of using GAM is that the model do, due to their additive structure, not allow for taking interactions between predictor variables into account. There are ways of overcoming this limitation, which should be further explored. For example, interaction terms may be added manually to a model, by simply creating interaction terms e.g. by multiplying two predictors and adding the result as a separate predictor. Another approach for adding interaction terms can be found in Maggini et al. (2006). A regression tree can be fitted on the residuals of a first model, where after the branches/leaves of the resulting regression tree is used to classify each sample. Then a new model is built using the resulting classification as an additional predictor. This procedure is a way of incorporating interactions between all variables in a single term.

The map predictions are based on life-stage specific relationships from a limited number of study sites and a limited set of environmental variables, which in a GIS have been recalculated to show probability of occurrence. Conceptually, the maps therefore show the potential distribution of the modelled life-stages based on the environmental maps, rather than the true distribution. For juvenile pike-perch, the habitat map appears to overestimate the distribution of the species. This pattern may be an effect of limitations in earlier life stages, for example in access to suitable spawning habitats. Thus, areas lacking spawning sites will naturally also lack Y-O-Y fishes even though suitable habitats for juveniles are abundant. An interesting application of this kind of modelling work may thus be to identify habitat bottlenecks, as well as regions where habitat degradation has had negative effects on fish stocks.

6.10.5 Perspectives

There is a high demand for detailed maps of essential fish habitats for a range of physical planning activities. The habitat maps presented in this report are already used by several regional authorities, for example in fisheries restoration and management plans and in the design and zonation of forthcoming MPAs.

So far, few studies exist that use statistical modelling coupled with GIS for large-scale mapping of essential fish habitats. Based on our experiences so far we believe that this approach will become widely used in the future, and that these habitat maps will probably become a central constituent in marine spatial planning. Both techniques and data basis are in a phase of rapid development, and both the spatial coverage and the accuracy of the maps can therefore be expected to increase steadily.

The success of fish habitat modelling initiatives will, however, not only depend on the understanding of the dynamics of fish populations and their reaction to environmental variables, but also on the accuracy of the maps of the environmental variables that the predictions are based upon. Currently, a lack of high-resolution maps of for example bathymetry, surface sediments, hydrography, and in the case of young fishes, vegetation coverage, is limiting the production of accurate habitat maps. For bathymetry, this deficiency may be alleviated for example by opening access to classified maps, and by easing restrictions on collection and usage of bathymetric data. For other environmental variables, and for bathymetry in some areas, additional high-resolution mapping is needed. Development of new techniques, such as remote sensing for identification of coastal habitat characteristics (Bergström et al. 2007), as well as GIS-modelling techniques similar to those used within this BALANCE report, may provide efficient tools for producing high-resolution maps at reasonable costs.

6.11 PA 4. Modelling of species habitats in Estonian waters

Authors: Jonne Kotta, Kristjan Herkül, Helen Orav-Kotta, Mart Simm, Georg Martin.

6.11.1 Pilot Area 4

The Gulf of Riga is a relatively shallow and isolated water-body. On eastern and southern sides it is surrounded by the Estonian and Latvian mainland and on the northern side by Saaremaa and Muhu islands. The Gulf of Riga is connected to the Baltic Proper via the Irbe Strait and to the Väinameri Archipelago Sea by the Suur Strait. Annual river inflow ranges between 18 and 56 km^3 (an average 32 km^3) while the volume of the gulf is 424 km^3. Residence time of the water masses is 2–4 years (HELCOM 1996). The Gulf of Riga receives fresh water from a huge drainage area (134,000 km^2) and the majority of it enters the southern part of the basin (Andrushaitis et al. 1995). In general, the bottom relief of the area is quite flat with gentle slopes towards deeps. The northern part of the gulf is characterized by a wide coastal zone with diverse bottom topography and extensive reaches of boulders. The southern part of the Gulf of Riga is more exposed, steep and soft substrate prevails. In the deeper parts of the gulf silty sediments prevail.

The average salinity of the Gulf of Riga varies from 0.5–2 PSU in surface layers in its southern and north eastern areas to 7 PSU at the Irbe Strait. In most parts of the gulf the salinity is 5–6.5 PSU. During the ice-free season the salinity is higher in the bottom layers and lower in the surface layer. However, due to its shallowness the Gulf of Riga lacks a permanent halocline (Berzinsh 1995, Raudsepp 2001).

As the Gulf of Riga is a shallow water basin the changes in air temperature have a direct influence on the dynamics of both surface and deep water. In a “typical” year the water is cold and no clear thermocline occurs till May. Later the surface water temperature raises to about 17–20 ºC and a thermocline builds up. The water temperature below 30 m remains relatively stable at 3 ºC. The thermocline reaches a depth of 25 m in August and disintegrates in September–October due to intensive wind mixing. In the course of autumn storms the surface water cools down and the deep water temperature rises to 5–10 ºC (Raudsepp 2001).

The duration of ice season has a large interannual variability. The range of variation of the number of ice days is several months. The average number of ice days varies spatially from 80 days in the Irbe Strait and open Gulf of Riga to 150 days in Pärnu Bay. The number of ice days decreased at a rate of 5–7 days in the last century (Jevrejeva 2000).

The sea level of the Gulf of Riga was modelled based on realistic meteorological forcing and historical data. The extremely low levels (-1.25 m below the mean sea level) do not generally occur locally whereas the high levels (up to 2.75 m above the mean as measured in Pärnu Bay in 2005) are short term and local. The shallow and narrow bays exposed to the direction of the strongest possible storm winds (SW and W) are prerequisites of these high values (Suursaar et al. 2003ab). The most effective forcing function for the system is the wind stress above the straits, having a considerable role in motions with time scale between 1 yr and 1–2 d. In winter the water flows and exchange depend strongly on the existence of ice-cover in the Gulf of Riga. The seasonality of the water exchange process is governed by seasonal changes in the large-scale atmospheric circulation scheme above the North Atlantic (Otsmann et al. 1997, Otsmann et al. 2001). The Gulf of Riga is connected with the Baltic Proper mainly through Irbe Sound which covers about 80–85% of the total exchange. The average measured velocities are at 12 cm s^{-1} and the maximum values up to 72 cm s^{-1}, respectively. The water flow in the Suur Strait is uniform but temporal variability of the currents is quite complex. The average measured velocities are at 19 cm s^{-1} and the maximum values up to 1 m s^{-1}, respectively. The total volume of the water running through the strait in both directions is about 100 km^3 yr^{-1} (Suursaar et al. 1995). The average flow velocities in Pärnu Bay are estimated at 4–11 cm s^{-1} and the maximum value at 90 cm s^{-1} (Suursaar et al. 2002).

The oxygen regime of the Gulf of Riga is relatively good due to its shallowness and strong vertical mixing. In most areas oxygen concentrations are higher than 5 ml l^{-1}. From April to the middle of October seasonal stratification may restrict vertical water exchange, promoting thus oxygen depletion and storage of nutrients in the bottom water until the water column is remixed in autumn. Concentrations below 2 ml l^{-1} have occasionally been found in the deepest part of the gulf (> 45 m). Since the middle of 1960s a statistically significant decreasing trend in the concentration of oxygen was observed in the study area (Berzinsh 1995, Yurkovskis 2004).

The gulf is on average twice as eutrophicated as the Baltic Proper and the outflow of nutrients through the straits is bigger than the inflow (Mägi and Lips 1998). The seasonal and vertical regime of nutrients in the gulf differs somewhat from that in the open sea. Nutrient concentrations are affected by the occasional inflows of saline and nutrient rich deep water from Gotland Basin via the Irbe Strait and year-to-year variations in river inflows. The rivers play a crucial role in the total input of nutrients and exceed the combined contribution from atmospheric deposition, point emission from cities and industries along the coast, and nitrogen fixation by marine organisms. Higher concentrations of nutrients are found in the southern and north-eastern parts of the gulf, i.e. adjacent to the mouths of the Daugava, Lielupe, Gauja and Pärnu rivers. Because of its shallowness, the Gulf of Riga has no clear chemocline. Although higher concentrations of nutrients are often observed in deeper water in May–October, this pool is accessible for the surface layers through occasional mixing events. Particularly strong vertical mixing processes in autumn and winter result in high nutrient content of upper layer in January and February. Both dissolved inorganic nitrogen and phosphate pools of the upper mixed layer are exhausted by mid-May, except at the river mouths where the nutrient concentrations decline only in July. In summer and early autumn the concentrations remain very low. Since November a gradual increase takes place due to higher intensity of vertical mixing. Both nitrate and phosphate contents increased during 1974–1988 and decreased in recent years. However, the decreasing trend for total P was not so clear. The obvious depletion of the silicate-Si pool in 1985–1991 reversed after 1995 (Suursaar 1995, Astok et al. 1999, Stålnacke et al. 1999, Põder at al. 2003, Yurkovskis 2004). The sedimentation rate is roughly 2 mm yr^{-1} and carbon accumulation 5 g C m^{-2}, respectively (Danielsson et al. 1998).

Pärnu Bay is a shallow semi-enclosed water basin in the NE Gulf of Riga. The surface area of the bay is about 700 km^2 and its volume is 2 km^3. The maximum depth increases gradually from 7.5 m in its inner part (NE of the Liu–Tahku line) to 23 m in the SW part. The hydrological conditions of the bay are formed under the complex influence of meteorological processes, the river discharge (Pärnu River, freshwater inflow 2 km^3 annually), and the water exchange with the open part of the Gulf of Riga. The currents are generally weak in the area and are mainly wind induced.

The bay is suffering from a heavy anthropogenic eutrophication. The town of Pärnu with its 70,000 inhabitants and the Pärnu River are the major sources of pollution in the bay. The contents of total N, total P, and silicate increased on average two times in the seawater and the primary production of phytoplankton increased substantially in the 1970s and 1980s (Ojaveer 1995, Tenson 1995). Since 1990 the wastewater of the town of Pärnu has been mechanically and biologically treated. However, the Pärnu River, which is responsible for about 10% of the total riverine runoff to the Gulf of Riga and annually brings about 40–50 t of total P and more than 4000 t of total N into Pärnu Bay, is still a significant source of nutrients (Suursaar 1995).

6.11.2 Introduction

The brown alga *Fucus vesiculosus* is the dominant macroalgal species in the Baltic Sea comprising up to 43% of the benthic plant biomass (Kautsky & Kautsky 1995). In recent years the biomass of the species has notably diminished at many localities. This decline was attributed to their lower competitiveness at higher nutrient concentrations (Pedersen & Borum 1996) and the shading effect by the filamentous alga *Pylaiella littoralis* (L.) Kjellman combined with increased herbivory by *Idotea baltica* (Pallas) (Kangas et al. 1982). Offering habitat and food for many macroalgal and invertebrate species the species is recognized as one of the keystone species in the Baltic Sea area.

The eelgrass *Zostera marina* is the most common marine angiosperm in the Northern Hemisphere (den Hartog 1970). It is well represented also in the brackish Baltic Sea where the species grows at its lower salinity tolerance limit. Yet eelgrass is one of the most abundant macrophyte on exposed sandy bottoms in the Baltic Sea and is regarded as a key-species of this habitat. In the north-eastern part of Baltic Sea, the coastal waters of Estonia, the distribution of eelgrass has never been directly studied and, thus, the information on eelgrass communities is scarce and occasional.

Charophytes are a highly developed and diverse group of algae. They are widely distributed in freshwater, brackish, and marine habitats from tropical to polar regions (Wood & Imahori 1965). In the recent decades, species number, distribution area, and biomass of charophytes have significantly declined virtually in the whole Baltic Sea. This decline has been attributed to increased nutrient loads resulting in higher productivity of phytoplankton, epiphytic algae, and angiosperms and indirectly resulting in elevated grazing of mesoherbivores on charophytes (Kotta et al. 2004).

With the rise of new, powerful GIS tools and statistical techniques, the development of predictive habitat distribution models has rapidly increased in ecology. Such models are static and probabilistic in nature, since they statistically relate the geographical distribution of species or communities to their present environment. A wide array of models has been developed to cover aspects as diverse as biogeography, conservation biology, climate change research, and habitat or species management (Guisan & Zimmermann 2000).

Aims. The aim of this study was to model the probability of occurrence of *Fucus vesiculosus*, *Zostera marina*, and Charophytes in the BALANCE PA 4 in the northern

Baltic Sea. The available information on the species is very scattered and up to date we lack reliable information on the distribution of the species and this exercise is the first trial to overcome this shortcoming.

6.11.3 Material & Methods

The study area. For details on PA4 see section 2.5.1 herein.

Field sampling & data sources. The phytobenthos and associated environmental data were obtained from different sources during the period ** - **:

- Estonian Phytobenthos Monitoring – specially designed to obtain cover and biomass data of all macro phyte species. However, this information is collected from a limited number of transects only (<10 transects on the Estonian coast).

- Monitoring of different port areas. These studies provide good information on the small-scale variability of cover and biomass of phytobenthos species.

- Different mapping studies usually performed for other scientific purposes than the mapping of *F. vesiculosus*, *Z. marina,* and charophytes. These studies usually provide good information on the meso-scale variability of cover and biomass of phytobenthos species including of *F. vesiculosus*, *Z. marina*, and charophytes.

Samples were collected by a diver along transects in June–July **. Transects were situated perpendicular to the shore down to the depth limit of macro algae (** m). For each vegetation type three quadrate samples were taken (** m^2). At non-vegetated sites, sediment was collected with a core sampler (surface area 315 cm^2, sampled sediment layer 15 cm). Samples were sieved with a 0.25 mm mesh and frozen at –20 °C. At each sampling site sediment type, depth, coverage of phytobenthos, dominant species, and thickness of algal canopy were recorded. In the laboratory all samples were sorted under a binocular microscope (20–40 × magnification). All macro algae were identified to the species level, and the number of individuals of all species were counted and weighed. Prior to weighing the algae were dried at 60 °C for two weeks.

The quality of all data was quality ensured and subsequently transferred to a single geodatabase. Only recent data (<10 year old) were used for the analyses.

Predictor layers. To predict the distribution of the key macrophyte species the following layers were used: depth raster of sea area (50 m resolution), raster of seabed slope (50, 100, 500, 1000, 5000 m resolutions), raster of seabed sediment type (50 m resolution), and coastline vector data set.

Response variables. Point data on the presence/absence or biomass of phytobenthic species in the sampling stations were used as response variables. Prior to analyses, the different datasets on phytobenthos were pooled together (now available at the database of the Estonian Marine Institute).

Modelling. Generalized regression analysis and spatial prediction of the GRASP extension for the statistical software *S-PLUS* was used. Data (cell values) from all raster data sets were collected for each sampling station as well as for points of 50 m grids covering the whole pilot area using the Sample tool of ArcInfo. Probability of the presence of a phytobenthic species in each 50 m grid point was calculated using GRASP. AIC was used to select variables for the optimal model.

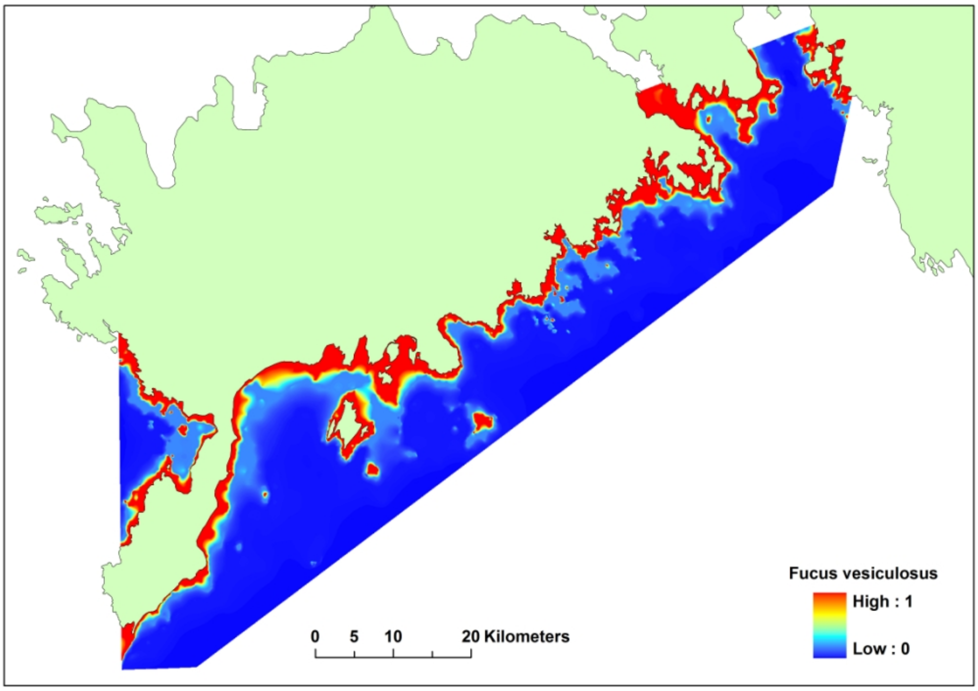


*Fig. **. Potential distribution of* **Fucus vesiculosus** *habitats in PA 4.*

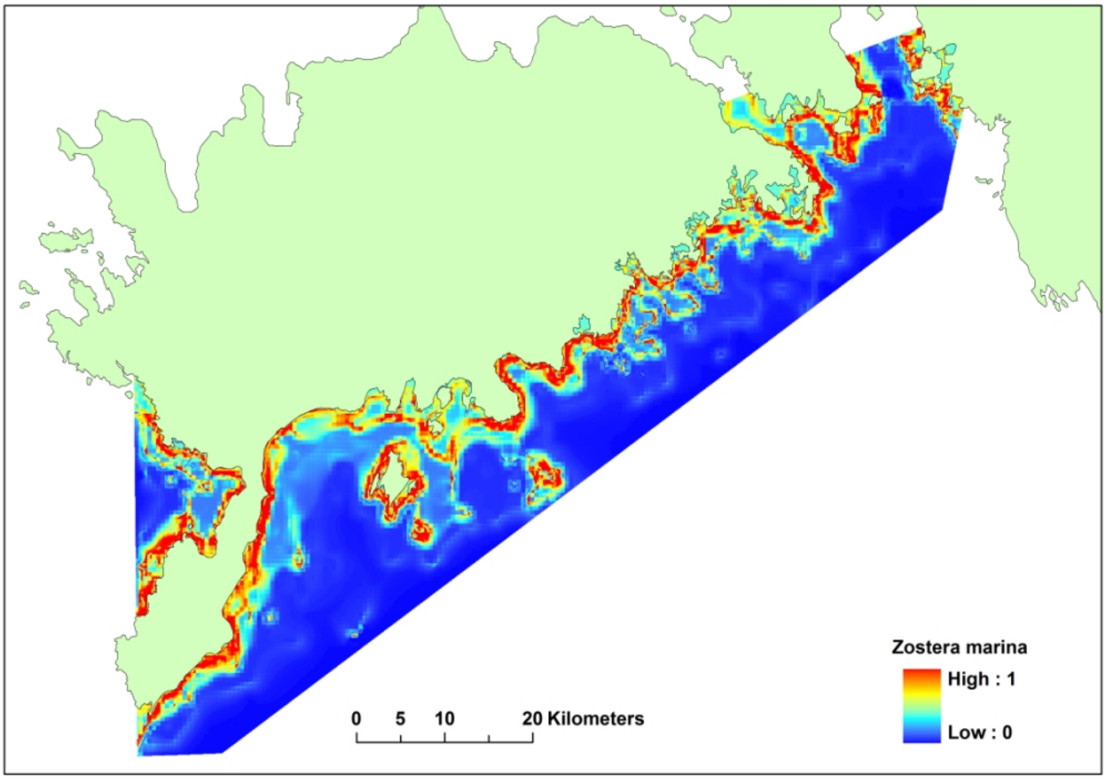


*Fig. **. Potential distribution of* **Zostera marine** *habitats in PA 4.*

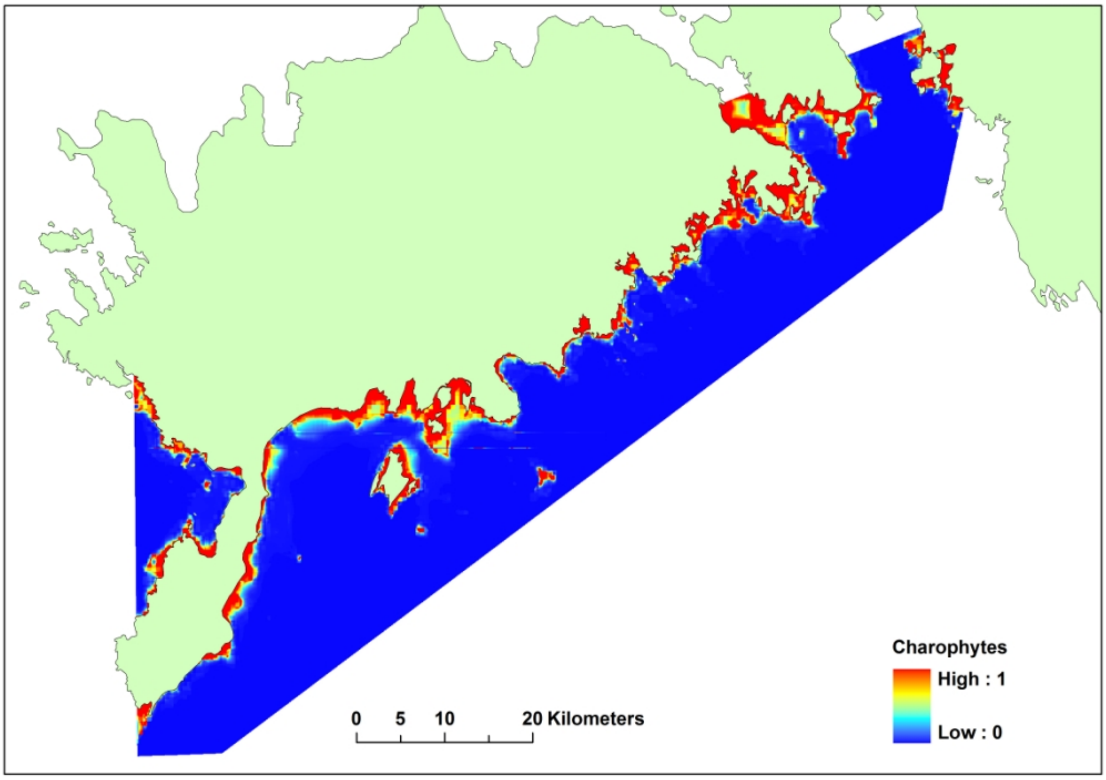


*Fig. **. **Potential distribution of charophytes habitats in PA 4.***

6.11.4 Results

The modelling results comprise of maps of potential occurrence of the habitat forming macrophyte species *Fucus vesiculosus*, *Zostera marina,* and charophytes in PA 4.

Validation. In GRASP, grasp.validate (GRASP.MOD.VALIDATE) has been added to allow a visual check of the relationship between fitted and observed data. A cross-validation was made with subsets of the entire dataset, where each subset contains an equal number of randomly selected data points. Each subset was then dropped from the model, the model was recalculated, and predictions made for the omitted data points. Combination of the predictions from the different subsets was then plotted against the observed data. A ROC (area under curve) and a COR (Spearman Correlation) statistics were used with binomial data (see Fielding & Bell 1997). A simple correlation coefficient was calculated for Poisson and normally distributed data.

6.11.5 Discussion

The observed effects of environmental factors on phytobenthic communities in the study area are many. The methods are very useful for this type of habitat modelling and the results are not very sensitive to the quality of existing data. To obtain high-quality, small-scale habitat maps from predictive modelling, it is essential to include the environmental variables only that are know to affect the distribution of the specific species.

Moreover, the variables have to available at a scale of both grain and extent necessary to catch the discrete boundaries of the habitat in question.

Substrate. In the Gulf of Riga the type of substrate was found to be the most important structuring factor for the phytobenthic communities in areas not influenced by direct riverine inflow. This was also true for the Väinameri where the dominating sandy and soft substrates in sheltered areas favoured development of rich phanerogam communities. The main differences in the phytobenthic communities of the Gulf of Riga and Väinameri compared to those of the northern and western coasts of the Baltic proper are caused mainly by dominance of different substrate types.

Salinity. In the Gulf of Riga the salinity gradient is more or less stable, with certain frontal areas near the major fresh-water inflows where the salinity gradient becomes especially steep. In these areas phytobenthic communities dominated by Chlorophyceae and phanerogams occur. In the Väinameri area the situation is different as the area is divided into two, more or less hydrologically distinct sub-basins. One has usually higher salinity and is influenced by the Baltic proper and the other is under the influence of the Gulf of Riga's frontal area causing fluctuations of salinity as well as nutrient concentrations with sometimes high amplitude (Suursaar *et al*., 1998). The latter causes complex effect on the phytobenthos communities e. g. the salinity is most probably responsible for the low species diversity of the area and the domination of the phanerogam species, which is also favoured by the presence of suitable soft substrates.

Light. In the Gulf of Riga the variation of light conditions on the seafloor contribute to the spatial differences in the structure of phytobenthos. The observed zones of phytobenthos follow the distribution of major freshwater inflows accompanied by decrease of water transparency. In Väinameri area the variation of light climate has more temporal character and less spatial influence. The periodical decrease of light quality could contribute to the smaller phytobenthos species diversity in the area but at the same time other factors (as substrate quality and wave exposure) seem to be more important on the formation of the phytobenthos.

Temperature. In the Väinameri, where the bottom water temperature can rise up to 20-24˚C even at the depth of 7-9 m, it ought to have a certain impact on the benthic communities inhabiting the seabed and thus may explain low biomass of benthic invertebrates.

Ice. In the vicinity of major riverine inflows - Pärnu Bay and the southern part of the Gulf of Riga - the severe ice conditions can cause a decrease of salinity in the phytobenthic zone during the winter period, preventing the wind-induced water mixing and formation of stable freshwater zone (1-3 m thick) below the ice. The effect of periodical changes of salinity conditions, most probably, contribute to the situation with dominance of green algae, low species diversity, and concentration of phytobenthos biomass to the shallowest part of the coastal sea observed in areas close to the Pärnu Bay and southern part of the Gulf of Riga. In the Väinameri area a strong correlation between the total biomass of the loose *Furcellaria lumbricalis-Coccotylus truncatus* community and the length of the ice cover period was established. Here, the effect of the length of the ice cover on the biomass of the loose red algae community was considered to be expressed in preventing the wind induced washing ashore of the loose algae. On the other hand, in the Gulf of Riga the observed biomass maximum usually occurred at the depth of 1-2 m and was caused by the perennial *Fucus* communities, which have been reported to be tolerant of limited ice scraping (Kiirikki & Ruuskanen 1996). Hence, the effect of ice scraping in this sea area seems to be quite limited. The lower phytobenthic biomass near the water surface is probably caused by the combined effect of the water-level fluctuation and ice.

Water movement. Changes in the water level in PA 4 may fluctuate with amplitude of more than 1 m, which can cause desiccation of the algae close to surface. Waves can stimulate the growth of macro algae, such as *Fucus vesiculosus*, through active ventilation of dense stands increasing the light exposure of lower canopies and removal of silt and metabolites. Moderate waving activity helps *Fucus* to control the epiphytic growth of filamentous algae (Kiirikki 1996b). In the Gulf of Riga exposed shores are common throughout the area. Wave activity is commonly high and thus having minor importance in explaining the observed differences in phytobenthic communities. In the Väinameri area, where numerous bays and islets create sheltered habitats, the situation is different. The variation in the wave exposure within the Väinameri area, affected the variability in the structure of phytobenthic communities which was also proved by the multivariate analyses carried out during our study.

Nutrients. In the Gulf of Riga the communities dominated by annual species occur in areas close to and in the Pärnu Bay and in the southern part of the Gulf. Also, the areas on the southern coast of the Saaremaa Island were dominated by annual species. Here, this was probably due to the salinity regime and high proportion of unstable soft substrate rather than nutrient conditions. As the background concentrations of nutrients are much higher in the Gulf of Riga than in the open Baltic proper, the extremely high biomass found in the Gulf could be caused by better nutrient availability throughout the vegetation period.

Sedimentation. In the case of the Gulf of Riga with open shores the sedimentation in general is probably not an important factor influencing the depth limit of the phytobenthic communities. In most cases the observed substrate was not limiting the depth distribution of phytobenthos species. Evidently, the sedimentation plays very important role in the areas of riverine inflow in the southern part of the Gulf of Riga and in the Pärnu Bay area where the lack of suitable substrate for the majority of the phytobenthic species was observed.

6.11.6 Perspectives

Valid habitat maps require knowledge of the factors that potentially affect the distribution of the species. Ideally, only these key environmental factors should be included in the model. However, often there exists no data on the environmental predictor variables required, or they are not available at a suitable scale. It is known that processes affect ecosystem simultaneously at various spatial scales resulting different spatial patterns of abiotic and biotic environment (Gutt & Piepenburg 2003, Denny et al. 2004). It has been suggested that strong abiotic disturbance may reduce the importance of the biotic interactions within communities in the Baltic Porper (Flöder & Sommer 1999, Buckling et al. 2000, Worm et al. 2002). In rare occasions, physically driven fluxes may override the effects of biological interactions (Herkül et al. 2006). When this is the case, changes in the physical environment explain a larger part of the variability of species distribution in shallow water in the Baltic Proper (Kotta et al. 2007). In PA 4, changes in the physical environment explain the larger part of the variability in benthic macro algae communities. It is therefore rewarding to seek the relationship between phytobenthos and available physical environment at multitude of spatial scales. For example bottom slopes at various spatial scales characterise different abiotic processes operating at different spatial scales and thus may describe better the distribution of phytobenthic species than depth or slope value at a single spatial scale alone.

6.12 *PA 4. Spatial prediction of* Furcellaria lumbricalis *in Lithuanian coastal waters*

Authors: D. Daunys, M. Bucas, and P. Zemlys.

6.12.1 *Introduction*

**

Aims. The aim was to use predictive modelling to find the spatial distribution of a reef habitat formed by the perennial red algae *Furcellaria lumbricalis.* In conditions of high fetch in the southeastern Baltic Sea, this species is the only habitat forming species in the coastal waters.

6.12.2 *Material & Methods*

The study area. The study was carried in in PA 4 along the continental part of the Lihtuanian coast from Klaipeda to the Latvian border. It covers the photic zone of coastal waters from 0 to 20 m depth, and an area of approx. 300 km^2 (fig. 1).

Field sampling & data sources. **

Predictor layers. Three principally different variables are included in the model: substrate, depth and exposure. We use three classes of sediment in the model: 1) sand and gravel, 2) pebble, and 3) cobbles and boulders in depths between 1 and 22.5 m. In case of exposed coastline, exposure estimate was used to take into account effects of a shelter provided by underwater seabed topography. Exposure parameter was calculated following fetch approach and defined as an average distance from a given location to the boundary depth in three main directions of the strongest winds. Two different boundary depths (20 and 30 m) were used for modeling (fig. 2).

Response variables. The response variable comprises of a dataset with presence/absence data of *F. lumbricalis* reefs in the area carried out at 460 sites during the last 15 years. Identification of the reef habitat is based on interaction of two biological variables: substrate coverage by algae and mussels. Since mussels are out competed by red algae in the reef, its presence is denoted by combination of algae coverage >60 % and coverage of mussels <60 %. These cases account for 8 % (n=37) of the total number of observations (fig. 1).

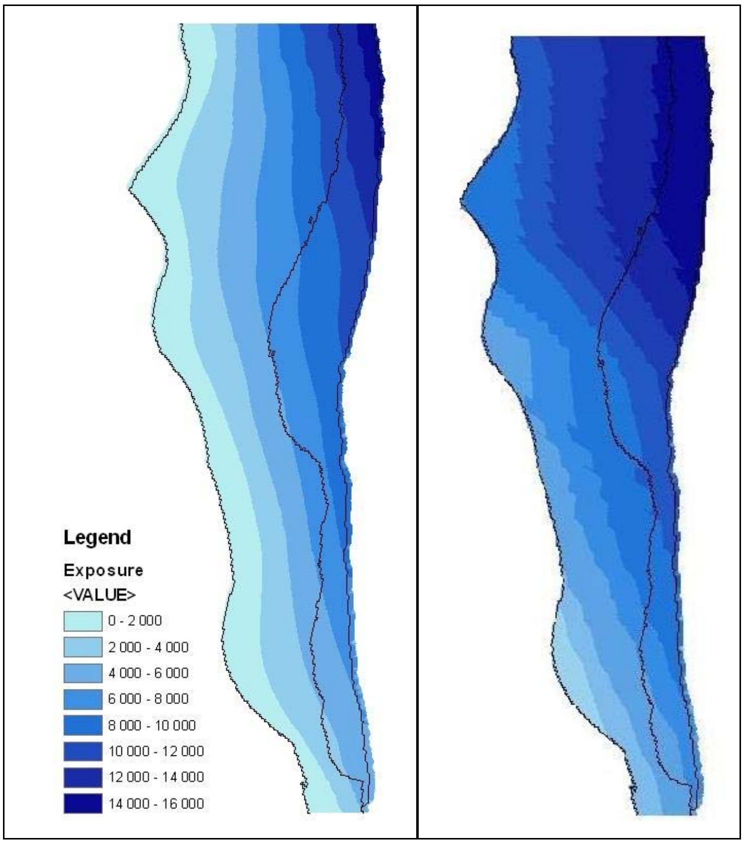


Figure 2. Calculated exposure values (meters) for the Lithuanian coastal waters: based on distance to the depths of 20 m (left) and 30 m (right).

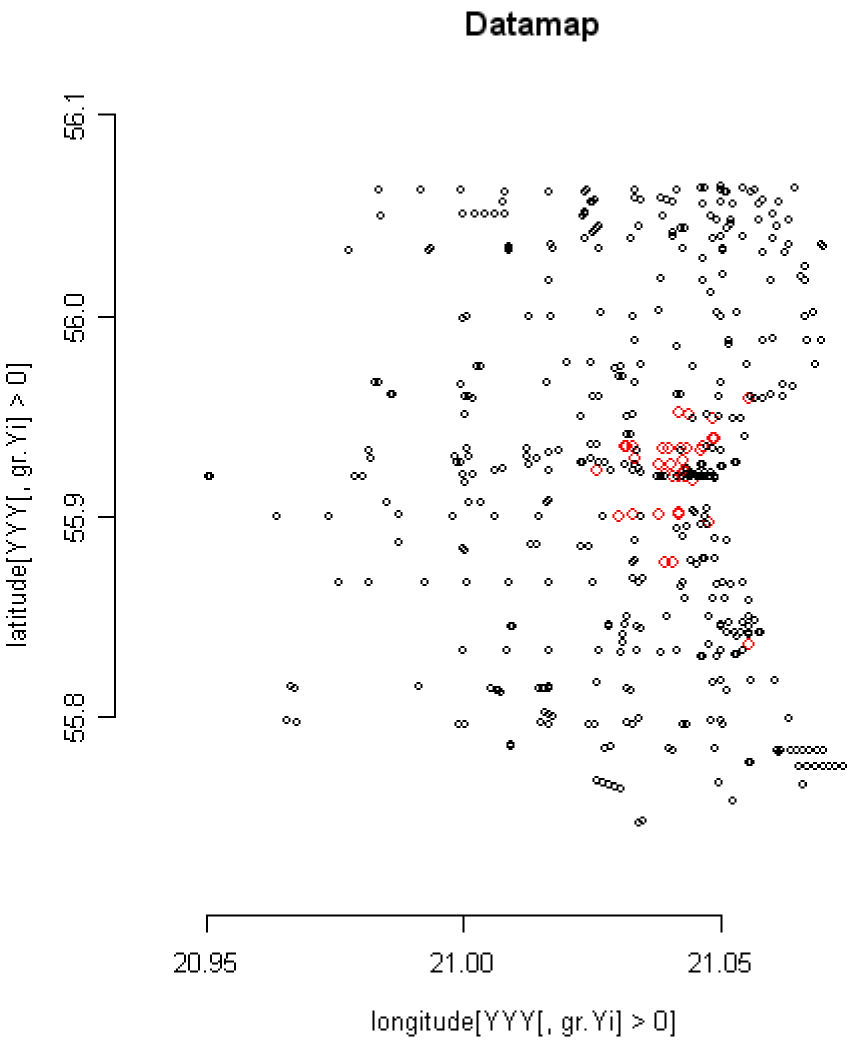


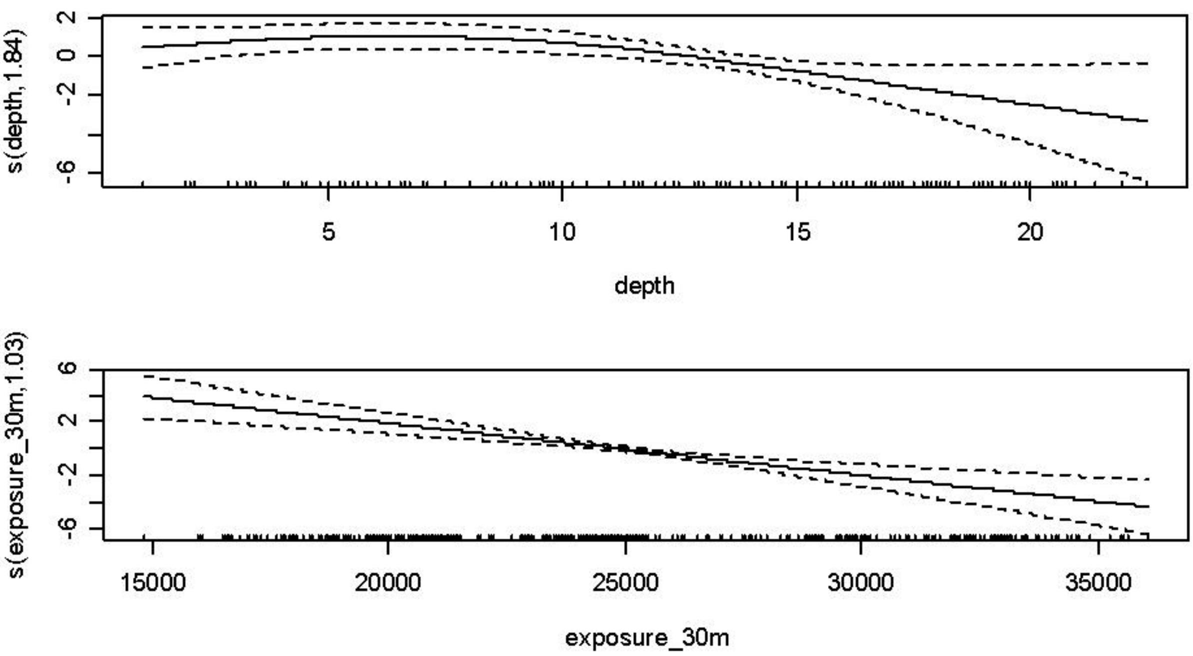
Figure 1. Spatial distribution of reefs (dots in red) formed by perennial red algae Furcellaria lumbricalis in the Lithuanian coastal waters.

Model selection. The generalized regression and spatial prediction (GRASP) (Lehmann et al. 2002) was used for R for the modelling and spatial prediction of the habitat.

**

6.12.3 Results

Based on quasibinomial distribution (binary response variable), the model explains 49.4 % of the deviance in the distribution of the habitat. Selection of variables revealed sediment, depth, and exposure based on 30 m boundary depth as significant predictors, exposure being the most important in the model. There was negative relationship between exposure and occurrence of the reefs with the bell-shaped response curve for the relationship between depth and reef distribution (fig. 3). By factor drop contribution, elimination of exposure factor results in 23 % decrease of explained total deviance, whereas elimination of the depth leads to 4 % decrease only. In respect to factors contribution alone, both sediment and depth are equally important and each contribute with approx. 15 % explanation of the reef occurrence.



Validation.**

6.12.4 Discussion

Model validation results show lower prediction for the reef occurrence in comparison to its absence. The model gives negative reef predictions in 407 cases out of 424 actual negative observations (96 % of matching observations and predictions). However, there is much lower prediction accuracy for positive outcomes. Out of 37 positive cases the model gives only 18 adequate predictions (49 % of matching observations and predictions). We believe that model limitations in predicting positive outcomes are caused by two factors associated with small scale environmental heterogeneity. Point measurements of the depth at the observation sites (input data) do not provide information on local conditions of exposure (sheltered or not) caused by seabed elevations, which are distant from the observation sites. This phenomena will also be hardly simulated by spatial prediction taking into account spatial resolution of available bathymetry data. On the other hand, the importance of distant sand fields for protection from waves of the habitat forming red algae species was recently demonstrated (Bucas et al. in press). It was found, that neighboring sand (i.e. a mobile sediment) may effectively limit colonization of the substrate by the red algae through abrasive effects. This effect may also be captured by data at a finer spatial resolution of sediment data and using polygon based data for statistical model rather than point observations. In spite of these shortcomings originating from the quality of available data, prediction accuracy possibly will be increased during the final stage of testing predicted habitat maps.

6.12.5 Perspectives

**

About the **BALANCE** project:

The BALANCE project aims to provide a transnational marine management template based on zoning, which can assist stakeholders in planning and implementing effective management solutions for sustainable use and protection of our valuable marine landscapes and unique natural heritage. The template will be based on data sharing, mapping of marine landscapes and habitats, development of the blue corridor concept, information on key stakeholder interests and development of a cross-sectoral and transnational Baltic zoning approach. BALANCE thus provides a transnational solution to a transnational problem.

The work is part financed by the European Union through the development fund BSR INTERREG IIIB Neighbourhood Programme and partly by the involved partners. For more information on BALANCE, please see www.balance-eu.org and for the BSR INTERREG Neighbourhood Programme, please see www.bsrinterreg.net

The **BALANCE Report Series** includes:

BALANCE Interim Report No. 1	Delineation of the BALANCE Pilot Areas
BALANCE Interim Report No. 2	Development of a methodology for selection and assessment of a representative MPA network in the Baltic Sea – an interim strategy
BALANCE Interim Report No. 3	Feasibility of hyperspectral remote sensing for mapping benthic macroalgal cover in turbid coastal waters of the Baltic Sea"
BALANCE Interim Report No. 4	Literature review of the "Blue Corridors" concept and its applicability to the Baltic Sea
BALANCE Interim Report No. 5	Evaluation of remote sensing methods as a tool to characterise shallow marine habitats I
BALANCE Interim Report No. 6	BALANCE Cruise Report – The Archipelago Sea
BALANCE Interim Report No. 7	BALANCE Cruise Report – The Kattegat
BALANCE Interim Report No. 8	BALANCE Stakeholder Communication Guide
BALANCE Interim Report No. 9	Model simulations of blue corridors in the Baltic Sea
BALANCE Interim Report No. 10	Towards marine landscapes of the Baltic Sea
BALANCE Interim Report No. 11	Fish habitat modelling in a Baltic Sea archipelago region
BALANCE Interim Report No. 12	Evaluation of remote sensing methods as a tool to characterise shallow marine habitats II
BALANCE Interim Report No. 13	Harmonizing marine geological data with the EUNIS habitat classification
BALANCE Interim Report No. 14	Intercalibration of sediment data from the Archipelago Sea
BALANCE Interim Report No. 15	Biodiversity on boulder reefs in the central Kattegat
BALANCE Interim Report No. 16	The stakeholder – nature conservation's best friend or its worst enemy?
BALANCE Interim Report No. 17	Baltic Sea oxygen maps
BALANCE Interim Report No. 18	A practical guide to Blue Corridors
BALANCE Interim Report No. 19	The BALANCE Data Portal
BALANCE Interim Report No. 20	Pelagic habitat mapping: A tool for area-based fisheries management in the Baltic Sea
BALANCE Interim Report No. 21	Mapping of marine habitats in the Kattegat
BALANCE Interim Report No. 22	E-participation as tool in planning processes
BALANCE Interim Report No. 23	The modelling of *Furcellaria lumbricalis* habitats along the Latvian coast
BALANCE Interim Report No. 24	Towards a representative MPA network in the Baltic Sea
BALANCE Interim Report No. 25	Towards ecological coherence of the MPA network in the Baltic Sea
BALANCE Interim Report No. 26	What's happening to our shores?
BALANCE Interim Report No. 27	Mapping and modelling of marine habitats in the Baltic Sea
BALANCE Interim Report No. 28	GIS tools for marine planning and management
BALANCE Interim Report No. 29	Essential fish habitats and fish migration patterns in the Northern Baltic Sea
BALANCE Interim Report No. 30	Mapping of Natura 2000 habitats in Baltic Sea archipelago areas
BALANCE Interim Report No. 31	Marine landscapes and benthic habitats in the Archipelago Sea
BALANCE Interim Report No. 32	Guidelines for harmonisation of marine data
BALANCE Interim Report No. 33	The BALANCE Conference

In addition, the above activities are summarized in four technical summary reports on the following themes 1) Data availability and harmonisation, 2) Marine landscape and habitat mapping, 3) Ecological coherence and principles for MPA selection and design, and 4) Tools and a template for marine spatial planning. The BALANCE Synthesis Report *TOWARDS A BALTIC SEA IN BALANCE* integrates and demonstrates the key results of BALANCE and provides guidance for future marine spatial planning.