K. Herkül, R. Aps, K. Kostamo, J. Kotta, L. Laamanen, J. Lappalainen, K. Lokko, A. Peterson, R. Varjopuro

Deliverable T2.1.1

“The Gulf of Finland marine and coastal environmental vulnerability profile”

SYKE.FI/PROJECTS/PLAN4BLUE

ABSTRACT

Title: Deliverable T2.1.1 “The Gulf of Finland marine and coastal environmental vulnerability profile”

Authors: K. Herkül, R. Aps, K. Kostamo, J. Kotta, L. Laamanen, J. Lappalainen, K. Lokko, A. Peterson, R. Varjopuro

Abstract:

The EU Marine Spatial Planning (MSP) Directive establishes a framework for maritime spatial planning aimed at promoting the sustainable growth of maritime economies, the sustainable development of marine areas and the sustainable use of marine resources. The marine environment is heavily impacted by human activities especially in intensively used sea areas such as the Baltic Sea where the assessments of environmental vulnerabilities and cumulative risks are increasingly demanded in environmental decision and policymaking. In this study we developed the Gulf of Finland marine and coastal environmental vulnerability profile as a spatial data layer that incorporates the distribution of nature values and their sensitivities to disturbances. Marine and coastal environmental vulnerability profile is covering the marine open sea area as well as the coastal shallow sea area of the Gulf of Finland.

The aim of this study was to develop cross-border marine and coastal environmental vulnerability profile of the Gulf of Finland, which can be used for ecosystem based MSP processes in Estonia and Finland, in order to find solutions that lead to sustainable use of resources and to improved planning and management of the marine and coastal areas. The main product of this report was:

- Environmental vulnerability profile (EVP) – a spatial data layer that incorporates the distribution of nature values and their sensitivities to disturbances; higher value indicates a presence of more sensitive nature values.

The distribution of the following nature values were included in the calculation of EVP

- Key seabed flora and fauna: bladder wrack, red seaweed *Furcellaria lumbricalis*, filamentous algae, epibenthic bivalves, infaunal bivalves, vascular plants, charophytes;
- Species richness of seabed flora and fauna;
- Water birds;
- Seals.

Acknowledgements

The authors are thankful for INTERREG Central Baltic programme, Estonian Environmental Investment Centre and the Regional Council of Southwest Finland for co-funding the Plan4Blue project.

The Estonian Environmental Investment Centre and European Cohesion Fund supported the **aerial wintering birds survey project** across the Estonian coastal sea. This work is and will be financed during the years 2014-2020 by the budget of structural support priority direction No. 8 "Green infrastructure and increasing the preparedness of emergencies" measure activity 8.1.6 "Protected species and habitats inventories and data collection". All the work was carried out by the Estonian University of Life Sciences.

Finnish seal data that was used in this project, originated from Parks & Wildlife Finland and was issued by the “Government Decree 736/2001”.

Estonian seal data that was used in this project, originated from EELIS (Estonian Natura Information System) – Estonian Environmental Register: Estonian Environment Agency.

The document should be sited as follows:

K. Herkül, R. Aps, K. Kostamo, J. Kotta, L. Laamanen, J. Lappalainen, K. Lokko, A. Peterson, R. Varjopuro. 2017. Plan4Blue report Deliverable T2.1.1 “The Gulf of Finland marine and coastal environmental vulnerability profile”.

Contents

1 Introduction

1.1 Concept of environmental vulnerability

Human use of marine and coastal areas is increasing worldwide, resulting often in conflicts between different interests for the space and marine resources and environmental sustainability. The marine environment is quite frequently heavily impacted by human activities especially in intensively used sea areas like the Baltic Sea (Korpinen et al. 2012), where competition on the access to sea area exceed national borders. The multiple competing uses of marine and coastal areas have resulted in a rapid increase of marine spatial planning initiatives to sustainably use marine resources as well as to solve cross-sectoral and transboundary conflicts over space (Douvere and Ehler 2010; Stelzenmüller et al. 2015). Marine spatial planning (MSP) requires a cross-sectoral, transboundary and ecosystem based approach. Assessing ecological vulnerability can thus provide an important tool to support blue growth and to preserve the capacity of ecosystems to provide valued services. Thus, vulnerability assessments are increasingly used and demanded in environmental decision-making and policy-making.

The concept of vulnerability in ecology is relatively new and has been increasingly used only in the last few decades (Beroya-Eitner 2016). The environmental vulnerability has been assessed using a single species or specific target group, e.g., water birds (Sonntag et al. 2012; Ogden et al. 2014) communities, habitats and whole ecosystem (e.g. Certain et al. 2015), and a single pressure (e.g. Hiddink et al. 2007) to multiple pressures (e.g. La Rivière et al. 2016). Most vulnerability assessments that aim to contribute directly to a MSP are either regional or national (e.g. Foley et al. 2013; La Rivière et al. 2016) and only seldom multinational/cross-border (e.g. Martin et al. 2009).

The observable effects of intensive use of marine resources and climate changes have rendered vulnerability assessment to a necessity (Hinkel 2011), essential for delivering the objectives set out under European Directives, including the EU MSP Directive (EU 2014). However, the term vulnerability has many definitions and there is an array of other terms with overlapping meaning (Hinkel 2011; Beroya-Eitner 2016). Moreover, the definitions do not provide much guidance for designing methodologies for assessing vulnerability, as vulnerability itself is truly a theoretical concept and thus immeasurable (Hinkel 2011). On the most general level, vulnerability can be seen as a measure of possible future harm or potential threats (Cutter et al. 2003; Hinkel 2011).

As with the definition, there are also many approaches for assessing vulnerability – there is no generally accepted or conceptual framework, despite several attempts to clarify concepts and methodologies (Hinkel 2011; Beroya-Eitner 2016). Thus, methodologies for assessing vulnerability are generally developed based on the specific case and/or area, although there are several guidelines available (e.g. Villa and McLeod 2002; Ippolito et al. 2010; Cormier et al. 2013; Ardron et al. 2014).

While vulnerability indicators provide a numerical scoring of the status/prognosis, a spatially explicit assessment can be preferential for MSP. The terms used for quantification of vulnerability include among others vulnerability profile (as in Cormier et al. 2013), ecological vulnerability (as in Okey et al. 2015) and sensitivity assessment (as in La Rivière et al. 2016). We find the environmental vulnerability profile as the most appropriate term for a spatially explicit assessment of environmental vulnerabilities.

Regardless of different definitions and methodologies, environmental vulnerability assessment is usually based on sensitivity to pressures, expressed as resistance and/or recovery potential, i.e. resilience, and exposure to the pressures (Bax and Williams 2001; Hiddink et al. 2007; De Lange et al. 2010; Beroya-Eitner 2016). Sensitivity has been expressed as the inverse of recovery time for a nature value, e.g. community, habitat, ecosystem, after being exposed to a pressure (Hiddink et al. 2007), or as a “capacity to tolerate a pressure (resistance) and the time needed to recover after an impact (resilience)” (La Rivière et al. 2016).

The actual assessment or quantification of ecosystem sensitivity is practically impossible, as ideally it should include all species, habitats and their interactions, and all the relevant economic activities in the maritime sectors that may have an impact on ecosystem. Moreover, the pressures generated by different activities interact and make the picture immensely complex. Marine spatial planning, however, requires adequate vulnerability assessments with the available data, within the given timeframes. Thus, it is aimed to produce the least disappointing vulnerability profile with the resources and time available (Villa and McLeod 2002), so that best-informed decisions could be made in MSP.

While developing methodology for vulnerability assessment, it would be a good starting point to address the concept of value (Villa and McLeod, 2002): which nature values should be preserved/maintained to achieve MSP goals? These nature values (or significant ecosystem components as used in Stelzenmüller et al. 2015) can be, e.g., ecologically important habitats and species, attributes of an important ecosystem service, rare or endangered species. The criteria that has been used to quantify their sensitivity often include uniqueness or rarity of species or habitat, functional significance, e.g., habitat-forming species, fragility or susceptibility to degradation of species or habitat, and life history traits that are related to recovery potential (Ardron et al. 2014). Expert-derived ratings or rankings based on available scientific literature are widely used to assess intrinsic sensitivity of the nature values and their sensitivities to stressors (Villa and McLeod 2002; De Lange et al. 2010; Okey et al. 2015; La Rivière et al. 2016). In addition to intrinsic factors of being vulnerable, exposure and sensitivity to pressures should be addressed as suggested by De Lange et al. (2010): is the ecosystem component exposed to a pressure, what is the effect of pressure and what is the time needed for recovery (La Rivière et al. 2016)? While creating vulnerability profile, it should be carefully considered how to combine together the scores of nature values/ecosystem components sensitivities as simply adding scores together might be improper, e.g., when the scores are statistically independent (Ardron et al. 2014).

Data availability plays a major role in developing methodology for vulnerability assessment (Hinkel 2011), however, a comprehensive vulnerability assessment is pressure-driven and includes exposure, sensitivity and recovery of a nature value/ecosystem component to pressures (De Lange et al. 2010), and is based on best available knowledge (La Rivière et al. 2016). In this respect, the Baltic Sea can provide an interesting possibility to develop the methodology due to the fact that extensive datasets are available for analyses.

The Baltic Sea is the largest brackish-water basin in the world. The catchment area covers over 1,700,000 km^2 and is home for over 84 million inhabitants (HELCOM 2011). The combination of vertical stratification, high population density and well-developed agricultural sector in the catchment area and a small body of water with limited exchange with the North Sea makes the Baltic Sea vulnerable and sensitive to nutrient enrichment and eutrophication (HELCOM 2009). Because of large freshwater inflow and limited connection to the North Sea, the salinity in the Baltic Sea is much lower than in true oceanic waters, which makes the sea even more sensitive as relatively few species can thrive in such brackish-water conditions (HELCOM 2009).

The Gulf of Finland is considered one of the most eutrophicated basins in the Baltic Sea area with the nutrients input and trophic state increasing from west to east (HELCOM 2003; Pitkänen et al. 2007). As compared to other basins in the Baltic Sea, the Gulf of Finland has a relatively large catchment area and the greatest freshwater inflow that results in a strong horizontal salinity gradient. The surface salinity in the gulf varies from 0 in its eastern end to 7 ppt in the western areas (Pitkänen et al. 2008).

1.2 Nature values of marine environment

In order to develop spatially explicit vulnerability profile only the spatially mappable ecosystem components (species or groups) were included in this project. In addition, the data availability had a major role in selection of the input biological components in this study. Due to these constraints seabed flora and fauna, birds, and seals were included. Fish and plankton were not included because of both, the high spatial and temporal dynamics of these ecosystem components, and the lack of georeferenced data. However, the distribution of seabed biota has a strong linkage to fish because there are certain essential fish habitats (spawning, feeding) that are primarily defined by the benthic plants and invertebrates.

Several macrophyte and benthic invertebrate species found in the Baltic Sea can be considered as habitat-forming species: they are capable of creating a specific local environment that facilitates colonization of other species that otherwise would not be present in the area (Martin et al. 2013). Seaweeds provide food, habitat or both to many invertebrates, fish, and birds. Canopy-forming species modify the spatial complexity and their loss can cause drastic changes in the structure of associated assemblages (Bertocci et al. 2010). Similarly to seaweeds, also macrobenthos plays essential role in the functioning of marine ecosystems. Benthos influences through food web all trophic levels from phytoplankton to seabirds (e.g. Beukema and Cadée 1996), serving as an important food source for other fauna (e.g. larger invertebrates, birds, fish) or cleaning the water by removing sediments and organic matter (e.g. filter feeders) and controlling the phytoplankton biomass (Manganaro et al. 2009).

Brown algae bladder wrack (*Fucus vesiculosus*) is a common and widespread canopy-forming perennial species in the Baltic Sea. It is an important habitat for a number of specific faunal species and supports a high biomass of invertebrates (Wikström and Kautsky 2007). *F. vesiculosus* hosts an array of epiphytic taxa, including macroalgae, sessile invertebrates, and mobile invertebrates (Kersen et al. 2011).

Red algae *Furcellaria lumbricalis* is another important habitat-forming species in the Baltic Sea. It has been shown to support elevated macrozoobenthos densities by offering a secondary substrate (Kotta and Orav 2001). Moreover, stony bottom reefs with *F. lumbricalis* have been considered as the most valuable community in the southeastern Baltic Sea, based on species richness (Šiaulys and Bučas 2015). In addition, it serves as a spawning ground for herring.

Filamentous algae can be seen as a nuisance as their accumulations are not only unsightly, but decomposing drifting mats can cause hypoxia and considerable losses to benthic life in areas in which they cover large surfaces for long periods of time (Arroyo and Bonsdorff 2016). However, they are also a food source for e.g. protozoans, insects, and fish. Filamentous algae, e.g. *Cladophora glomerata*, support rich and highly abundant macrofaunal assemblages, with more than 40 different taxa of macrofauna represented, thus serve as useful habitats for littoral macrofauna (Salovius and Kraufvelin 2004). It has been found that herring eggs attached to filamentous algae have remarkably low mortality, thus filamentous algae are important spawning ground for herring (Rajasilta et al. 2006).

Seagrass patches are among the most diverse coastal ecosystems in the northern Baltic Sea. Highly productive seagrass meadows serve as nursery and feeding areas, support high faunal diversity, stabilise sediment, prevent resuspension and play important role in carbon cycle (Duarte 2002). In the Baltic Sea, eelgrass meadows support the highest diversity of angiosperms and abundant invertebrate communities (Boström et al. 2014).

Charophytes are significant feeding and nursery areas for several fish species and birds and an important habitat for many of invertebrate species, providing a refuge for zooplankton from fish predation (Torn 2008). Charophytes are important in benthic invertebrate’s diet and can be extensively consumed by waterfowl, thus forming a substantial component of the food-web (Schmieder et al. 2006). They might also promote increased water transparency (Nõges et al. 2003).

Bivalves play an important role in the Baltic Sea food webs. The blue mussel *Mytilus trossulus* is considered as one of the key species in the Baltic Sea ecosystem (Koivisto and Westerbom 2010). As a habitat-modifying species, the blue mussels promote and maintain biodiversity, thus have fundamental importance to community structure (Koivisto and Westerbom 2010). Epibenthic bivalves are important food source for fish and benthos feeding birds (Rask 1989; Stempniewicz and Meissner 1999; Lappalainen et al. 2004). The suspension feeding bivalves form a very important trophic link between pelagic and benthic systems (Lauringson et al. 2009)

Birds are considered to be good environmental indicators and are often used to assess the status of marine environment. The Gulf of Finland is important to water birds due to its location on East-Atlantic flyway used by most Arctic water bird species during their migration between breeding and wintering areas. The shoals in Estonian coastal sea are suitable intermediate feeding and resting sites for further migration. The same shoals are often used also for moulting and wintering. Abundance of those bird species are strongly influenced by climate change and pressures to the marine environment (Luigujõe and Auniņš 2016).

Seals are one of the most important top predators of the food chains in the Baltic Sea. Seals are considered very sensitive component of the Baltic ecosystem as their populations decreased dramatically in by the 1970s due to excessive hunting and environmental pollutants in the Baltic Sea. Although nowadays the populations have increased, they continue to be vulnerable due to coastal development, overfishing, environmental contaminants and entanglement of young seals in fishing gear (HELCOM 2013).

1.4 Aims of the report

The aim of this study was to develop transboundary marine and coastal environmental vulnerability profile for the Gulf of Finland, which can be used for ecosystem based MSP processes in Estonia and Finland. Developing environmental vulnerability profiles can facilitate discovering solutions that lead to sustainable use of marine resources and improve planning and management of the marine and coastal areas. Marine and coastal environmental vulnerability profile is covering the marine open sea area as well as the coastal shallow sea area of the Gulf of Finland. The main products that were developed were georeferenced spatial layers of:

- Distribution of benthos related nature values: benthic species/groups, benthic biodiversity;

- Environmental vulnerability profile (EVP) of the Gulf of Finland project area – a spatial data layer that incorporates the distribution of all nature values and their sensitivities to disturbances;

2 Material and methods

2.1 Benthic species

2.1.1 Study area

The tideless Baltic Sea can be characterized by a steep salinity gradient resulting in a variable fauna and flora, which tolerates well the prevailing environmental conditions. Materials for the study originated from Estonian and Finnish marine areas (Figure 2.1.1.1), located in the northern Baltic Sea. Marine and coastal environmental vulnerability profile is covering the marine open sea area as well as the coastal shallow sea area of the Gulf of Finland.

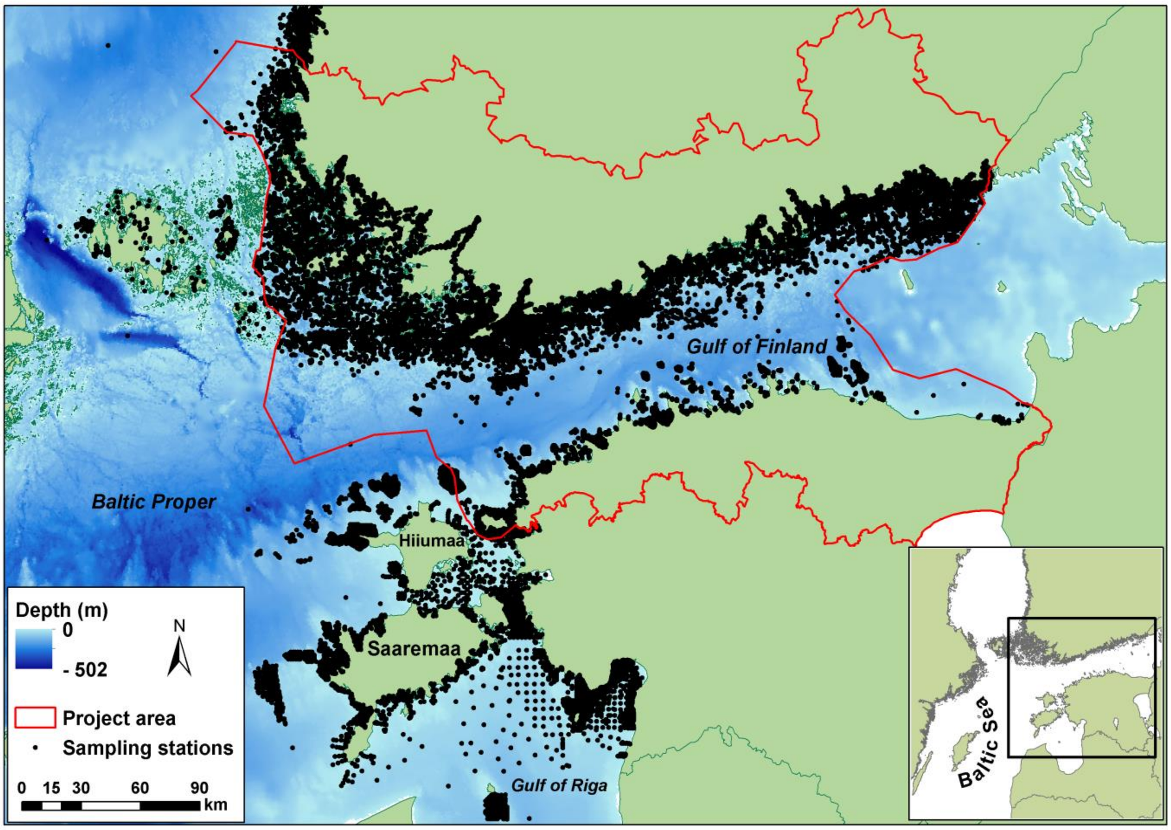

Figure 2.1.1.1. Locations of sampling stations.

The area includes seven major sub-basins of the Baltic Sea: Archipelago Sea, Bothnian Bay, Bothnian Sea, the Northern Baltic Proper, Gulf of Finland, Gulf of Riga and the Quark. The sea area of Åland was excluded from the area due to lack of field observations (except for the benthos samples that were included). All of the sub-basins exhibit strong gradients of wave exposure, depth, and salinity. The sea areas west of the islands Saaremaa and Hiiumaa and the southwestern outer Archipelago Sea are exposed to the open Northern Baltic Proper and have a wave fetch of hundreds of kilometres. In contrast, the inner reaches of the bays of the mainland are very sheltered both by the mainland and by islands. Salinity exceeds 7 PSU in the westernmost study area while it falls to almost 0 PSU in the inner parts of bays with riverine inflow and also in the Bothnian Bay (Kautsky and Kautsky 2000; Karlson et al. 2002; Zettler et al. 2013).

2.1.2 Biological data

The data for the work was collected from different sources. A macrobenthos database of the Estonian Marine Institute, University of Tartu and HERTTA database of the Finnish Environment Institute were used as data sources for Estonian and Finnish datasets, respectively. Data from 11523 Estonian and

2888 Finnish benthos sampling stations were used as an input for species distribution and biodiversity models. Estonian dataset covered both flora and fauna observations while in Finland HERTTA database was used only for benthic invertebrate data. All the samples were collected between the years 2005 and 2015 in Estonia and 2000 and 2015 in Finland. The sampling stations covered a depth range of 0.1 meters to 193 meters in Estonian waters and 0.1 to 286 meters in Finnish waters.

Finnish macrophyte data were collected by The Finnish Inventory Programme for the Underwater Marine Environment (VELMU) and the dataset consisted of 27367 dive transect points recorded between the years 2004 and 2016 that were used for assessing the macrophyte species richness. In addition to dive data, Finnish macrophyte models included also drop-video data that consisted of 93363 points that were partly stratified randomized along environmental variable gradients and partly placed in grids. The exact amount of point data used in modelling varied slightly between the models due to data type used in modelling (dive or dive and video data). All models were predicted to whole national sea areas using spatially comprehensive datasets but the final products were calculated for the Plan4Blue study area (Figure 2.1.1.1).

Ekman and Van Veen type bottom grab samplers were used for benthic invertebrate samples on soft sediments in both countries. Scuba divers collected all the fauna and flora inside a Kautsky sampler that is a 0.04 m^2 metal frame. In Estonia this data were used to analyse both flora and fauna while in Finland only benthic invertebrates were examined from the samples. Benthic samples were sorted in a laboratory and all macrobenthic organisms were identified under a microscope. Abundances of all taxa were quantified. Sampling and analysis followed the guidelines developed for the HELCOM COMBINE programme (HELCOM 2015).

In Finland, all macrophyte species within an inspection square, usually 4 m^2, were recorded to species level whenever possible. The unidentified species were identified in laboratory. Macrophytes were also determined to the species level with some exceptions. Finnish drop-video data were only applicable with large, easily identifiable species and were not used with every modelled species or group. The full list of macrobenthos taxa with additional information is presented in Appendix 1.

There were ten important benthic species or groups of species with different ecosystem functions and recovery potentials chosen to represent benthic nature values: bladder wrack (*Fucus vesiculosus*), the perennial red seaweed *Furcellaria lumbricalis*, filamentous algae, epibenthic bivalves (*Mytilus trossulus*, *Dreissena polymorpha*), vascular plants (excluding *Zostera marina*), eelgrass (*Zostera marina*), charophytes (*Chara* spp., *Tolypella nidifica*, *Nitella* spp.), infaunal bivalves (*Limecola balthica*, *Cerastoderma glaucum*, *Mya areanaria*), sea birds and seals. In addition, total species richness was calculated for each sampling station.

Species richness is referred to as the number of species in a given space (Magurran 2004), a sampling station in our case. Regardless of some inevitable deviations in the taxonomic resolution (see previous paragraph), the term “species richness” was still used to express the total number of taxa occurring in a site. In addition, presence and absence status of modelled species or species groups (presence if any of the included species is present) was defined for every observation point.

Bladder wrack (*F. vesiculosus*) is the main habitat-forming perennial macroalgal species in the Gulf of Finland on shallow hard substrate dominated bottoms. Bladder wrack forms habitats that are one of the most diverse within the Baltic Sea (Kautsky et al. 1992). The species provides a habitat for a number of invertebrates and juvenile fish as well as a growing surface for epiphytic species (Råberg and Kautsky 2007, Wikström and Kautsky 2007). *Fucus* grows on hard substrates in sublitoral from shallow water down to 4-5 meters depth, the depth distribution varying spatially and temporally depending on local environmental conditions. Bladder wrack populations have been declining in many areas (e.g. Archipelago Sea discussed in Vahteri and Vuorinen 2016) and the maximum growing depth has diminished remarkably (Torn et al. 2006). The reason for the distribution changes remains still somewhat unclear (see Vahteri and Vuorinen 2016 for discussion) even though eutrophication and restricted light conditions have been usually accounted for the population decline.

Furcellaria lumbricalis is a habitat-forming perennial red alga that grows on rocky substrates in the Baltic Sea. The species grows under *F. vesiculosus* canopy but forms a red algal belt in deeper parts of rocky shores often with blue mussel (*Mytilus trossulus*). In some parts of Estonian coast the species also colonizes sandy bottoms where its population is loose-lying (Martin et al. 2006). *Furcellaria lumbricalis* grows down to 15-20 meters in the Gulf of Finland.The filamentous algal group consists of a number of macroalgal species. Many of them are annual and fast-growing species (Kiirikki and Lehvo 1997). Many species benefit from eutrophication and increase in biomass in eutrophicated conditions. Filamentous algae can also out-compete other macroalgal species, e.g., by inhibiting the establishment of germlings by covering all available space or by shad-owing.

Epibenthic bivalves, blue mussel (*Mytilus trossulus*) and zebra mussel (*Dreissena polymorpha*), grow on hard substrates along the Gulf of Finland. The distribution of blue mussel is restricted to the western Gulf of Finland due to low salinity and the biomass and growth rate decrease towards less saline east (Westerbom et al. 2002). The distribution of zebra mussel on the other hand is limited to the eastern parts of Finnish coast which is caused by higher salinities in the west (Antsulevich et al. 2003). The transition zone between the two species lies between the cities Porvoo and Kotka.

In the Estonian coast, *Mytilus* is present widely while the distribution of *Dreissena* is mostly related to Pärnu Bay outside the project area but there are also some findings from the Gulf of Finland. Epibenthic bivalves provide an important food source for many birds and fish species and they are important filter feeders that can filter remarkable amounts of seawater while feeding plankton. They also produce a three dimensional habitat for other invertebrates increasing the bottom biodiversity (see e.g. Koivisto and Westerbom 2010).

The group of aquatic vascular plants (*Zostera marina* excluded) consists of a number of plant species, inhabiting the shallow coastal areas, generally down to few meters depth, due to high demand for light. Vascular plants in the Baltic Sea prefer soft or sandy bottoms in sheltered or moderately exposed areas. Species prefer different environmental conditions and submerged vascular plants can exist from river inlets containing fresh water to sheltered bays in outer archipelago. Common aquatic vascular plants in the study area are, for example, *Ceratophyllum demersum*, *Myriophyllum* spp., *Najas* spp., *Potamogeton* spp., *Ranunculus* spp., *Ruppia* spp., *Stuckenia* spp. and *Zannichellia* spp. and they form important habitats providing reproduction and nursery areas for fishes and habitats for invertebrates.

The only seagrass species inhabiting the northern Baltic Sea, eelgrass (*Zostera marina*), forms dense meadows on shallow sand bottoms. As a marine species with a salinity optimum of >10 PSU (Nejrup and Pedersen 2008), the distribution of eelgrass is limited by low salinity to the western Gulf of Finland and Southwestern Finland. Within the study area the species reproduces only vegetatively. Eutrophication related declines in depth limitation and distributional range have been recorded (Boström et al. 2014).

Charophytes, morphologically complex green algae, grow on shallow soft or sandy bottoms in brackish water. They can form dense meadows on shallow sheltered bays and flads. Some species, like *Chara aspera*, also grow on moderately exposed sandy bottoms. The genera found within the project area include *Chara* spp., *Nitella spp*., *Nitellopsis* spp. and *Tolypella nidifica*.

Infaunal bivalves (*Limecola balthica*, *Cerastoderma glaucum* and *Mya arenaria*) live on sand-dominated bottoms from shallow to deep waters. They are filter feeders that burrow into the sediment, feeding on the seabottom. The species' distribution is limited for example by suitable bottom substrate, wave energy on the bottom and oxygen conditions.

2.1.3 Abiotic environmental data

The abiotic environmental variables in this study included different bathymetrical (depth, slope of seabed, topographical position), hydrodynamic (wave exposure, currents), geological (seabed substrate), and physico-chemical (temperature, salinity, transparency, nutrients, ammonium, ice conditions) variables. Altogether 18 Estonian and 23 Finnish environmental variables (Tables 2.1.3.1 and 2.1.3.2) were used in the modelling. The resolution of the layers was 100 m in Estonia and 20 m in Finland.

Table 2.1.3.1. Georeferenced environmental variables that were used in biodiversity modeling in Estonia

Variable	**Abbreviation**	**Source**
Water depth	depth	1
Average water depth in 2000 m radius	depth2	1
Slope of seabed	slope	1
Slope of seabed in 2000 m radius	slope2	1
Salinity	salinity	2,4
Wave exposure based on simplified wave model	wave	5
Chlorophyll a content of sea surface based on satellite imagery	chl	2
Water transparency estimated as attenuation coefficient based on satellite imagery	attenuation	2
Ice coverage	ice	6
Water temperature in cold season	tempcold	3
Water temperature in warm season	tempwarm	3
Current velocity	current	3
Orbital speed of water movement at seabed induced by wind waves	orbspeed	7
Proportion of soft sediment	softsed	2
Secchi depth	secchi	2
Concentration of ammonium	ammonium	3
Concentration of nitrates	nitrate	3
Concentration of phosphates	phosphate	3

Sources:

1 – Bathymetric data by Estonian Maritime Administration

2 – Databases of the Estonian Marine Institute, University of Tartu

3 – Hydrographic model developed by the Marine Systems Institute, Tallinn University of Technology (Maljutenko and Raudsepp 2014)

4 – COHERENS ocean circulation model (Bendtsen et al. 2009)

5 – Simplified wave model based of fetch and wind data (Nikolopoulos and Isæus 2008)

6 – Finnish Meteorological Institute

7 – SWAN hydrodynamic model (Suursaar et al. 2014)

Table 2.1.3.2. Georeferenced environmental variables that were used in biodiversity modeling in Finland

Variable	Source
Bathymetric Position Index (BPI) 100x4000	1
Bathymetric Position Index (BPI) 1200x500	1
Bathymetric Position Index (BPI) 20x100	1
Bathymetric Position Index (BPI) 300x1000	1
Concentration of humic substances	1
Concentration of oxygen on the bottom	1
Concentration of phosporus on the bottom	1
Coverage of rock	1
Coverage of sand	1
Coverage of stones and boulders	1
Depth attenuated wave exposure	1
Distance to sandy shore	1
Euphotic depth	1
Maximum temperature on the bottom	1
Minimum temperature on the bottom	1
Natural habitats	2
Salinity on the bottom	1
Salinity on the surface	1
Share of the sea area (1 km radius)	1
Share of the sea area (10 km radius)	1
Share of the sea area (5 km radius)	1
Slope of seabed	1
Water depth	1

Sources:

1 – Finnish Environment Institute SYKE

2 – Geological Survey of Finland / Åbo Akademi / Parks & Wildlife Finland

2.1.4 Modeling methods

The most widely used benthic sampling devices such as grabs, trawls and underwater video or photography (Eleftheriou and McIntyre 2005) yield information only from the visited sites (point-wise data), leaving most of the study area unsampled (Herkül et al. 2013). Mathematical predictive modeling based on species–environment relationships provides a useful framework to synthesize information from scattered samples into coherent seamless maps of distributions of species and habitats, species richness, ecological goods and services (Guisan and Zimmerman 2000; Guisan and Thuiller 2005). These models are numerical methods that relate measurements of biotic variables (e.g. species occurrence or abundance, species richness) to environmental variables (Elith and Leathwick 2009). These relationships are further used to predict the distribution of values of biotic variables across different spatial and/or temporal scales (Elith and Leathwick 2009) (Figure 2.1.4.1).

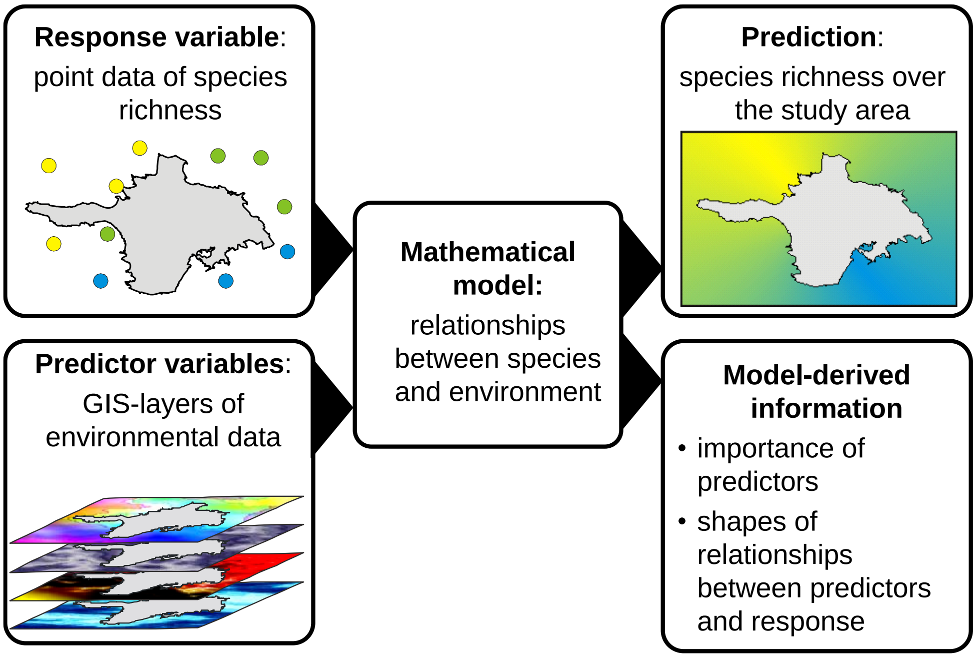

Figure 2.1.4.1. Conceptual scheme of spatial predictive modeling for deriving spatially continuous data from point-wise biological sampling data.

The spatial distributions of benthic species, species groups and biodiversity variables were modeled. The key species and species groups, chosen as important ecosystem components, were *Fucus vesiculosus*, *Furcellaria lumbricalis*, filamentous algae, epibenthic bivalves (*Mytilus trossulus*, *Dreissena polymorpha*), vascular plants (excluding *Zostera marina*), *Zostera marina*, Charophytes (*Chara* spp., *Tolypella nidifica*), infaunal bivalves (*Limecola balthica*, *Cerastoderma glaucum*, *Mya areanaria*), sea birds and seals. Due to low data availability, birds were not modelled in Finland and important seal areas were only identified using seal protection areas (Finland: Finnish government decree 376/2001; Estonia: EELIS (Estonian Natura Information System) – Estonian Environmental Register: Estonian Environment Agency) as proxy variable as no georeferenced field observations were available.

Several candidate models were built for each biodiversity variable using boosted regression trees (BRT). The candidate model with the best predictive performance was chosen to produce the final distribution maps. BRT is an ensemble method that combines the strength of two algorithms: regression trees and boosting (Elith et al. 2008). Regression trees are good at selecting relevant predictor variables and can model interactions. Boosting enables building of a large number of trees in a way that each successive tree adds small modifications in parts of the model space to fit the data better (Friedman et al. 2000). The algorithm keeps adding trees until finding the optimal number of trees that minimizes the predictive deviance of a model. The predictive performance of BRT has been shown to be superior to most other modeling methods (Elith et al. 2006; Revermann et al. 2012). Important parameters in building BRT models are learning rate, tree complexity, and bag fraction (Elith et al. 2008). Learning rate determines the contribution of each tree to the growing model and tree complexity defines the depth of interactions allowed in a model. Bag fraction determines the proportion of data to be selected randomly at each iteration. Different combinations of these parameters may yield variable predictive performance but generally a lower learning rate and inclusion of interactions gives better results. For each group of species richness predictions, BRT models with tree complexity of 5 in Estonia and 7 or 9 in Finland were built. For example, a tree complexity of 5 fits a model with up to five-way interactions. The learning rates of Estonian and Finnish models were set to 0.005 and 0.01, respectively, and the bag fraction was set at 0.5 which is the recommended default value for presence-absence models (Elith et al. 2008). Modeling was done in the statistical software R 3.3.1 (R Core Team 2016) using the packages *gbm* (Ridgeway 2007) and *dismo* (Elith and Leathwick 2017) for BRT. Finnish benthic biodiversity was calculated by combining invertebrate and macrophyte species richnesses modelled from taxa observed within every sampling station (invertebrates) or 200 m wide grid cell (macrophytes) using random forest in R utilizing package *randomForest* (Liaw and Wiener 2002).

The input data was randomly partitioned into calibration and validation datasets (85 % and 15 % in Estonia and 80 % and 20 % in Finland). The validation dataset contained data that was not included in model calibration. Calibrated models were used to predict the species richness spatially with a grid size of 100 meters covering the whole Estonian sea area from the coastline to the outer border of the exclusive economic zone. In Finland, the spatial prediction was calculated comparably for the whole sea area using a grid size of 20 meters. Values of species richness and probabilities of occurrence for every modelled species or group were predicted for each cell and the modeling outputs were converted to raster layers in ESRI ArcGIS 10.2.1 or in R using raster package. Raster layers of predictions were visually assessed to identify possible overfitting and other model- or data-driven artifacts that may not be directly reflected in mathematical validation. Based on both mathematical validation and visual expert assessment, the best performing modeling algorithm was selected. Importance of environmental predictor variables was assessed using percentage relative influence in BRT.

2.2 Birds

The results of the aerial mapping and modeling study by Luigujõe and Auniņš 2016 on the distribution of wintering water birds in Estonia was used as an input in this study. The affiliations of this work are shown in the Acknowledgements.

Two bird species groups were used for the study. Marine bird species obtaining food (molluscs, crustacean, insects, aquatic macrophytes, algae, etc.) from the sea bottom are referred to as benthos feeders. Fish feeders contain bird species feeding mainly on fish (Luigujõe and Auniņš 2016). Benthic feeders are mainly in sea areas less than 30 m deep, fish eating birds less than 50 m deep, as the diving depth of birds is limited (Luigujõe and Auniņš 2016).

The aggregating seabirds in Estonian marine areas dataset, which was used in our study, based on aerial surveys. Considering, that areas deeper than 50 m are not suitable feeding areas for benthic feeders or fish feeders, counting transects were planned up to the 50 m depth contour line. Final size of the monitored area was 22000 km^2, which is about 60% of the total Estonian sea surface area (Figure 2.2.1). The used counting method based on internationally recommended standards (Pihl and Frikke 1992; Camphuysen et al. 2004) and their later modifications (Fox et al. 2006). To provide the most precise data for further bird distribution modelling, transects were placed every 3 km, which is the minimum distance for the used methodology. In deep areas transects were placed in every 6 km.

The original model showed abundance of bird species/groups individuals per one km^2 (Luigujõe and Auniņš 2016). For our final birds' data layer we summed together bird species/groups layers from the original model of wintering birds (benthos feeders; fish feeders, gulls and swans).

Final summed wintering bird's data layer in the vulnerability profile included:

- Benthos feeders (*Clangula hyemalis*; *Bucephala clangula*; *Somateria mollissima*; *Polysticta stelleri*; *Melanitta nigra*; *Melanitta fusca*; *Aythya fuligula; Aythya marila*)
- Fish feeders (*Gavia* sp.; *Gavia stellata*; *Mergus serrator*; *Mergus merganser*; *Mergus albellus*; *Phalacrocorax carbo*; *Alca torda*)
- Gulls and swans (*Larus* sp.; *Larus minutus*; *Larus canus*; *Larus argentatus*; *Larus canus*/*Larus argentatus*; *Cygnus* sp.)

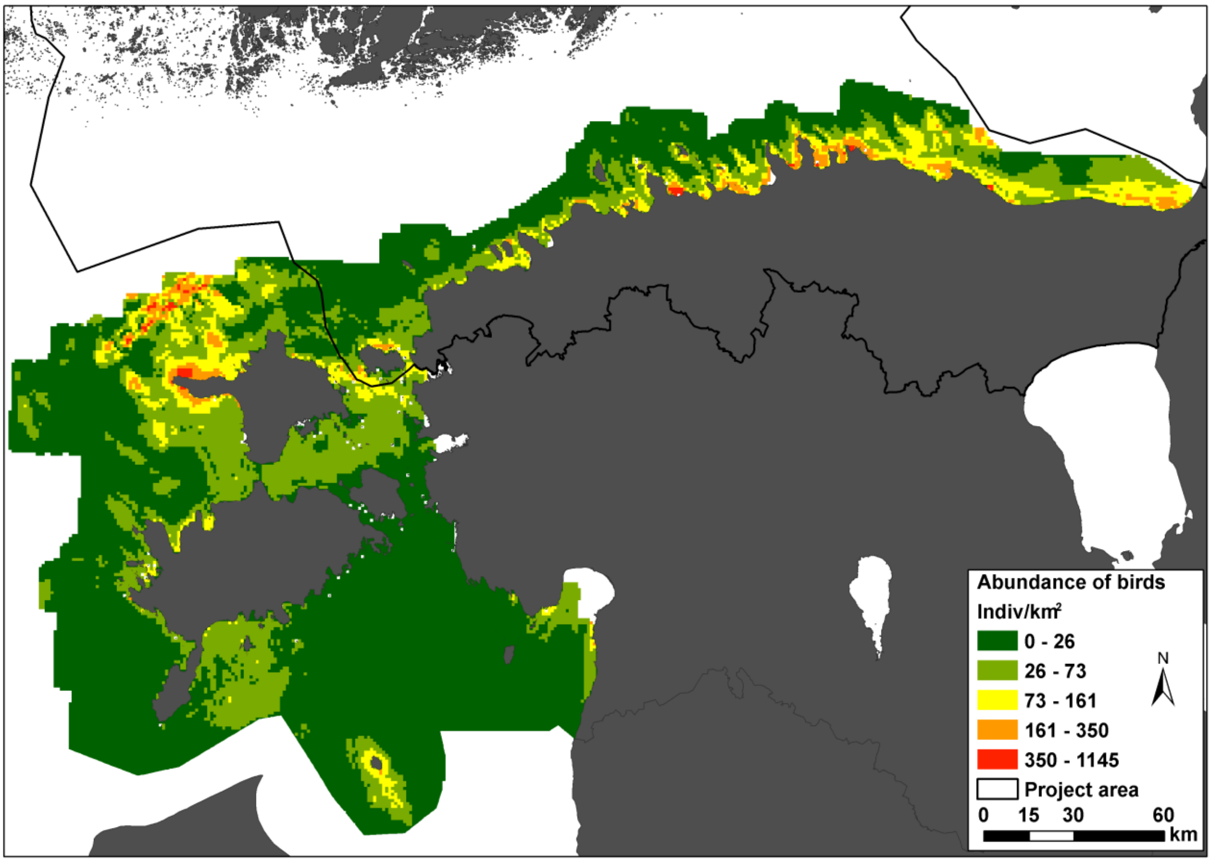

Figure 2.2.1. Total abundance of wintering birds based on aerial survey and modeling study by Luigujõe and Auniņš (2016) that was used as an input in the current study.

Birds were predicted to be present almost everywhere on shallow areas, with particularly high abundances close to the coast, islands and on offshore shallows (Figure 2.2.1). One exceptional area, where high abundances were related to deeper areas, was located northwest from Hiiumaa Island. Those high abundances were due to high abundance of gulls registered in this area.

2.3 Seals

Due to the lack and poor access of spatial data of seals, polygons of seal protection areas for both the Estonian and Finnish areas were used as input data for analysis (Figure 2.3.1; the affiliations of seal data are shown in the Acknowledgements). The polygons indicated nationally protected moulting, resting or breeding areas of seals. If new high quality distribution data of seals become available during the project, then the results will be recalculated using the new data.

Both, the Finnish and Estonian seal areas are mainly located around the islands or on offshore shallows (Figure 2.3.1). Most of the Estonian seals areas are in the Väinameri, around Hiiumaa and Vormsi islands, and only a few of them are located in the Gulf of Finland. In Finland, the seal protection areas are mainly located in the Gulf of Finland.

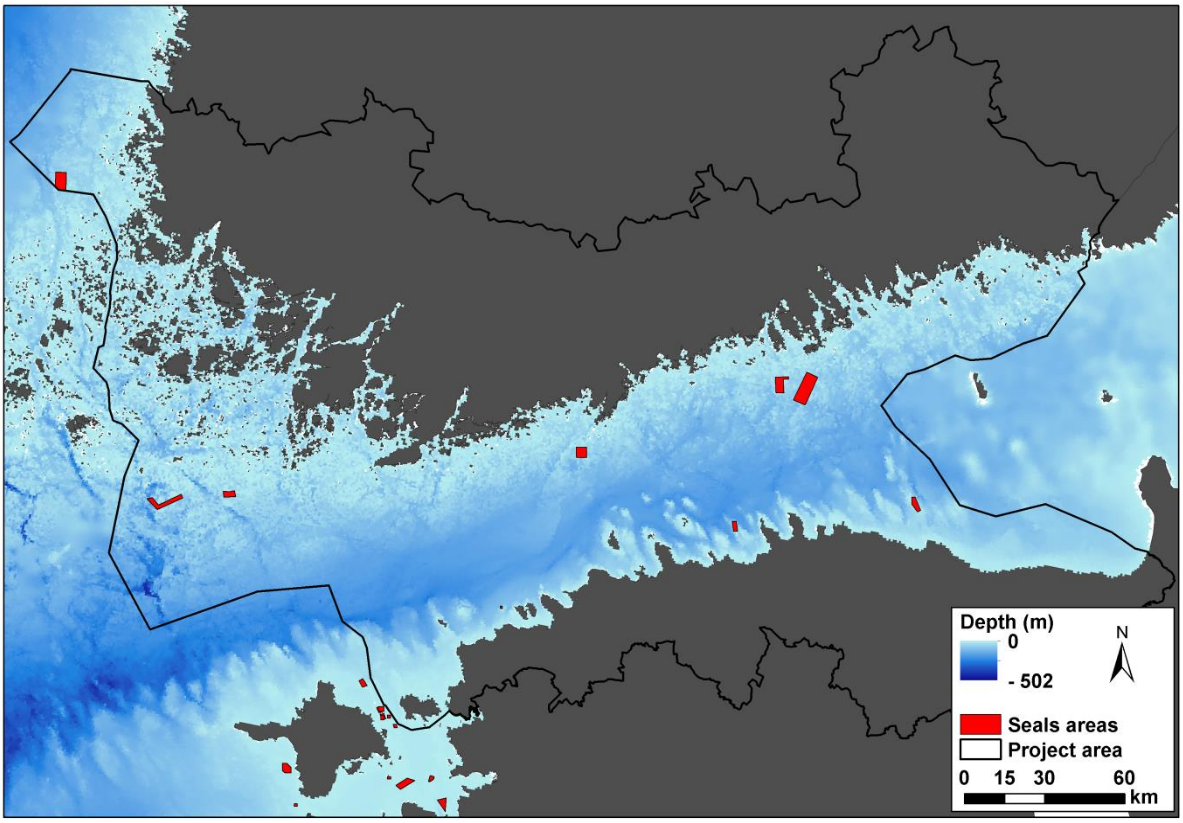

Figure 2.3.1. Nationally protected moulting, resting, and breeding areas of seals.

2.4 Sensitivity of nature values

Assessing pressure specific sensitivity is very challenging because different pressures impact the marine environment simultaneously. Furthermore, the magnitude of this simultaneous cumulative impact is a function of a complicated set of different environmental variables like salinity, depth, hydrodynamic activity etc. which all can vary spatially and temporally on a specific site. There is a lack of such empirical knowledge to quantitatively formalize species sensitivity as functions of environmental variables. A practical approach to this complex problem can be the use of the recovery potential of an environmental value that is measured in time that is needed to recover from a destruction after an impact has ceased. For example, a reefs habitat type with ephemeral algae would recover (given that the geological structure is still present) within one year or a growing season because the spores of ephemeral algal species are produced in abundance and disperse and colonize efficiently available habitats. However, the recovery of a nature value (NV), example reefs habitat type with bladder wrack community would require 2-3 years. A ringed seal population would need more than 10 years to recover. Although the exact time needed for recovery is difficult to estimate and depends on the prevailing environmental conditions, generalizations based on known biological parameters of species or taxonomic groups can be utilized as a basis for analyses.

The recovery estimations are based on literature, combining species relevant life history traits and observed events of recovery or (re)colonization in the Baltic Sea and/or areas similar to Gulf of Finland. We divided the nature values into 5 classes according to their recovery potential, i.e. time needed for recovery, to provide optimal differentiation between rapidly recovering filamentous algal species, slower recovering perennial algal species, benthic fauna and vascular plants and very slowly recovering vertebrates. In case of multiple species within a nature value we collected information on as many species as possible (or relevant within the nature value) and assigned recovery class considering the slower recovering species. However, if only one or a few species within a nature value re-establishes slowly and the habitat functionality is restored quickly by other species within the nature value, the recovery class is assigned based on the recovery of majority of the species (recovery of habitat functionality).

2.5 Calculation of environmental vulnerability profile

Marine and coastal environmental vulnerability profile is covering the marine open sea area as well as the coastal shallow sea area of the Gulf of Finland.

- Environmental vulnerability profile (EVP): a georeferenced data layer that incorporates the distribution of all nature values and their sensitivities to disturbances

The spatial analyses were based on the European Environmental Agency's 1 km rectangular grid (http://www.eea.europa.eu/data-and-maps/data/eea-reference-grids), i.e., the 1 km × 1 km cells were the units of calculation. In each cell mean and maximum values of the following nature values were calculated using the modeled GIS layers that were produced in the previous step:

- bladder wrack (*Fucus vesiculosus*),
- the perennial red seaweed *Furcellaria lumbricalis*,
- filamentous algae,
- epibenthic bivalves (*Mytilus trossulus*, *Dreissena polymorpha*),
- vascular plants (excluding *Zostera marina*),
- eelgrass (*Zostera marina*),
- charophytes (*Chara* spp., *Tolypella nidifica*, *Nitella* spp.),
- infaunal bivalves (*Limecola balthica*, *Cerastoderma glaucum*, *Mya areanaria*),
- benthic biodiversity (summed total species richness of macrophytes and macroinvertebrates),
- wintering sea birds.

In addition, the presence (value 1) or absence (value 0) of a seal protection area in cells were assigned.

Different versions of EVP were calculated based on either mean or maximum values. The mean- and maximum-based products were assessed by Finnish and Estonian project team members and the maximum-based versions were selected to be used in the project as they (1) emphasized the maximum nature values in grid cells (i.e. the precautionary principle not to mask the presence of high nature values by using mean values), (2) more clearly revealed the differences between marine areas.

EVP is a standardized and weighed aggregation of all the nature values that were used in this study (see Table 3.2.1). The general scheme of calculations is shown in figure (Figure 2.5.2). The calculation of EVP included several steps all of which were proceeded in each 1 km grid cell:

1. Transformation of benthic biodiversity

   1.1 Benthic biodiversity was natural logarithm transformed to reduce the variation: ln (biodiversity + 1)

   1.2 The logarithmed biodiversity was divided by the maximum logarithmed biodiversity value over all cells to make the values vary between 0 and 1: ln(biodiversity + 1) /max(ln(biodiversity + 1))

   1.3 1 was added to the product of the previous step to eliminate zero-values which would render further multiplication operations to zero: ln(biodiversity + 1) /max(ln(biodiversity + 1)) + 1

2. Weighing and aggregation of benthic NVs

   2.1 Excluding benthic biodiversity, each benthic NV was multiplied by its respective sensitivity coefficient (Table 3.2.1): $NV_i \times Sensitivity_i$

2.2 All NV_i × $Sensitivity_i$ values were averaged

2.3 The averaged NV_i × $Sensitivity_i$ product of the previous step was multiplied by the transformed biodiversity (point 1.3)

3. Weighing of bird and seal NVs: bird and seal grid values were multiplied by their respective sensitivity coefficients (Table 3.2.1)

4. Aggregation of all NVs to calculate the value of EVP

4.1 The values from weighed and aggregated benthic NVs, birds, and seals were averaged

4.2 The product of the previous step was divided by its maximum value over all cells to make the values vary between 0 and 1

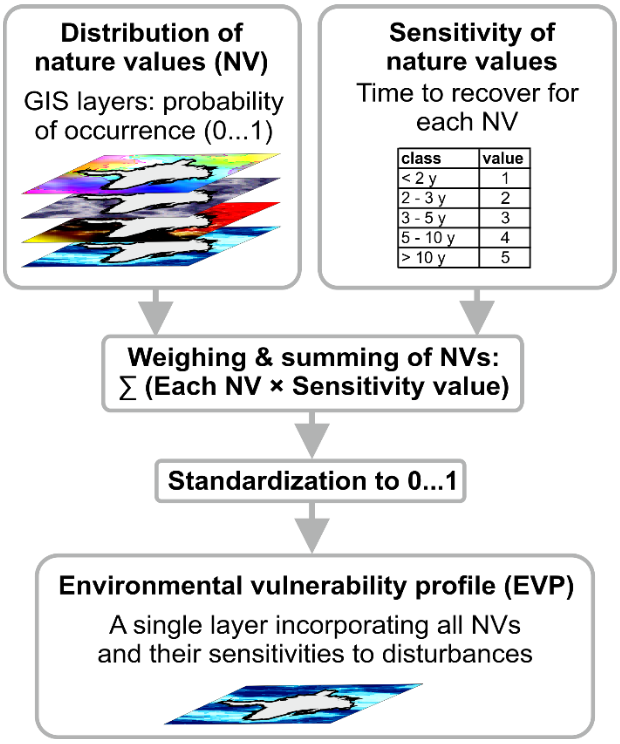

Figure 2.5.2. The general scheme of calculations of environmental vulnerability profile (EVP).

Due to data limitations of birds and seals data, the following separate layers of EVP were produced:

- EVP-F included all input data (i.e. benthos, birds, and seals). That layer was produced for Estonian area only where the bird data was available.
- EVP-BS included benthos and seal data.
- EVP-B included only benthos data.
  It is important to add that these are the suggested main products that cover the whole study area at the current state of the project (October 2017) as:
  - the suitable bird data is available only for the Estonian sea area;
  - the use of seal polygons in calculating EVP is discouraged because the polygons represent nature protection areas not the actual density or probability of occurrence of seal in the nature.

3 Results

The locations of the toponyms that were used in describing the results are shown in the Figure 3.1

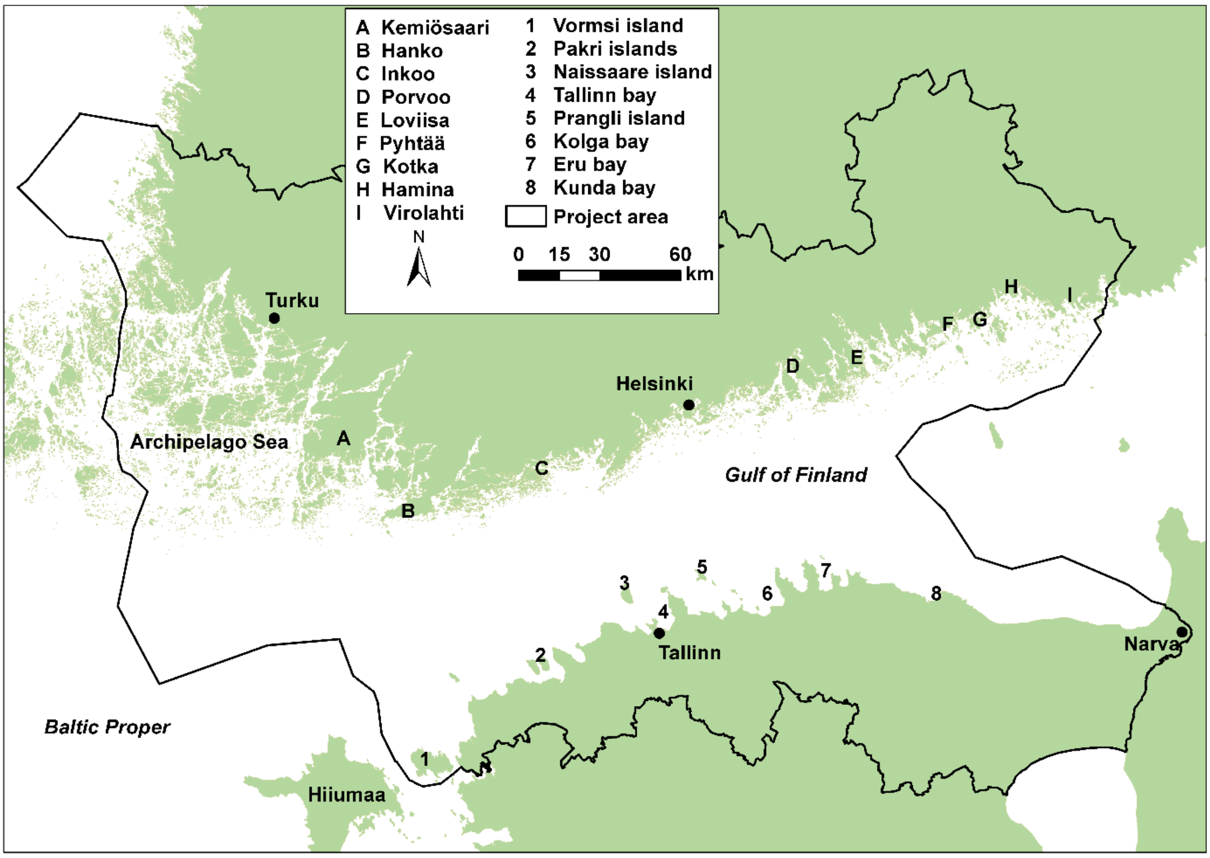

Figure 3.1. Locations of the toponyms that were used in describing the results.

3.1 Modeled distribution of benthic nature values

High biodiversity was related to shallow marine areas. The highest predicted total species richness within the Estonian project area was found around the islands of western Estonia (Figure 3.1.1). Diversity was high also on the exposed coast especially around the peninsulas. Total species richness decreased towards deep areas being the lowest on the deepest parts (> 75 m) of the Gulf of Finland that are largely devoid of macrobenthos due to permanent hypoxia. The eastern coast also showed signs of lower species richness compared to other parts of Estonian coast. Water depth, proportion of soft sediment and water temperature in warm season had the highest influence in the model prediction.

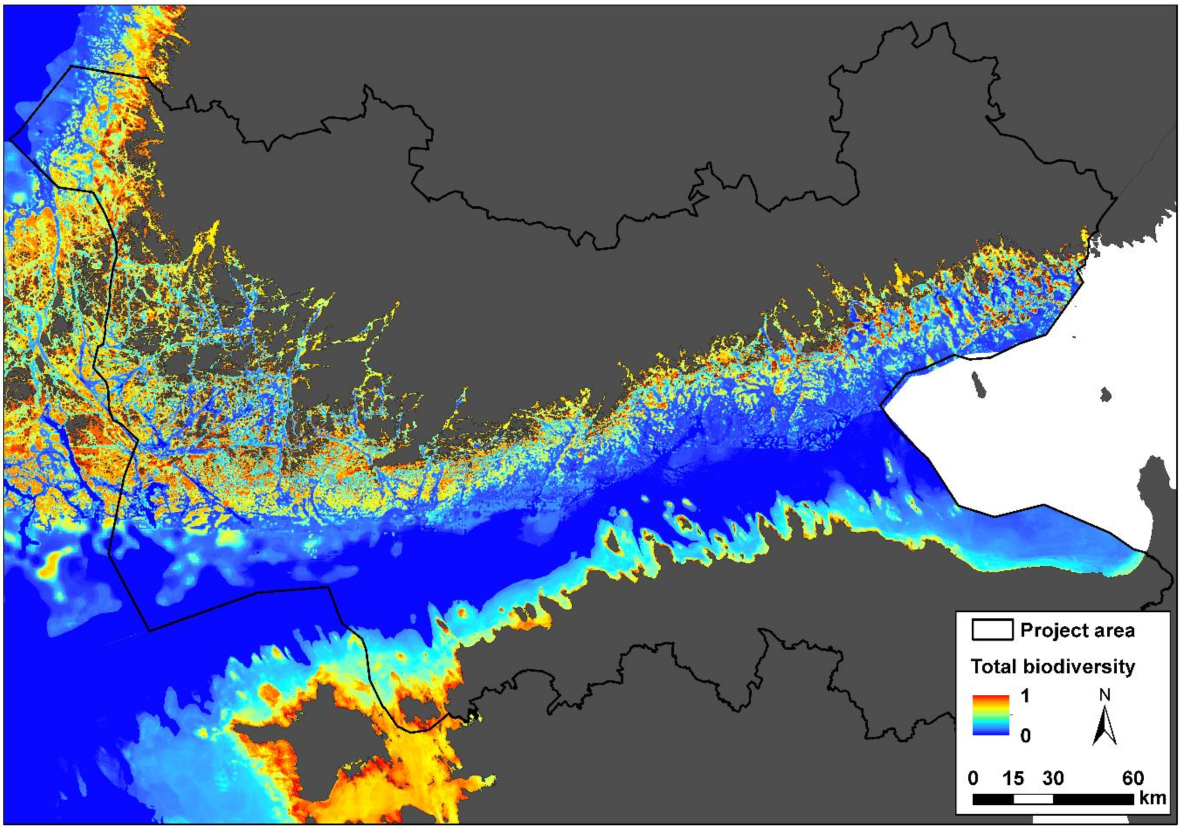

Figure 3.1.1. The total logarithmed standardized biodiversity across the project area. Biodiversity values vary between 0 and 1, where 1 is the highest species richness.

Fucus vesiculosus was predicted to be present almost everywhere in outer and middle archipelago and the exposed coast of southern Gulf of Finland (Figure 3.1.4). The distribution area was a narrow band following shoreline and shallow areas and the distribution did not usually cover areas deeper than some 4-6 meters. The distribution was limited in the sheltered bays where the low salinities and high sedimentation are most likely to prevent the presence of the species.

On Estonian coast, some sheltered bays were predicted to have low probabilities of occurrence while the highest probabilities were located on the exposed coast and around islands. In Finland, the middle and outer archipelago were in general potential *Fucus* areas but there were some areas of lower probability, like the middle part of the Archipelago Sea.

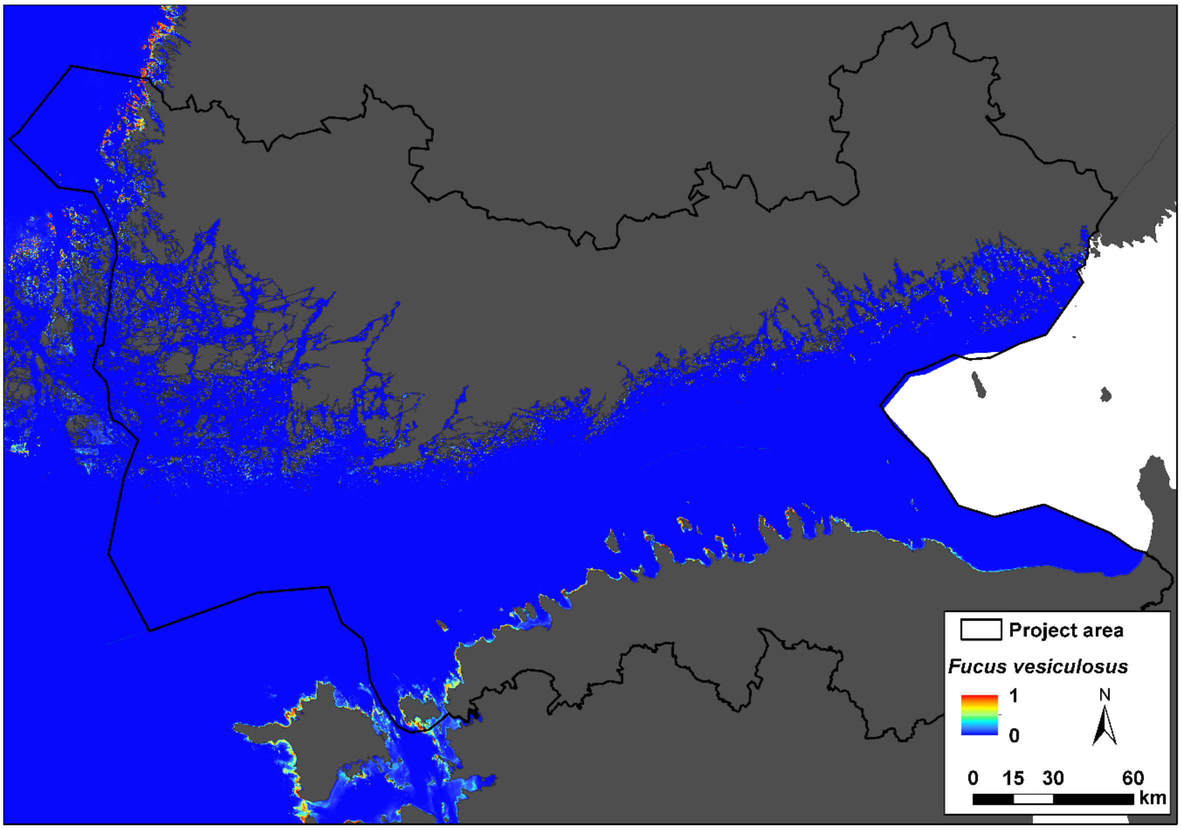

Figure 3.1.4. Distribution of *Fucus vesiculosus* across the project area. Distribution values vary from 0 to 1, where the value 1 states the highest probability of occurrence.

The distribution of *Furcellaria lumbricalis* was generally limited to exposed areas in outer archipelago. The species grows usually deeper or at the same depths than other macroalgal species which is the reason why its distribution area follows the shoreline and reef areas but its most probable distribution area was located a short distance away from the shore. In Estonia, the highest probabilities of occurrence were located north of Vormsi Island and around the exposed islands and tips of peninsulas (Figure 3.1.5). There was also high predicted probability for the species' occurrence around small islands in the Gulf of Finland. In contrast eastern part of Estonia showed less probable distribution areas for the species. In Finland the species' distribution was at its widest in the northwest parts of the region of Southwest Finland (Figure 3.1.5). Large potential distribution area was also located in the southern Archipelago Sea facing the Baltic Proper. Inside the archipelago, especially in the inner archipelago the probability of occurrence was generally low. There was a visible trend of decline in the distribution area towards the eastern part of Gulf of Finland and the species was predicted to be present only in outer archipelago within rather small areas compared to similar areas in the west.

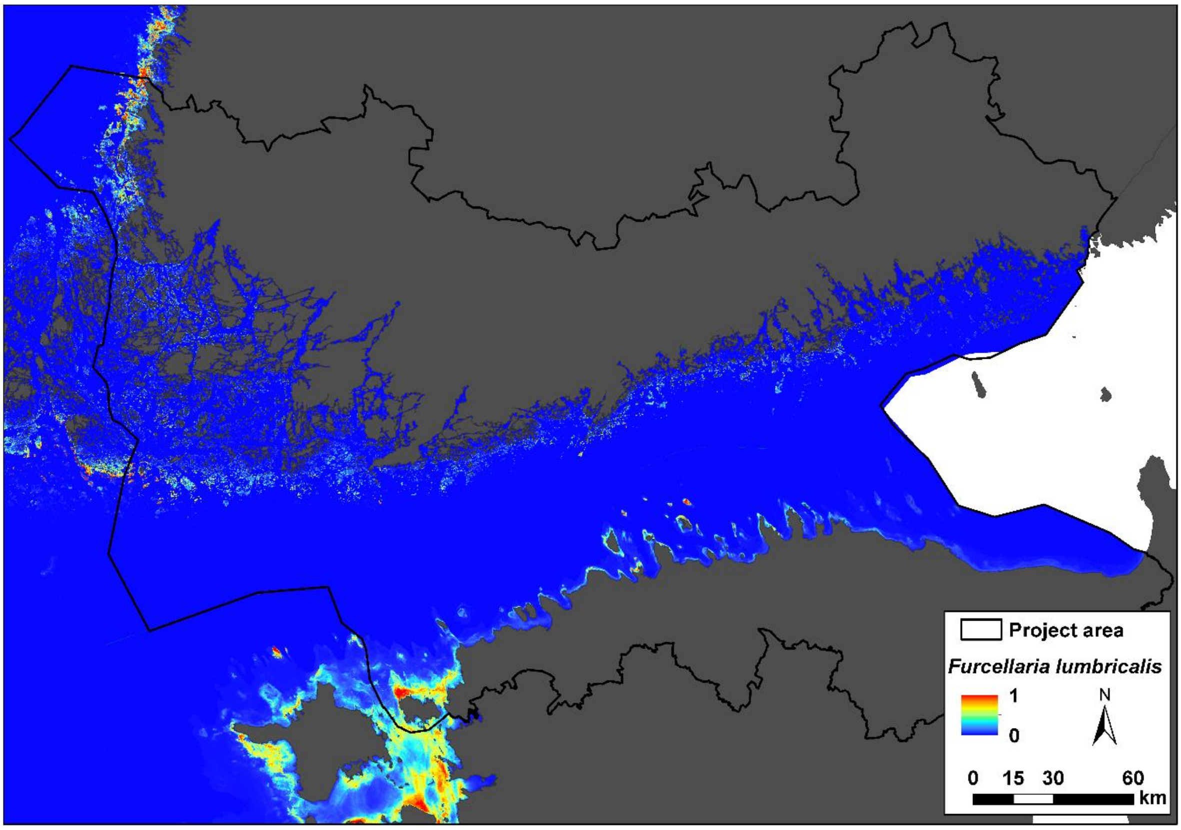

Figure 3.1.5. Distribution of *Furcellaria lumbricalis* across the project area. Distribution values vary from 0 to 1, where the value 1 states the highest probability of occurrence.

Filamentous algae were predicted to be present almost everywhere on shallow areas (Figure 3.1.6). The probability of occurrence was the highest in the outer archipelago and decreases towards sheltered inner archipelago in Finland. In Estonia, the highest probabilities for filamentous algal growth were around Vormsi and other islands and the outer parts of peninsulas. In Finland, the large areas of probable distribution areas were located in the outer archipelago, especially in the northern part of southwestern Finland and the archipelago facing the Baltic Proper and the Gulf of Finland open sea.

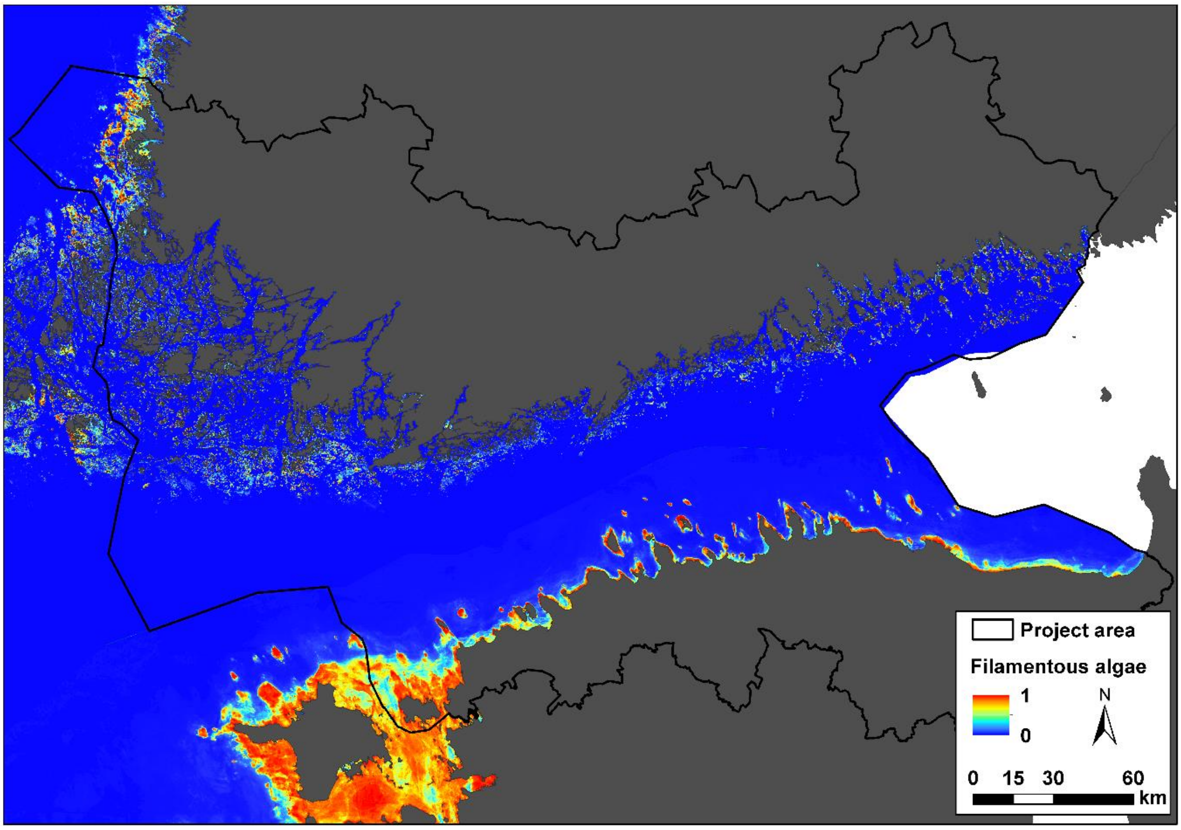

Figure 3.1.6. Distribution of filamentous algae across the project area. Distribution values vary from 0 to 1, where the value 1 states the highest probability of occurrence.

Epibenthic bivalves were predicted to cover large areas of sea bottom (Figure 3.1.7). The two species modelled and combined as a group were *Mytilus trossulus* and *Dreissena polymorpha*. As a marine species, *Mytilus trossulus*' distribution area covered large areas in the middle and outer archipelago. The distribution was limited by low salinity in the eastern Gulf of Finland and in less saline inner bays where the river inflow decreases the salinity of the sea water. In Estonia, the species was predicted to be present mostly in exposed areas north of Vormsi and around other islands (Figure 3.1.7). Close to the Estonian mainland the probabilities of occurrence were generally lower and the sheltered bays were not favorable for the species. In the eastern part of Estonia, the probabilities of occurrence were moderate. In Finland, the predicted distribution area of *Mytilus trossulus* covered wide areas in the middle and outer archipelago of Southwestern Finland but did not generally reach inner archipelago (Figure 3.1.7). The low salinity in the easternmost parts of Gulf of Finland restricts the distribution of *M. trossulus* east of Loviisa. The distribution of *Dreissena polymorpha* was very limited in the Estonian parts of the project area. The Finnish part of the eastern Gulf of Finland, from rather sheltered areas to the outer archipelago was predicted to host *Dreissena* populations. The distribution area reaches from Virolahti to Pyhtää where the high salinities start to limit the distribution. There was a transition from *Mytilus trossulus* to *Dreissena polymorpha* between Loviisa and Pyhtää and the distributions of the two species did not overlap in general.

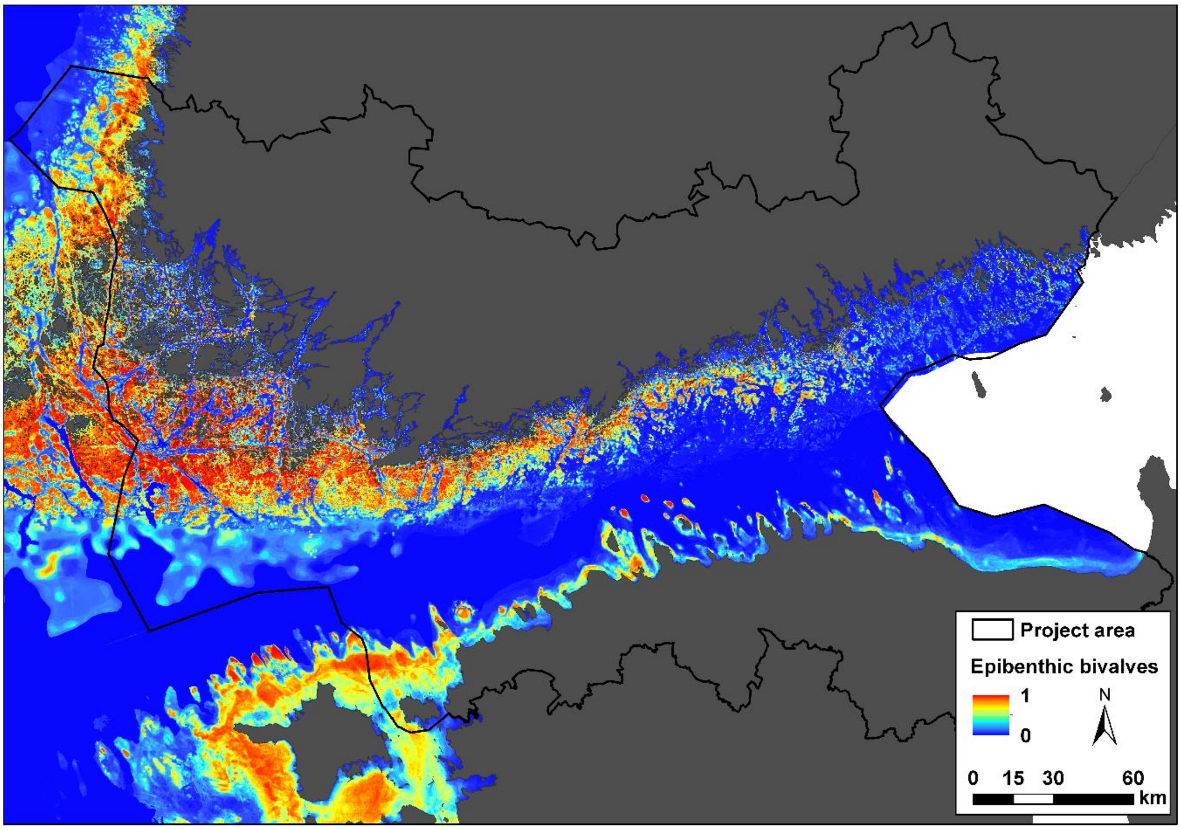

Figure 3.1.7. Distribution of epibenthic bivalves across the project area. Distribution values vary from 0 to 1, where the value 1 states the highest probability of occurrence.

High probabilities of occurrence of vascular plants (*Zostera marina* excluded from the group) were clearly related to shallow areas close to shore especially in sheltered or moderately exposed areas. The probabilities varied between locations and high probabilities can be found from sheltered inner archipelago to more exposed sites in outer archipelago. Moderately exposed areas in middle archipelago had also high probabilities of occurrence. In Estonia, large potential distribution areas were located south of Vormsi, south of Pakri islands and in Eru Bay (Figure 3.1.8). The vascular plants were predicted to be present practically everywhere along the Finnish coast inside the project area (Figure 3.1.8). In outer archipelago the probabilities were lower on the exposed side of the islands and around small skerries compared to the sheltered sides of larges islands.

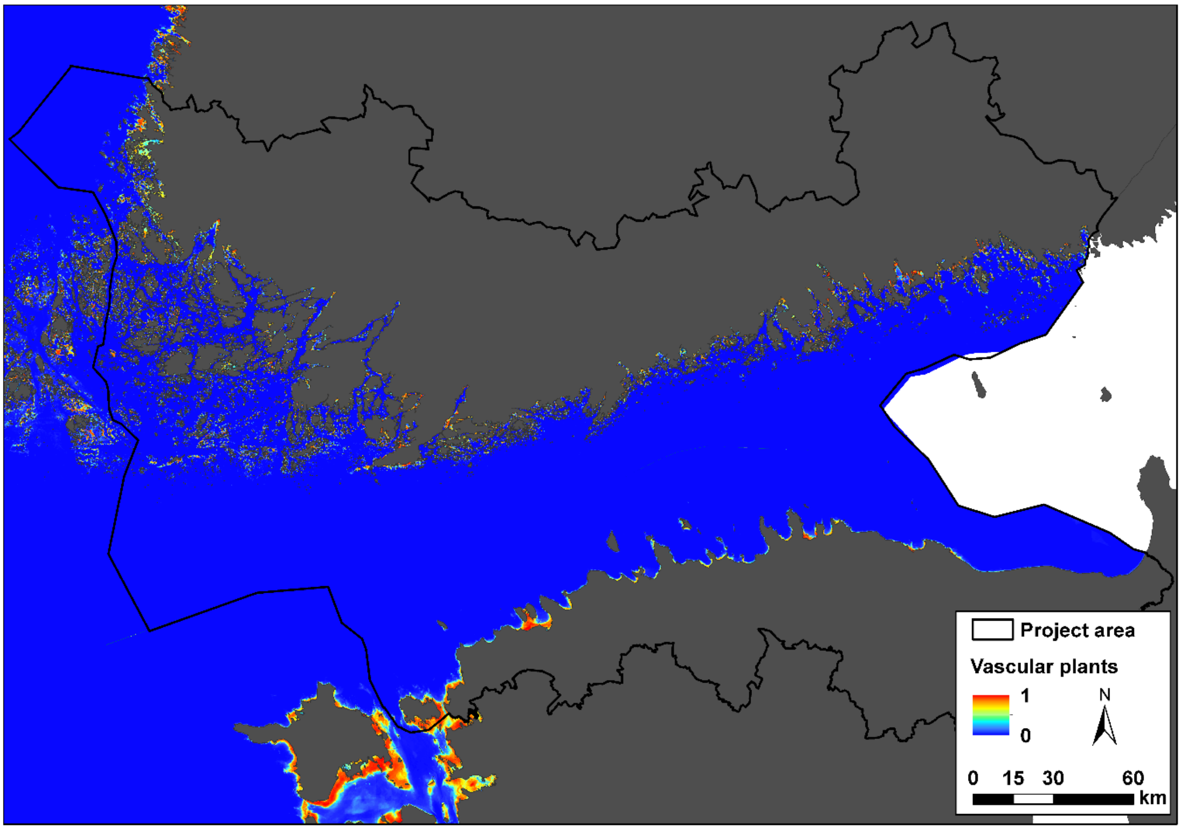

Figure 3.1.8. Distribution of vascular plants across the project area. Distribution values vary from 0 to 1, where the value 1 states the highest probability of occurrence.

Zostera marina had a rather narrow distribution compared to other modelled species (Figure 3.1.9). The distribution of the eelgrass was limited mainly to the western Gulf of Finland and to the Archipelago Sea due to the species' demand for high salinity. In Estonia the moderately exposed bays had moderate or low probabilities of occurrence from Vormsi to Kunda bay (Figure 3.1.9). The eastern parts of Estonia were not predicted to be suitable for *Zostera marina,* probably due to high wave exposure and low salinity. In Finland, the largest predicted suitable areas for the species were around Hanko Peninsula and in the southern Archipelago Sea (Figure 3.1.9). There were some small suitable areas for the species across the Archipelago Sea but the distribution was not predicted to reach inner archipelago in general. The distribution in the Gulf of Finland was predicted to be low which was most likely due the salinity gradient. There were some small probable distribution areas with rather low probabilities between Hanko and Porvoo.

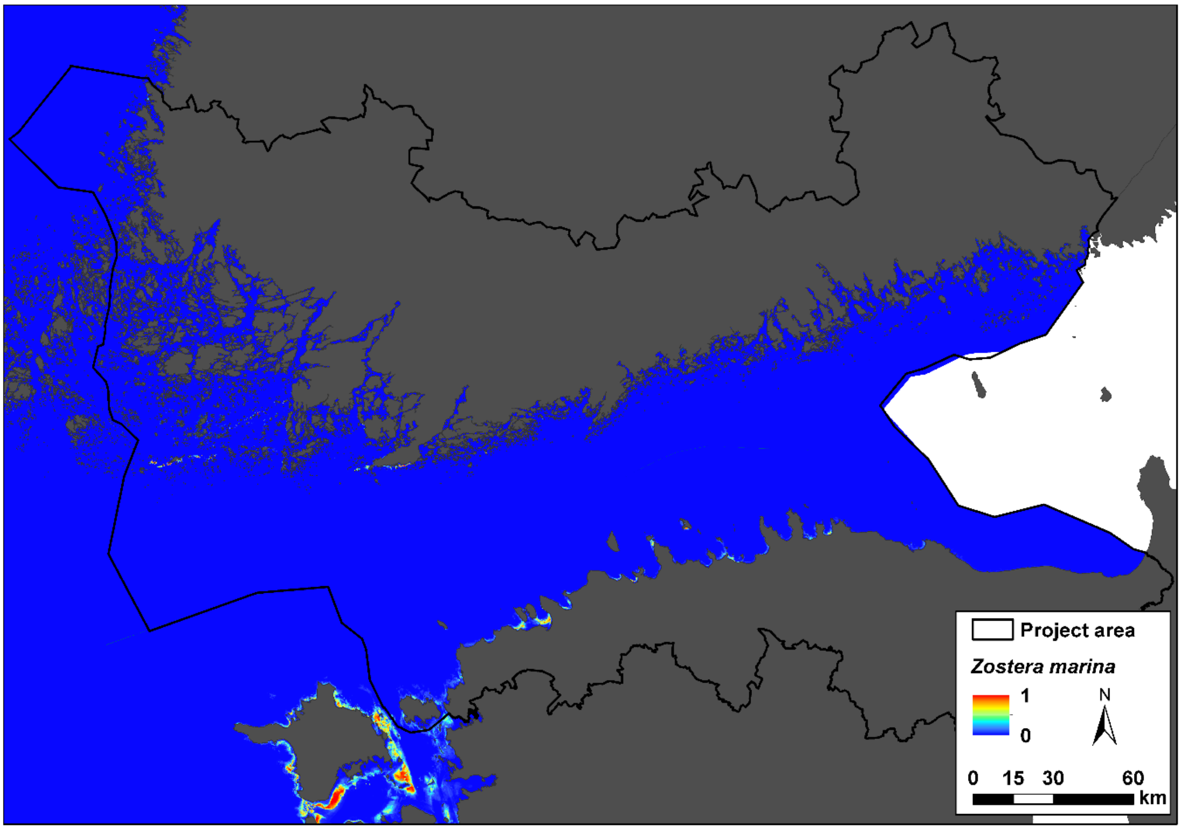

Figure 3.1.9. Distribution of *Zostera marina* across the project area. Distribution values vary from 0 to 1, where the value 1 states the highest probability of occurrence.

The distribution of charophytes was generally limited to shallow water of middle and inner archipelago (Figure 3.1.10). The highest probabilities of occurrence were related to sheltered areas. On the Estonian side of the project area the largest potential distribution areas of charophytes were situated in the bays of south Vormsi and around Pakri islands (Figure 3.1.10). In Finland, the distribution of charophytes was rather scattered around the archipelago (Figure 3.1.10). In the Archipelago Sea, the probabilities of occurrence were generally moderate across the middle archipelago while the outer archipelago did not have extensive suitable areas for charophytes. There were some distinctive patterns in the predicted distribution: the probable distribution areas were located in middle and inner archipelago in general but in the eastern Gulf of Finland the sheltered bays had high probabilities of occurrence compared to other shallow areas in Gulf of Finland. These bays were example sheltered bays around Kotka. In the western Gulf of Finland some of the suitable areas included for example southern and norteastern part of Hanko peninsula.

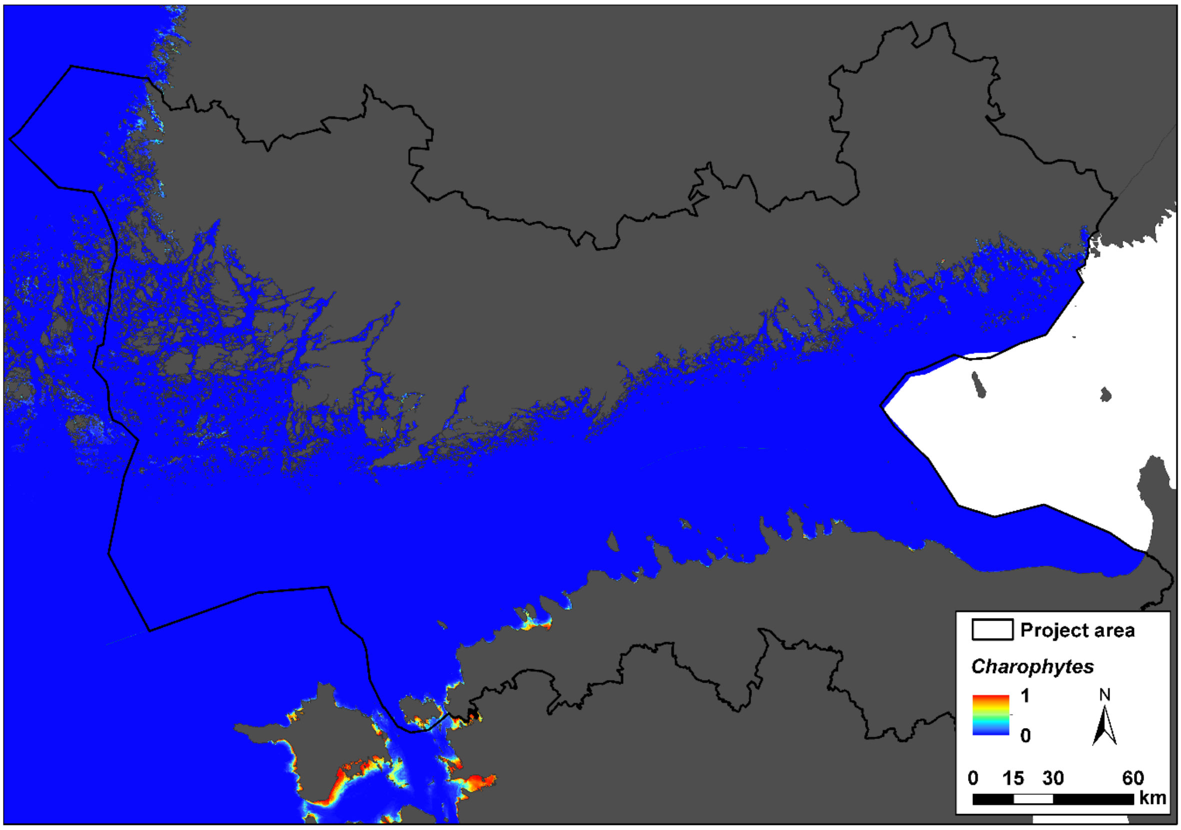

Figure 3.1.10. Distribution of charophytes across the project area. Distribution values vary from 0 to 1, where the value 1 states the highest probability of occurrence.

The distribution of infaunal bivalves was predicted to be wide. Both the Finnish and Estonian coasts were predicted to host bivalves almost everywhere in the project area except for the deepest parts of open sea (Figure 3.1.11). In Estonia, the distribution covered the coastal waters along the coast excluding the very shallow and exposed areas in the eastern Gulf of Finland and deep areas in the open sea. In Finland, the predicted distribution area was from the inner or middle archipelago to the open sea leaving most of the inner bays empty from infaunal bivalves.

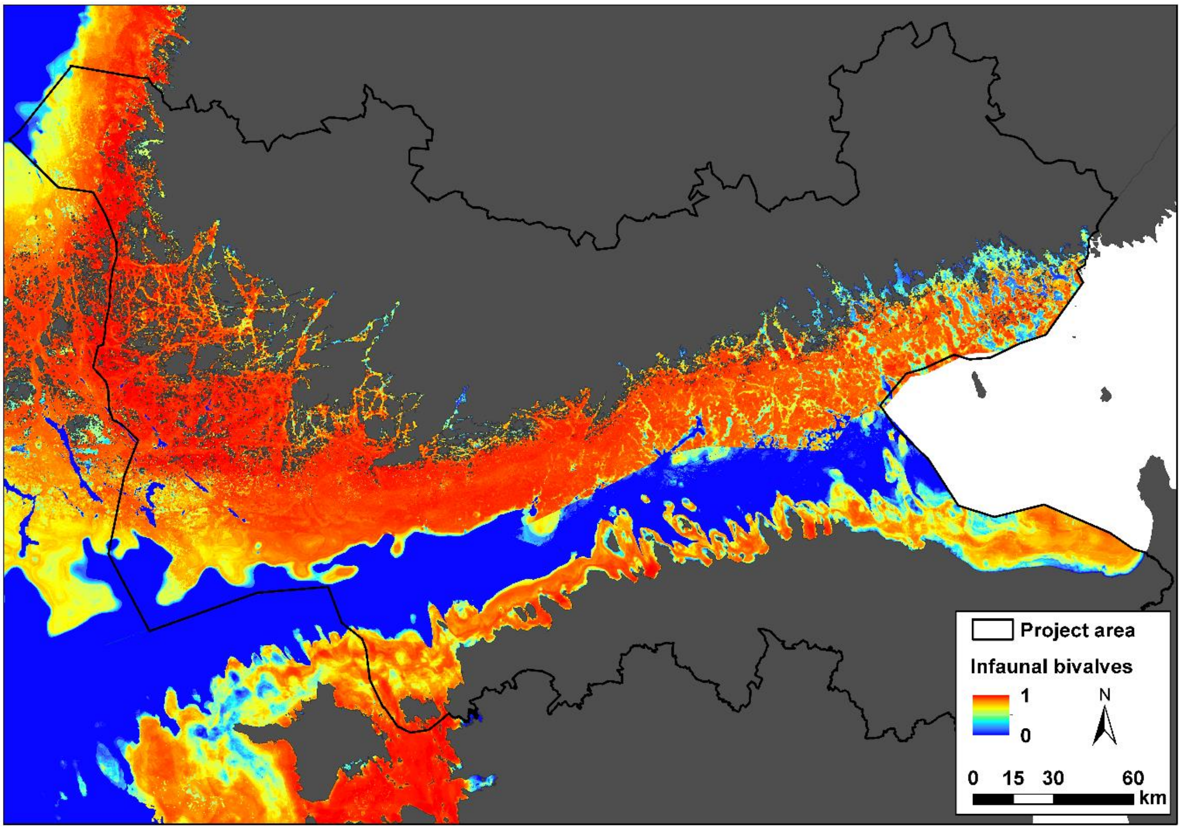

Figure 3.1.11. Distribution of infaunal bivalves across the project area. Distribution values vary from 0 to 1, where the value 1 states the highest probability of occurrence.

3.2 Sensitivity of nature values

The recovery classes and the respective sensitivity coefficients of NVs together with the rationale and references are shown in Table 3.2.1.

A recovery class <2 years (and respectively coefficient 1 for calculation) was assigned to filamentous algae as this group recovers most rapidly after a total loss.

Fucus vesiculosus, charophytes and infaunal bivalves were assigned to recovery class 2-3 years as the recolonization and re-establishment occurs within a few years, given that the suitable physical environment is present. These groups contain species that might need longer recovery time, however other species within the same group that have shorter recovery time can perform the functionality.

Vascular plants (excl. *Zostera marina*) and epibenthic bivalves were assigned to recovery class 3-5 years as the recolonisation may not succeed every year and re-establishment (reaching biomasses necessary for providing habitat functionality) may take several years.

Furcellaria lumbricalis were assigned to recovery class 5-10 years due to its slow growth rate (Bird et al. 1979; Martin et al. 2006), long time to reach maturity (Austin 1960), small effective dispersal distance, and mainly vegetative reproduction (Kostamo and Mäkinen 2006).

The highest coefficient (5) was assigned to *Zostera marina*, birds and seals. *Zostera marina* reproduces in the northern Baltic Sea mainly vegetatively (Olsen et al. 2004) and grows slowly (Boström et al. 2014). Therefore recovery, depending only on vegetative dispersal, will be extremely slow. While birds and seals have the ability, at least to some extent, to move away if their habitat becomes unsuitable (e.g. due to disturbance) and recolonize the area after the pressure has disappeared, they are long-lived organisms with low fecundity and thus can only slowly increase their overall abundance in the whole study area. Loss of only a few habitats (e.g. some nesting, feeding or breeding areas) or disturbances that renders these habitats unsuitable will easily result in population decline in the whole Gulf of Finland (and adjacent areas).

Table 3.2.1. The recovery classes and the respective sensitivity coefficients of NVs together with a short explanation and reference.

Species/group	Recovery class (years)	Coefficient in calculations	Rationale, references
Fucus vesiculosus	2-3	2	As northern Baltic Sea littoral habitats are well connected (Rothäusler et al. 2015), recolonization will possibly occur during the next reproduction period (after complete removal, if conditions for growth are adequate and substrate present), but gamete dispersal is rather limited and can slow recovery (Serrão et al. 1999). Reaching to canopy state prior to removal, takes time. It took about 2 years (in Nova Scotia) after ice scouring completely eliminated fucoid assemblages (incl. *F. vesiculosus*) to re-colonise and fully recover (to canopy state similar to pre-scouring; recolonisation took less than 7 months; Minchinton et al. 1997).
Furcellaria lumbricalis	5-10	4	Recovers slowly due to low growth rate (Bird et al. 1979; Martin et al. 2006), long time to reach maturity (5 years in Wales; Austin 1960) and recruitment that usually occurs in the vicinity of parent plant (Rayment 2008). In addition, due to low salinity in the Gulf of Finland, vegetative reproduction prevails (Kostamo and Mäkinen 2006) rendering the recovery even slower.
Filamentous algae	< 2	1	After ice scouring that completely removed algae, it took less than few months to establish filamentous algae cover on rocks (including, among others: *Pilayella littoralis, Polysiphonia sp., Ectocarpus sp., Ceramium virgatum, Cladophora sp.*), in Nova Scotia (Minchinton et al. 1997). Artificial substrates became colonised by *Pilayella littoralis* in 3 months during winter and within one month in spring (Kraufvelin et al. 2007). Also other filamentous algae colonized artificial substrates within a year (during the first spring/summer; Kraufvelin et al. 2007).
Epibenthic bivalves (*Mytilus trossulus, Dreissena polymorpha*)	3-5	3	After ice scouring that completely removed bivalves, *Mytilus* spp. recolonised substrates in about 1-1.5 years; distribution similar to pre-scouring was observed within 5 years (Nova Scotia; Minchinton et al. 1997). While growth rates are lower in the Gulf of Finland than in optimal conditions (Kautsky 1982a), recruitment is possible all year round in the Baltic Sea (Kautsky 1982b). *D. polymorpha*: high fecundity, good dispersal abilities and rather fast growth rates (Mackie et al. 1989) support fast recovery. Becomes sexually mature in second year of life (Mackie et al. 1989). However, recovery may be slowed in the Gulf of Finland by limited or low spawning/recruitment in

			years with cold summers (Orlova and Panov 2004).
Vascular plants (excl. *Zostera marina*)	3-5	3	*Ruppia maritima* (and *Najas marina*) – annual species with short life cycle; high production rate (Kautsky 1988) and high seed production (Silberhorn et al. 1996) indicates fast recovery. *Stuckenia pectinata* was successfully established in habitats created 3-5 years ago (Boedeltje et al. 2001). In Lake Balaton vascular plants (that occur also in the Baltic Sea/ Gulf of Finland) colonised rapidly de-vegetated areas via rhizomes, fragments of plants etc (i.e. vegetative reproduction) from adjacent vegetated areas (Vári and Tóth 2017), also in a Danish lake submerged macrophytes recolonised the lake within 5 years (up to 90% coverage; Lauridsen et al. 1994). *Zannichellia palustris* have a wide-ranging generative recolonization potential (Steinhardt and Selig 2007). The dispersal and recolonization of aquatic plants and charophytes are encouraged by local propagule banks (Steinhardt and Selig 2007) and waterbirds transporting the seeds (mostly in their guts) especially on local scale (Green et al. 2002).
Zostera marina	> 10	5	Recolonization after total loss (i.e. no seed bank) can be extremely difficult (Holt et al. 1995). If there is adequate seed bank in sediments: recolonisation (after anoxic event) was observed during next summer, but seedling mortality is huge (~99%), so recovery takes definitely several years (Greve et al. 2005). In France it took less than 9 months till biomasses similar to pre-destruction and 2 years till flowering (Plus et al. 2003), but here, in colder Gulf of Finland eelgrass grows slower (Boström et al. 2014). In the northern Baltic Sea *Z. marina* commonly reproduces vegetatively (Olsen et al. 2004), thus there is no seed bank in sediments. Therefore, re-establishment will be very slow due to very limited vegetative dispersal (Holt et al. 1995) and possibly impoverished gene bank (Boström et al. 2014), and lack of suitable genotypes (as hypothesized for lower zones in Wadden Sea: van Katwijk et al. 2000). No recovery was observed two years post-dredging in New England (seed bank removed together with sediment, but *Z. marina* growing in the area; Sabol et al. 2005).
Charophytes (*Chara* spp, *Tolypella nidifica*)	2-3	2	*Chara vulgaris* was successfully established in newly created habitats in less than 3 years (Boedeltje et al. 2001). If there is a sufficient oospore bank in sediment, it may greatly enhance recolonisation (C. *aspera* in shallow lake; Van den Berg et al. 2001). However, charophyte recovery (by biomass) can take more than 2 years (Torn et al. 2010). *C. tomentosa* has not recolonised Byviken in Hanko, southwestern Finland since the dredging that took place many years ago.
Infaunal bivalves (*Limecola balthica*, *Cerastoderma*	2-3	2	In defaunated areas, infaunal bivalves (incl. *Limecola balthica, Mya arenaria* and *Cerastoderma edule*) biomass recovery takes several years (Beukema et al. 1999), but abundance reached to similar values as in undisturbed areas within 1 year (at least 1 summer needed) or even faster (juvenile

glaucum, *Mya areanaria*)			abundance; Van Colen et al. 2008).
Seals	> 10	5	Long-lived organisms with low fecundity and late reproductive maturity (compared to other organisms in the table).
Birds	> 10	5	Long-lived organisms with low fecundity and late reproductive maturity (compared to other organisms in the table).

3.3 Environmental vulnerability profile (EVP)

Three different versions of the environmental vulnerability profile (EVP) were produced. At this stage of the project, the main result among the versions of EVP is EVP-B that consists only of benthic species. Other two profiles produced are preliminary and they lack Finnish bird data or consist of seal data that was not obtained from the field observations and thus concerned only "proxy data". In profile EVP-BS also seals were included and the most comprehensive layer EVP-F consists of benthic, bird and seal data.

3.3.1 EVP-B

EVP-B, that was based only on the benthic nature values, had the highest values in Jurmo in the Archipelago Sea (EVP-B=1) and south of Vuosaari in Helsinki (EVP-B=0.99) (Figure 3.3.1.1). There were some areas in the Archipelago Sea where the values were above 0.9, especially southwest of Kemiönsaari, and areas above 0.8 are more pronounced. Larger areas hosting values above 0.8 can also be found south and west of Hanko Peninsula. Within the Estonian side of project area, there were no values above 0.80 that was located southwest of Vormsi Island (Figure 3.3.1.1). Values above 0.7 were found around Vormsi, Pakri and other smaller islands and in Tallinn area in Estonia and around the Archipelago Sea and southern Bothnian Sea, around Hanko peninsula and in some locations in the outer archipelago in Gulf of Finland on the Finnish side of the project area. Values above 0.6 were found especially in the Southwestern Finland and in the Archipelago Sea in middle and outer archipelago, around Hanko peninsula and in outer archipelago from Hanko to Pyhtää and south of Kotka and Hamina in the outer archipelago. In Estonia, values above 0.5 were less prominent but were sparsely located from Vormsi to Lahemaa.

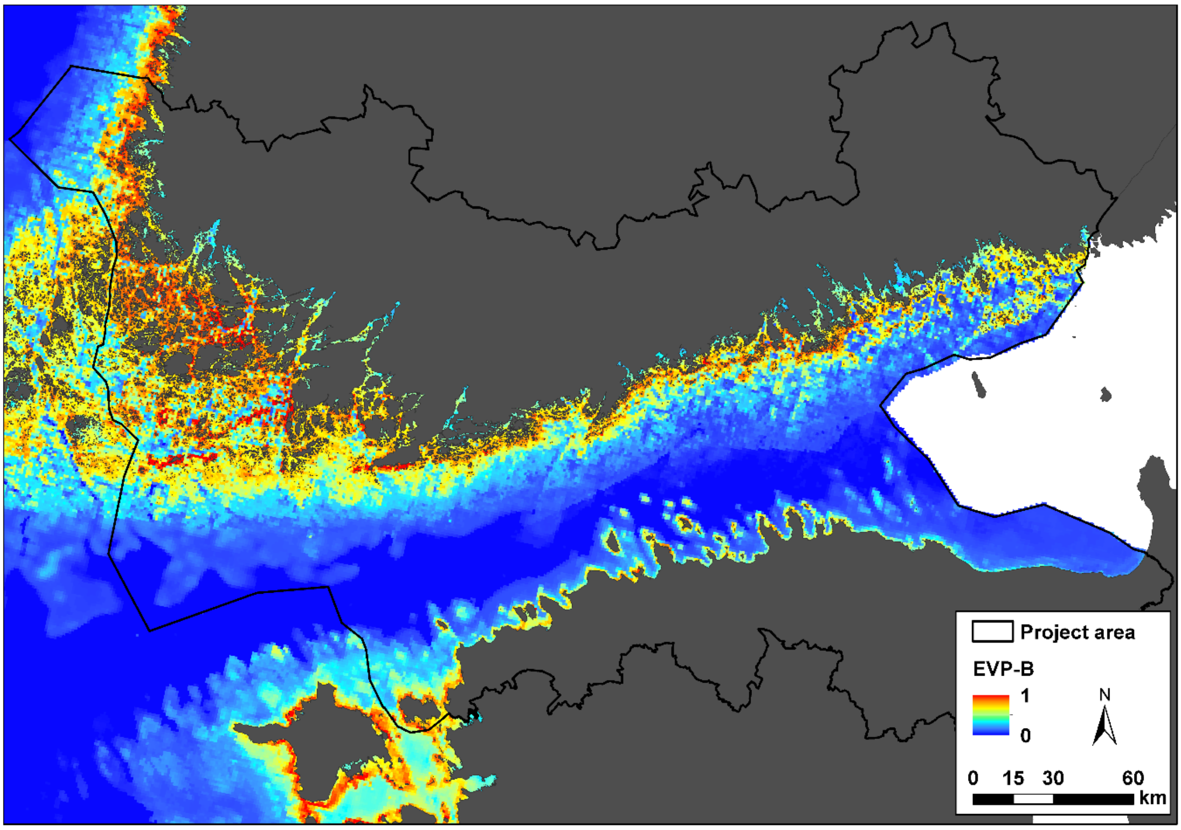

Figure 3.3.1.1. Environmental vulnerability profile based on benthic nature values (EVP). Values vary between 0 and 1, where 1 expresses the highest vulnerability.

3.3.2 EVP-BS

Environmental vulnerability profile including benthic groups and seals but no birds (EVP-BS) had the highest value (EVP-BS=1) on the border of the project area, southwest of Vormsi Island in Estonia and south of Loviisa in Finland (Figure 3.3.2.1). Values between 0.8 and 0.9 were found around Kolga Lahe in Estonia and in the outer archipelago south of Porvoo, Inkoo and the Archipelago Sea in Finland. Values above 0.6 included, in addition to the aforementioned areas, the areas around seal protection areas in Estonia and important seals areas in Finland. Many places in the Archipelago Sea and Bothnian Sea and the peninsula around Hanko had values 0.4-0.5 while in Estonia values this high could be found outside seal protection areas only in south of Pakri and Vormsi islands. Values under 0.4 were more common along the coastline, especially in outer archipelago and in Finland. Deep open sea areas had values under 0.1 in general.

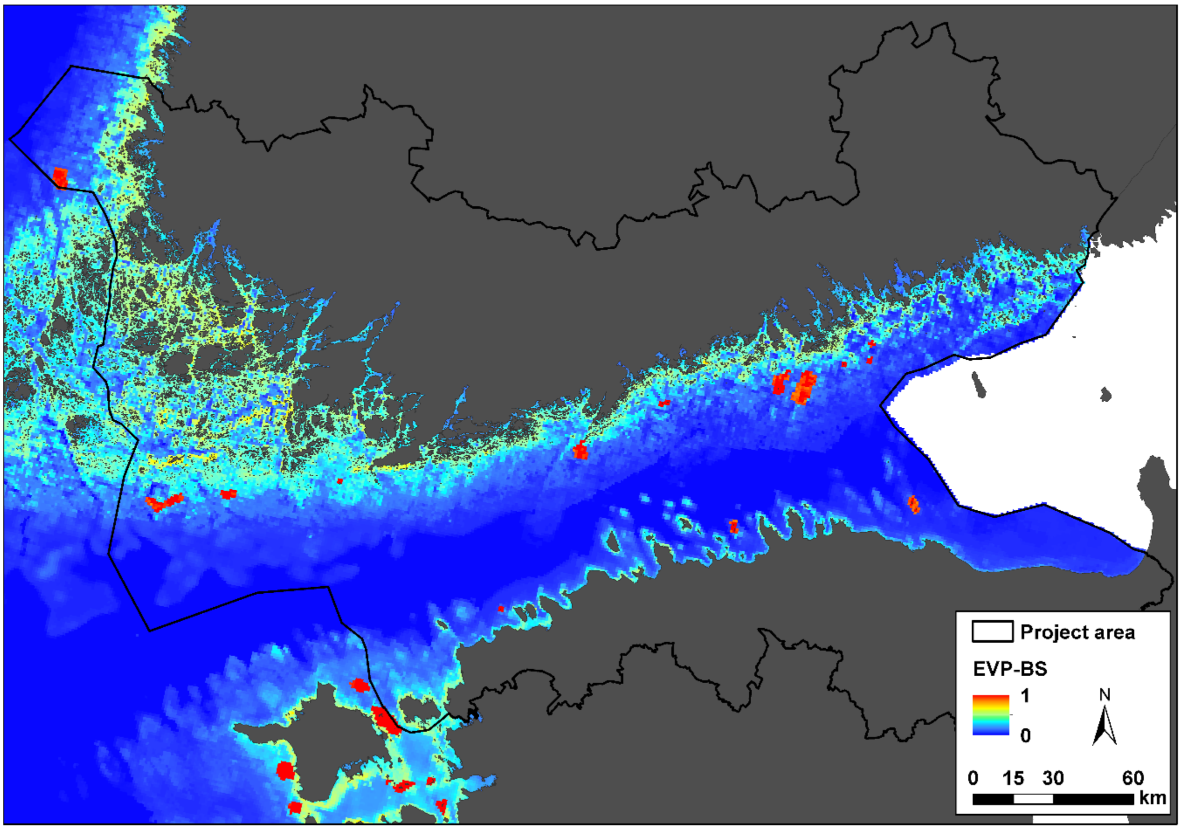

Figure 3.3.2.1. Environmental vulnerability profile, based on benthic nature values and seals (EVP-BS). Values vary between 0 and 1, where 1 expresses the highest vulnerability.

3.3.3 EVP-F

Environmental vulnerability profile including benthic groups, seals and birds (EVP-F) was only applicable in Estonia, where the highest values were related to the seal protection areas (Figure 3.3.3.1). Considering the project areas, the proportions of very high vulnerability values (> 0.75) were generally higher in the western and central area than in the eastern area. Areas of high vulnerability (other than those of seal protection areas) were related to shallow and topographically complex areas with numerous peninsulas, bay, and islands. Values 0.4-0.5 were common near the coastline except the eastern part of Estonia. Values above 0.3 covered most of the coastline while most of the deep open sea had values of 0.1-0.2.

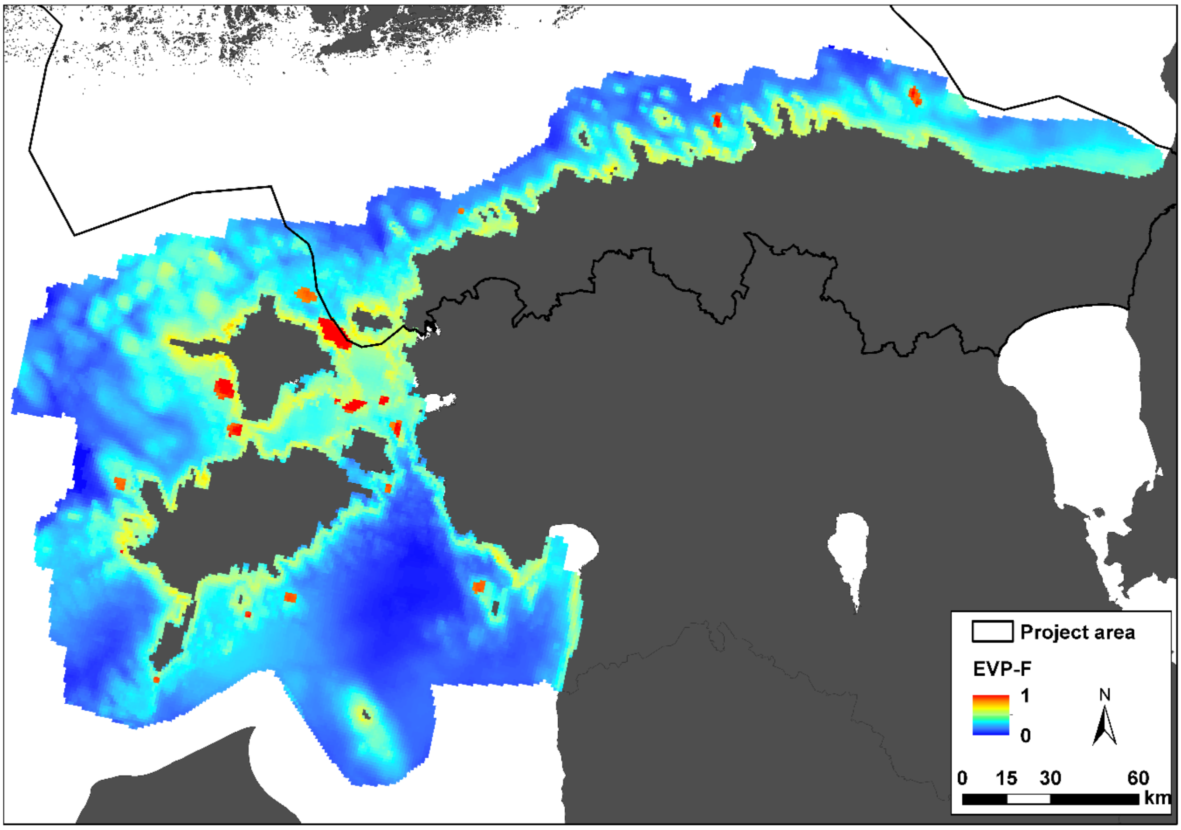

Figure 3.3.3.1. Environmental vulnerability profile based on benthic nature values, seals and birds (EVP-F). Values vary between 0 and 1, where 1 expresses the highest vulnerability.

4 Executive summary

Human use of marine and coastal areas is increasing worldwide, resulting in conflicts between different interests for the space and resources and environmental sustainability. The marine environment is increasingly stressed by human activities, especially in intensively used sea areas such as the Baltic Sea, where the competing interests and human pressures extend over national borders. To successfully support blue growth, while also preserving the capacity of ecosystems to provide valued services, marine spatial planning (MSP) processes are in a need of spatial data on nature values and human pressures to minimize the potential harm on ecosystem.

The aim of this study was to develop cross-border environmental vulnerability profile of the Gulf of Finland, which can be used for ecosystem based MSP processes in Estonia and Finland, in order to find solutions that lead to sustainable use of resources and to improved planning and management of the marine and coastal areas. The main product of this report was:

- Environmental vulnerability profile (EVP) – a spatial data layer that incorporates the distribution of nature values and their sensitivities to disturbances; higher value indicates a presence of more sensitive nature values;

The distribution of the following nature values were included in the calculation of EVP:

- Key seabed flora and fauna: bladder wrack, red seaweed *Furcellaria lumbricalis*, filamentous algae, epibenthic bivalves, infaunal bivalves, vascular plants, charophytes;
- Species richness of seabed flora and fauna;
- Water birds;
- Seals.

The general scheme of deriving EVP is shown in Figure 4.1.

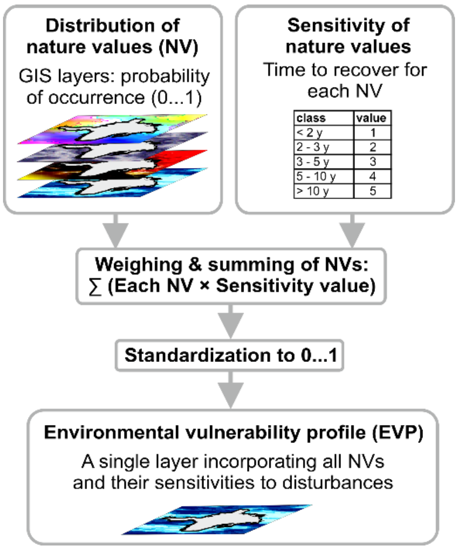

Figure 4.1. The general scheme of calculations of environmental vulnerability profile (EVP.

Due to data limitations of birds and seals data, the following separate layers of EVP were produced:

- EVP-F included all input data (i.e. benthos, birds, seals). That layer was produced for only Estonian area where the suitable bird data was available.
- EVP-BS included benthos and seal data.
- EVP-B included only benthos data. EVP-B is the suggested main product that cover the whole study area at the current state of the project (October 2017) as:
    - suitable bird data is available only for the Estonian sea area;

- use of seal polygons in calculating EVP is discouraged because the polygons represent nature protection areas not the actual density or probability of occurrence of seal in the nature.

If new high quality distribution data of seals (Finland and Estonia) and bird (Finland) become available during the project, then the results will be recalculated using the new data.

High vulnerability (EVP-B) was related to shallow areas in medium wave exposure (Figure 4.2). Generally, the vulnerability decreased towards deeper sea areas. The highest values of EVP-B were found near Jurmo and west of Kemiönsaari in the Archipelago Sea and south of Helsinki in the Finnish part of the project area. In the Estonian area, larger areas of high vulnerability were situated in the western study area (around Vormsi and Pakri islands) and around the peninsulas of the central area of the Gulf of Finland (Figure 4.2).

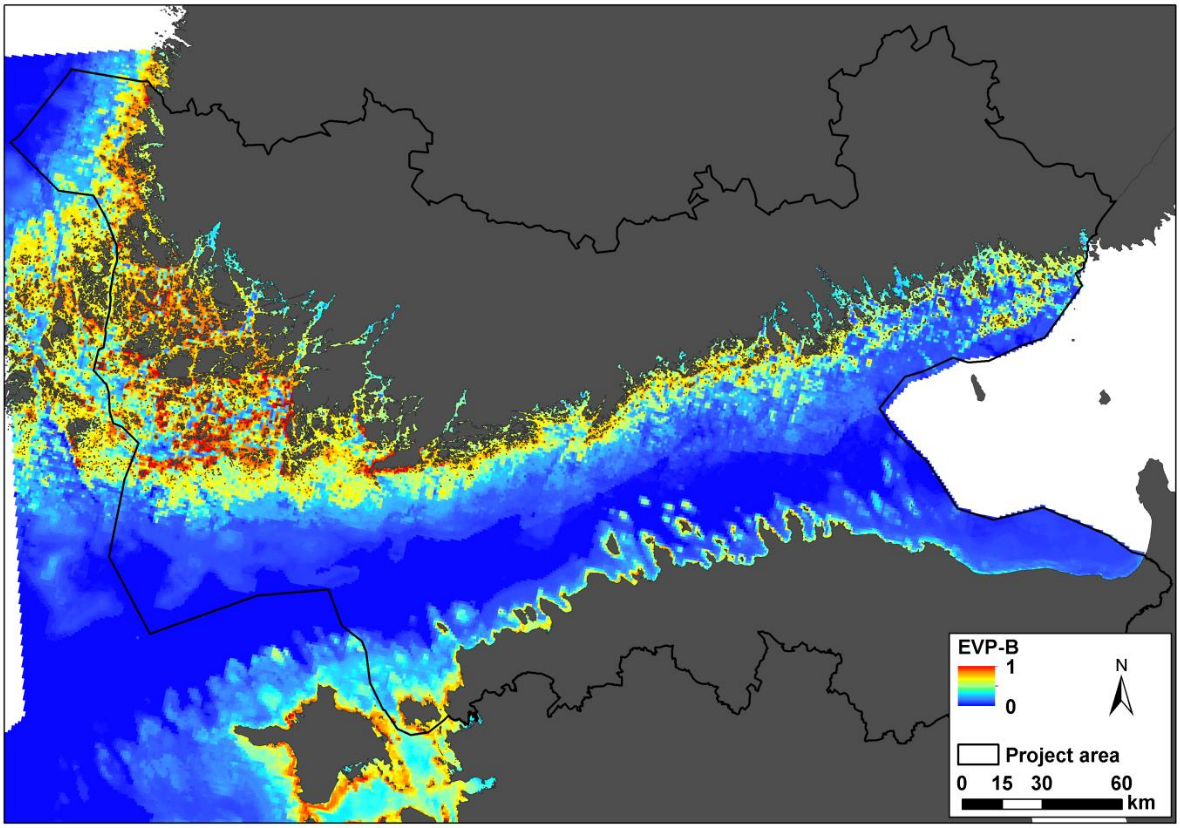

Figure 4.2. Environmental vulnerability profile EVP-B.

References

Antsulevich A, Välipakka P, Vaittinenc J (2003) How are the zebra mussels doing in the Gulf of Finland? Proc Estonian Acad Sci Biol Ecol 52:268-283.

Ardron JA, Clark MR, Penney AJ, Hourigan TF, Rowden AA, Dunstan PK, Watling L, Shank TM, Tracey DM, Dunn MR, Parker SJ (2014) A systematic approach towards the identification and protection of vulnerable marine ecosystems. Mar Policy 49:146-154.

Arroyo NL and Bonsdorff E (2016) The role of drifting algae for marine biodiversity. - in E. Olafsson (ed.): Marine Macrophytes as Foundation Species. CRC Press, Taylor & Francis Group. 285 pp.

Austin AP (1960) Observations on the growth, fruiting and longevity of *Furcellaria fastigiata* (L.) Lamouroux. Hydrobiologia 15:193-207.

Bax NJ and Williams A (2001) Seabed habitat on the southeastern Australian continental shelf: context, vulnerability and monitoring. Mar Freshw Res 52:491-512.

Bendtsen J, Gustafsson KE, Söderkvist J, Hansen JLS (2009) Ventilation of bottom water in the North Sea–Baltic Sea transition zone. J Marine Syst 75:138-149.

Beroya-Eitner MA (2016) Ecological vulnerability indicators. Ecol Indic 60:329-334.

Bertocci I, Arenas F, Matias M, Vaselli S, Araújo R, Abreu H, Pereira R, Vieira R, Sousa-Pinto I (2010) Canopy-forming species mediate the effects of disturbance on macroalgal assemblages on Portuguese rocky shores. Mar Ecol Prog Ser 414:107-116.

Beukema JJ, Cadée GC (1996) Consequences of the sudden removal of nearly all mussels and cockles from the Dutch Wadden Sea. Mar Ecol 17:279-289.

Beukema JJ, Flach EC, Dekker R, Starink M (1999) A long-term study of the recovery of the macrozoobenthos on large defaunated plots on a tidal flat in the Wadden Sea. J Sea Res 42:235-254.

Bird NL, Chen LC-M, McLachlan J (1979) Effects of temperature, light and salinity of growth in culture of *Chondrus crispus, Furcellaria lumbricalis, Gracilaria tikvahiae* (Gigartinales, Rhodophyta), and *Fucus serratus* (Fucales, Phaeophyta). Bot Mar 22:521-527.

Boedeltje G, Smolders AJP, Roelofs JGM, Van Groenendael JM (2001) Constructed shallow zones along navigation canals: vegetation establishment and change in relation to environmental characteristics. Aquatic Conserv: Mar Freshw Ecosyst 11:453-471.

Boström C, Baden S, Bockelmann AC, Dromph K, Fredriksen S, Gustafsson C et al. (2014) Distribution, structure and function of Nordic eelgrass (*Zostera marina*) ecosystems: implications for coastal management and conservation. Aquatic Conserv: Mar Freshw Ecosyst 24:410-434.

Camphuysen CJ, Fox AD, Leopold MF, Petersen IK (2004) Towards standardised seabirds at sea census techniques in connection with environmental impact assessments for offshore wind farms in the U.K. Report commissioned by COWRIE for the Crown Estate, London. Royal Netherlands Institute for Sea Research, Texel, 38pp.

Certain G, Jørgensen LL, Christel I, Planque B, Bretagnolle V (2015) Mapping the vulnerability of animal community to pressure in marine systems: disentangling pressure types and integrating their impact from the individual to the community level. ICES J Mar Sci 72:1470-1482.

Cormier R, Kannen A, Elliott M, Hall P, Davies IM (2013) Marine and coastal ecosystem-based risk management handbook. ICES Cooperative Research Report No. 313. 60 pp.

Cutter SL, Boruff BJ, Shirley WL (2003) Social vulnerability to environmental hazards. Soc Sci Quart 84:242-261.

De Lange HJ, Sala S, Vighi M, Faber JH (2010) Ecological vulnerability in risk assessment — A review and perspectives. Sci Total Environ 408:3871-3879.

Douvere F and Ehler CN (2010) The importance of monitoring and evaluation in adaptive maritime spatial planning. J Coast Conservat 15:305-311.

Duarte CM (2002) The future of seagrass meadows. Environ Conserv 29:192-206.

Eleftheriou A and McIntyre AD (2005) Methods for the Study of Marine Benthos, 3rd edn. Blackwell, Oxford

Elith J, Graham CH, Anderson RP, Dudik M, Ferrier S, Guisan A et al. (2006) Novel methods improve prediction of species' distributions from occurrence data. Ecography 29:129-151

Elith J, Leathwick JR, Hastie T (2008) A working guide to boosted regression trees. J Anim Ecol 77:802-813.

Elith J and Leathwick JR (2009) Species Distribution Models: Ecological Explanation and Prediction Across Space and Time. Annu Rev Ecol Evol Syst 40:677-697.

Elith J and Leathwick JR (2017) Boosted Regression Trees for ecological modeling. https://cran.r-project.org/web/packages/dismo/vignettes/brt.pdf Accessed 15 March 2017

EU (2014) Directive 2014/89/EU of the European Parliament and of the council of 23 July 2014 establishing a framework for maritime spatial planning. European Commission.

Foley MM, Armsby MH, Prahler EE, Caldwell MR, Erickson AL, Kittinger JN, Crowder LB, Levin PS (2013) Improving ocean management through the use of ecological principles and integrated ecosystem assessments. Bioscience 63:619-631.

Fox AD, Desholm M, Kahlert J, Christensen TK, Petersen IK (2006) Information needs to support environmental impact assessment of the effects of European marine offshore wind farms on birds. Ibis 148:129-144.

Friedman JH, Hastie T, Tibshirani R (2000) Additive logistic regression: a statistical view of boosting. Ann Stat 28:337-407.

Green AJ, Figuerola J, Sánchez MI (2002) Implications of waterbird ecology for the dispersal of aquatic organisms. Acta oecol 23:177-189.

Greve TM, Krause-Jensen D, Rasmussen MB, Christensen PB (2005) Means of rapid eelgrass (*Zostera marina* L.) recolonisation in former dieback areas. Aquat Bot 82:143-156.

Guisan A and Zimmermann NE (2000) Predictive habitat distribution models in ecology. Ecol Model 135:147-186.

Guisan A and Thuiller W (2005) Predicting species distribution: offering more than simple habitat models. Ecol Lett 8:993-1009.

HELCOM (2003) The Baltic marine environment 1999–2002. Baltic Sea Environment Proceedings 87. 46 pp

HELCOM (2009) Eutrophication in the Baltic Sea: An integrated thematic assessment of the effects of nutrient enrichment in the Baltic Sea Region. http://www.helcom.fi/Lists/Publications/BSEP115B.pdf

HELCOM (2011) The Fifth Baltic Sea Pollution Load Compilation (PLC-5). Baltic Sea Environment Proceedings 128. 217 pp.

HELCOM (2013) HELCOM Red List of Baltic Sea species in danger of becoming extinct. Baltic Sea Environment Proceedings No. 140

HELCOM (2015) Manual for Marine Monitoring in the COMBINE Programme of HELCOM. http://www.helcom.fi/action-areas/monitoring-and-assessment/manuals-and-guidelines/combine-manual Accessed 2 Februar 2017

Herkül K, Kotta J, Kutser T, Vahtmäe E (2013) Relating Remotely Sensed Optical Variability to Marine Benthic Biodiversity. PLoS ONE 8:e55624.

Hiddink JG, Jennings S, Kaiser MJ (2007) Assessing and predicting the relative ecological impacts of disturbance on habitats with different sensitivities. J Appl Ecol 44:405-413.

Hinkel J (2011) “Indicators of vulnerability and adaptive capacity”: Towards a clarification of the science–policy interface. Glob Environ Chang 21:198-208.

Holt TJ, Hartnoll RG, Hawkins SJ (1995) Sensitivity and vulnerability to man induced change of selected communities: Intertidal brown algal shrubs *Zostera* beds and *Sabellaria spinulosa* reefs. English Nature. ENRR No. 234. 97 pp.

Ippolito A, Sala S, Faber JH, Vighi M (2010) Ecological vulnerability analysis: a river basin case study. Sci Total Environt 408:3880-3890.

Karlson K, Rosenberg R, Bonsdorff E (2002) Temporal and spatial large-scale effects of eutrophication and oxygen deficiency on benthic fauna in Scandinavian and Baltic waters — a review. Oceanogr Mar Biol Annu Rev 40:427-489.

Kautsky N (1982a) Growth and Size Structure in a Baltic *Mytilus edulis* Population. Mar Biol 68:117-133.

Kautsky N (1982b) Quantitative Studies on Gonad Cycle, Fecundity, Reproductive Output and Recruitment in a Baltic *Mytilus edulis* Population. Mar Biol 68:143-160.

Kautsky L (1988) Life strategies of soft bottom macrophytes. Oikos 53:126-135.

Kautsky H, Kautsky L, Kautsky N, Kautsky U, Lindblad C (1992) Studies on the Fucus vesiculosus community in the Baltic Sea. Acta Phytogeogr Suec 78:33-48.

Kautsky L and Kautsky N (2000) The Baltic Sea, including Bothnian Sea and Bothnian Bay. In: Sheppard, C. (Ed.), Seas at the Millennium: An Environmental Evaluation. Elsevier, Amsterdam, pp. 121-133.

Kersen P, Kotta J, Bučas M, Kolesova N, Deķere Z (2011) Epiphytes and associated fauna on the brown alga Fucus vesiculosus in the Baltic and the North Seas in relation to different abiotic and biotic variables. Mar Ecol 32:87-95.

Kiirikki M and Lehvo A (1997) Life Strategies of Filamentous Algae in the Northern Baltic Proper. Sarsia 82:259-267.

Koivisto ME and Westerbom M (2010) Habitat structure and complexity as determinants of biodiversity in blue mussel beds on sublittoral rocky shores. Mar Biol 157:1463-1474.

Korpinen S, Meski L, Andersen JH, Laamanen M (2012) Human pressures and their potential impact on the Baltic Sea ecosystem. Ecol Indic 15:105-114.

Kostamo K and Mäkinen A (2006) Observations on the mode and seasonality of reproduction in *Furcellaria lumbricalis* (Gigartinales, Rhodophyta) populations in the northern Baltic Sea. Bot Mar 49:304-309.

Kotta J and Orav H (2001) Role of benthic macroalgae in regulating macrozoobenthic assemblages in the Väinameri (north-eastern Baltic Sea). Ann Zool Fenn 38:163-171.

Kraufvelin P, Ruuskanen AT, Nappu N, Kiirikki M (2007) Winter colonisation and succession of filamentous macroalgae on artificial substrates and possible relationships to *Fucus vesiculosus* settlement in early summer. Estuar Coast Shelf S 72:665-674.

La Rivière M, Aish A, Gauthier O, Grall J, Guérin L, Janson A-L, Labrune C, Thibaut T, Thiébaut E (2016) Assessing benthic habitats' sensitivity to human pressures: a methodological framework – Summary report. Rapport SPN 2016-87. MNHN. Paris, 42 pp.

Lauridsen TL, Jeppesen E, Sondergaard M (1994) Colonization and succession ofsubmerged macrophytes in shallow Lake Vaeng during the first five yearsfollowing fish manipulation. Hydrobiologia 275:233-242.

Lauringson V, Kotta J, Orav-Kotta H, Kotta I, Herkül K, Põllumäe A (2009) Comparison of benthic and pelagic suspension feeding in shallow water habitats of the Northeastern Baltic Sea. Mar Ecol 30:43-55.

Liaw a, Wiener M (2002). Classification and Regression by randomForest. R News 2(3), 18--22.

Luigujõe L and Auniņš A (2016) Talvituvate lindude rahvusvaheline lennuloendus. Report. http://www.keskkonnaamet.ee/public/LuigujoeAunins_2016_talvituvate_veelindude-rahvusvaheline_lennuloendus_lopparuanne.pdf Accessed 3 April 2017

Mackie GL, Gibbons WN, Muncaster BW, Gray IM (1989) The zebra mussel, *Dreissena polymorpha*: A synthesis of European experiences and a preview for North America. Report prepared for Water Resources Branch, Great Lakes Section. Available from Queen's Printer for Ontario, ISBN 0-7729-5647-2.

Magurran AE (2004) Measuring Biological Diversity. Blackwell Publishing company, Australia.

Maljutenko I and Raudsepp U (2014) Validation of GETM model simulated long-term salinity fields in the pathway of saltwater transport in response to the Major Baltic Inflows in the Baltic Sea. 2014 IEEE/OES Baltic International Symposium (BALTIC), J. Carroll, Ed., Tallinn, IEEE/OES 23-31 http://ieeexplore.ieee.org/lpdocs/epic03/wrapper.htm?arnumber=6887830. Accessed 5 April 2016

Manganaro A, Pulicanò G, Reale A, Sanfilippo M, Sarà G (2009) Filtration pressure by bivalves affects the trophic conditions in Mediterranean shallow ecosystems. Chem Ecol 25:467-478.

Martin G, Paalme T, Torn K (2006) Growth and production rates of loose-lying and attached forms of the red algae *Furcellaria lumbricalis* and *Coccotylus truncatus* in Kassari Bay, the West Estonian Archipelago Sea. Hydrobiologia 554:107-115.

Martin CS, Carpentier A, Vaz *S*, Coppin F, Curet L, Dauvin J-C et al. (2009) The Channel habitat atlas for marine resource management (CHARM): an aid for planning and decision-making in an area under strong anthropogenic pressure. Aquat Living Resour 22:499-508.

Martin G, Kotta J, Möller T, Herkül K (2013) Spatial distribution of marine benthic habitats in the Estonian coastal sea, northeastern Baltic Sea. Estonian Journal of Ecology 62:165-191.

Minchinton TE, Scheibling RE, Hunt HL (1997) Recovery of an Intertidal Assemblage Following a Rare Occurrence of Scouring by Sea Ice in Nova Scotia, Canada. Botanica Marina 40:139-148.

Nejrup L and Pedersen M (2008) Effects of salinity and water temperature on the ecological performance of *Zostera marina*. Aquat Bot 88:239-246.

Nikolopoulos A, Isæus M (2008) Wave exposure calculations for the Estonian coast. AquaBiota Water Research AB, Stockholm, Sweden.

Nõges P, Tuvikene L, Feldmann T, Tõnno I, Künnap H, Luup H, Salujõe J, Nõges T (2003) The role of charophytes in increasing water transparency: A case study of two shallow lakes in Estonia. Hydrobiologia 506:567-573.

Ogden JC, Baldwin JD, Bass OL, Browder JA, Cook MI, Frederick PC et al. (2014) Waterbirds as indicators of ecosystem health in the coastal marine habitats of southern Florida: 1. Selection and justification for a suite of indicator species. Ecol Indic 44:148-163.

Okey TA, Agbayani S, Hussein M, Alidina HM (2015) Mapping ecological vulnerability to recent climate change in Canada's Pacific marine ecosystems. Ocean Coast Manage 106:35-48.

Olsen JL, Stam WT, Coyer JA, Reusch TB, Billingham M, Boström C et al. (2004) North Atlantic phylogeography and large-scale population differentiation of the seagrass *Zostera marina* L. Mol Ecol 13:1923-1941.

Orlova MI and Panov VE (2004) Establishment of the zebra mussel, *Dreissena polymorpha* (Pallas), in the Neva Estuary (Gulf of Finland, Baltic Sea): distribution, population structure and possible impact on local unionid bivalves. Hydrobiologia 514:207-217.

Pihl S and Frikke J (1992) Counting Birds from Aeroplane. Report chapter. Iwrb Special Publication, 8-23.

Pitkänen H, Kiirikki M, Savchuk O, Räike A, Korpinen P, Wulff F (2007) Searching efficient protection strategies for the eutrophicated Gulf of Finland: the combined use of 1D and 3D modeling in assessing long-term state scenarios with high spatial resolution. Ambio 36:272-279.

Pitkänen H, Lehtoranta J, Peltonen H (2008) The Gulf of Finland. In: Schiewer, U (ed.) Ecology of Baltic Coastal Waters. Springer, Berlin. pp. 285–308.

Plus M, Deslous-Paoli J-M, Dagault F (2003) Seagrass (*Zostera marina* L.) bed recolonisation after anoxia-induced full mortality. Aquat Bot 77:121-134.

Råberg S and Kautsky L (2007) A comparative biodiversity study of the associated fauna of perennial fucoids and filamentous algae. Estuar Coast Shelf S 73:249-258.

Rajasilta M, Laine P, Eklund J (2006) Mortality of herring eggs on different algal substrates (*Furcellaria* spp. and *Cladophora* spp.) in the Baltic Sea—an experimental study. Hydrobiologia 554:127-130.

Rask M (1989) A note on the diet of roach, *Rutilus rutilus* L., and other cyprinids at Tvarminne, northern Baltic Sea. Aqua Fennica 19:19-27.

Rayment WJ (2008) *Furcellaria lumbricalis* A red seaweed. In Tyler-Walters H. and Hiscock K. (eds) *Marine Life Information Network: Biology and Sensitivity Key Information Reviews*, [on-line]. Plymouth: Marine Biological Association of the United Kingdom. http://www.marlin.ac.uk/species/detail/1616

R Core Team (2016) R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria

Revermann R, Schmid H, Zbinden N, Spaar R, Schröder B (2012) Habitat at the mountain tops: how long can Rock Ptarmigan (*Lagopus muta helvetica*) survive rapid climate change in the Swiss Alps? A multi-scale approach. J Ornithol 153:891-905.

Ridgeway G (2007) Generalized Boosted Models: A guide to the gbm package. http://www.saedsayad.com/docs/gbm2.pdf Accessed 31 March 2017

Rothäusler E, Corell H, Jormalainen V (2015) Abundance and dispersal trajectories of floating Fucus vesiculosus in the Northern Baltic Sea. Limnol Oceanogr 60:2173-2184.

Sabol BM, Shafer DJ, Lord E, (2005) Dredging Effects on Eelgrass (*Zostera marina*) in a New England Small Boat Harbor. *J Mar Environ Eng 8:*57-81.

Salovius S and Kraufvelin P (2004) The filamentous green alga *Cladophora glomerata* as a habitat for littoral macro-fauna in the northern Baltic Sea. Ophelia 58:65-78.

Schmieder K, Werner S, Bauer HG (2006) Submersed macrophytes as a food source for wintering waterbirds at Lake Constance. Aquat Bot 84:245-250.

Serrão EA, Brawley SH, Hedman J, Kautsky L, Samuelsson G (1999) Reproductive success of *Fucus vesiculosus* (Phaeophyceae) in the Baltic Sea. J Phycol 35:254-269.

Silberhorn GM, Dewing S, Mason PA (1996) Production of reproductive shoots, vegetative shoots, and seeds in populations of *Ruppia maritima* L. from the Chesapeake Bay, Virginia. Wetlands 16:232-239.

Sonntag N, Henriette Schwemmer H, Fock HO, Bellebaum J, Garthe S (2012) Seabirds, set-nets, and conservation management: assessment of conflict potential and vulnerability of birds to bycatch in gillnets. ICES J Mar Sci 69:578-589.

SteinhardtT and Selig U (2007) Spatial distribution patterns and relationship between recent vegetation and diaspore bank of a brackish coastal lagoon on the southern Baltic Sea. Estuar Coast Shelf S 74:205-214.

Stelzenmüller V, Fock HO, Gimpel A, Rambo H, Diekmann R, Probst WN, Callies U, Bockelmann F, Neumann H, Kröncke I (2015) Quantitative environmental risk assessments in the context of marine spatial management: current approaches and some perspectives. ICES J Mar Sci 72:1022-1042.

Stempniewicz L and Meissner W (1999) Assessment of the zoobenthos biomass consumed yearly by diving ducks wintering in the Gulf of Gdansk (southern Baltic Sea). Ornis Svec 9:143-154.

Suursaar Ü, Alari V, Tõnisson H (2014) Multi-scale analysis of wave conditions and coastal changes in the north-eastern Baltic Sea. J Coastal Res 70:223-228.

Šiaulys A and Bučas M (2015) Biological valorisation of benthic habitats in the SE Baltic Sea. Ecol Inform 30:300-304.

Torn K, Krause-Jensen D, Martin G (2006) Present and past depth distribution of bladderwrack (*Fucus vesiculosus*) in the Baltic Sea. Aquat Bot 84:53-62.

Torn K (2008) Distribution and ecology of charophytes in the Baltic Sea. Tartu University Press.

Torn K, Martin G, Kotta J, Kupp M (2010) Effects of different types of mechanical disturbances on a charophyte dominated macrophyte community. Estuar Coast Shelf S 87:27-32.

Vahteri P and Vuorinen I (2016) Continued decline of the bladderwrack, *Fucus vesiculosus*, in the Archipelago Sea, northern Baltic proper. Boreal Environ Res 21:373-386.

Van Colen C, Montserrat F, Vincx M, Herman PMJ, Ysebaert T, Degraer S (2008) Macrobenthic recovery from hypoxia in an estuarine tidal mudflat. Mar Ecol-Prog Ser 372:31-42.

Van den Berg MS, Coops H, Simons J (2001) Propagule bank buildup of Chara aspera and its significance for colonization of a shallow lake. Hydrobiologia 462:9-17.

Van Katwijk MM, Hermus DCR, de Jong DJ, Asmus RM, de Jonge VR (2000) Habitat suitability of the Wadden Sea for restoration of *Zostera marina* beds. Helgoland Mar Res 54:117-128.

Vári A and Tóth VR (2017) Quantifying macrophyte colonisation strategies—A field experiment in a shallow lake (Lake Balaton, Hungary). Aquat Bot 136:56-60.

Villa F and McLeod H (2002) Environmental Vulnerability Indicators for Environmental Planning and Decision-Making: Guidelines and Applications. Environ Manage 29:335-348.

Westerbom M, Kilpi M, Mustonen O (2002) Blue mussels, Mytilus edulis, at the edge of the range: population structure, growth and biomass along a salinity gradient in the north-eastern Baltic Sea. Mar Biol 140:991-999.

Wikström SA and Kautsky L (2007) Structure and diversity of invertebrate communities in the presence and absence of canopy-forming *Fucus vesiculosus* in the Baltic Sea. Estuar Coast Shelf S 72:168-176.

Zettler ML, Karlsson A, Kontula T, Gruszka P, Laine AO, Herkül K, Schiele KS, Maximov A, Haldin J (2013) Biodiversity gradient in the Baltic Sea: a comprehensive inventory of of macrozoobenthos data. Helgoland Mar Res 68:49-57.

Appendices

Appendix 1 List of macrobenthos taxa. Taxonomic level in brackets.

Estonian zoobenthos taxa

1 *Alderia modesta* (species)
2 *Amphibalanus improvisus* (species)
3 *Argulus* (genus)
4 *Argyroneta aquatica* (species)
5 *Asellus aquaticus* (species)
6 *Bathyporeia pilosa* (species)
7 *Bithynia tentaculata* (species)
8 *Bylgides sarsi* (species)
9 *Calliopius laeviusculus* (species)
10 *Cerastoderma glaucum* (species)
11 Ceratopogonidae (family)
12 *Chelicorophium curvispinum* (species)
13 Chironomidae (family)
14 Coleoptera (order)
15 *Cordylophora caspia* (species)
16 Corixidae (family)
17 Corophiidae juvenile (family)
18 *Corophium volutator* (species)
19 *Crangon crangon* (species)
20 *Cyanophthalma obscura* (species)
21 Diptera (order)
22 *Donacia* (genus)
23 *Dreissena polymorpha* (species)
24 *Echinogammarus stoerensis* (species)
25 Ephemeroptera (order)
26 *Erpobdella octoculata* (species)
27 *Gammarus duebeni* (species)
28 *Gammarus* juvenile (genus)
29 *Gammarus lacustris* (species)
30 *Gammarus locusta* (species)
31 *Gammarus oceanicus* (species)
32 *Gammarus salinus* (species)
33 *Gammarus* (genus)
34 *Gammarus tigrinus* (species)
35 *Gammarus zaddachi* (species)
36 *Glossiphonia complanata* (species)
37 *Gonothyraea loveni* (species)
38 *Halicryptus spinulosus* (species)
39 *Hediste diversicolor* (species)
40 *Hemimysis anomala* (species)
41 Hemiptera (order)
42 *Heterotanais oerstedii* (species)
43 Hirudinea (subclass)
44 Hydracarina (suborder)
45 *Peringia ulvae*, *Ecrobia ventrosa* (species
46 group)*
47 Hydrozoa (class)
48 *Idotea balthica* (species)
49 *Idotea chelipes* (species)
50 *Idotea granulosa* (species)
51 *Idotea* juvenile (genus)
52 *Idotea* (genus)
53 *Jaera albifrons* (species)
54 *Laomedea flexuosa* (species)
55 *Laonome armata* (species)
56 Lepidoptera (order)
57 *Leptocheirus pilosus* (species)
58 *Limapontia capitata* (species)
59 *Lymnaea stagnalis* (species)
60 *Limecola balthica* (species)
61 *Manayunkia aestuarina* (species)
62 *Marenzelleria neglecta* (species)
63 *Melita palmata* (species)
64 *Monoporeia affinis* (species)
65 *Mya arenaria* (species)

Mysis mixta (species)
Mytilus trossulus (species)
Nematoda (phylum)
Neomysis integer (species)
Odonata (order)
Oligochaeta (subclass)
Palaemon adspersus (species)
Palaemon elegans (species)
Paramysis intermedia (species)
Physa fontinalis (species)
Piscicola geometra (species)
Planorbarius corneus (species)
Planorbidae (family)
Plecoptera (order)
Pontogammarus robustoides (species)
Pontoporeia femorata (species)
Potamopyrgus antipodarum (species)
Praunus flexuosus (species)
Praunus inermis (species)
Praunus (genus)
Radix auricularia (species)
Radix baltica (species)
Rhithropanopeus harrisii (species)
Saduria entomon (species)
Stagnicola palustris (species)
Tenellia adspersa (species)
Theodoxus fluviatilis (species)
Trichoptera (order)

*considered as one group because identified as Hydrobia sp in earlier database records; formerly, both species belonged to the genus of Hydrobia

Estonian phytobenthos taxa

Aglaothamnion roseum (species)
Battersia arctica (species)
Ceramium tenuicorne (species)
Ceramium virgatum (species)
Ceratophyllum demersum (species)
Chaetomorpha linum (species)
Chara aspera (species)
Chara baltica (species)
Chara canescens (species)
Chara connivens (species)
Chara horrida (species)
Chara polyacantha (species)
Chara (genus)
Chara tomentosa (species)
Chorda filum (species)
Cladophora glomerata (species)
Cladophora rupestris (species)
Coccotylus truncatus (species)
Dictyosiphon foeniculaceus (species)
Elachista fucicola (species)
Elodea Canadensis (species)
Eudesme virescens (species)
Fontinalis (genus)
Fucus radicans (species)
Fucus vesiculosus (species)
Furcellaria lumbricalis (species)
Halosiphon tomentosus (species)
Hildenbrandia rubra (species)
Leathesia marina (species)
Monostroma balticum (species)
Myriophyllum spicatum (species)
Najas marina (species)
Percursaria percursa (species)
Pilayella littoralis, *Ectocarpus siliculosus* (species group)**
Polyides rotunda (species)

136 Polysiphonia fibrillose (species)
137 Polysiphonia fucoides (species)
138 Potamogeton perfoliatus (species)
139 *Pseudolithoderma* (genus)
140 *Punctaria tenuissima* (species)
141 *Ranunculus baudotii* (species)
142 *Rhizoclonium riparium* (species)
143 *Rhodochorton purpureum* (species)
144 *Rhodomela confervoides* (species)
145 *Ruppia cirrhosa* (species)
146 *Ruppia maritima* (species)
147 *Stictyosiphon tortilis* (species)
148 *Stuckenia pectinata* (species)
149 *Tolypella nidifica* (species)
150 *Ulothrix* (genus)
151 *Ulva* (genus)
152 *Urospora penicilliformis* (species)
153 *Vaucheria* (genus)
154 *Zannichellia palustris* (species)
155 *Zostera Marina* (species)

**considered as one group because identified as Pilayella/Ectocarpus in earlier database records

Finnish zoobenthos taxa

ACANTHOCEPHALA
ACARINA
AMPHIPODA
ANISOPTERA
ANNELIDA
ARACHNIDA
ARTHROPODA
BIVALVIA
BRACHYCERA
BRANCHIURA
CLADOCERA
COLEOPTERA
COPEPODA
CYCLOPOIDA
DIPTERA
EPHEMEROPTERA
GASTROPODA
HARPACTICOIDA
HIRUDINEA
HYDROZOA
INSECTA
ISOPODA
LEPIDOPTERA
MYSIDACEA
NEMATOCERA
NEMATODA
NEMATOMORPHA
NEMERTEA
ODONATA
OLIGOCHAETA
OPISTHOBRANCHIA
OSTRACODA
POLYCHAETA
TRICHOPTERA
TURBELLARIA
ZYGOPTERA
Ampharetidae
Astartidae
Baetidae
Caenidae
Ceratopogonidae
Chironomidae
Chrysomelidae
Coenagriidae
Culicidae
Dytiscidae
Elmidae
Enchytraeidae
Ephemeridae
Gammaridae
Halacaridae
Haliplidae

Heptageniidae
Hirudidae
Hydrobiidae
Hydroptilidae
Leptoceridae
Limnephilidae
Lymnaeidae
Murchisonellidae
Mysidae
Nephtyidae
Planorbidae
Psychodidae
Sialidae
Sphaeriidae
Tabanidae
Tipulidae
Tubificidae
Valvatidae
Chironominae
Chironomini
Orthocladiinae
Tanypodinae
Ablabesmyia
Agraylea
Amphibalanus
Anisus
Anodonta
Arctopelopia
Argulus
Athripsodes
Atrichopogon
Bithynia
Boccardia
Bosmina
Candona
Ceraclea
Chironomus
Cladotanytarsus
Coenagrion
Corophium
Cricotopus
Cryptochironomus
Cryptotendipes
Cyrnus
Daphnia
Dicrotendipes
Donacia
Endochironomus
Ephydatia
Erpobdella
Eteone
Gammarus
Glossiphonia
Gyraulus
Haliplus
Hemimysis
Hydra
Hydracarina
Hydrobia
Hydropsyche
Hydroptila
Idotea
Ilyocryptus
Jaera
Laomedea
Laonome
Limnodrilus
Lymnaea
Macoma
Macroplea
Marenzelleria
Mesostoma
Micropsectra
Mysis
Mystacides
Mytilus
Oecetis
Palaemon
Parvicardium
Piscicola
Pisidium
Planaria
Plectrocnemia
Polypedilum
Potamothrix
Potamothrix/Tubifex
Praunus
Procladius
Prostoma
Psectrocladius
Radix
Saduria

Sergentia
Sisyra
Sphaerium
Stictochironomus
Tanytarsini
Tanytarsus
Valvata

Finnish phytobenthos taxa

Batrachospermum
Hildenbrandia
Rhodocorton
Aglaothamnion roseum
Audouinella efflorescens
Audouinella purpurea
Bangia atropurpurea
Ceramium tenuicorne
Ceramium virgatum rubrum
Furcellaria lumbricalis
Phyllophora pseudoceranoides
Polyides rotundus
Polysiphonia fibrillosa
Polysiphonia fucoides
Rhodomela confervoides
Chaetomorpha
Ulva
Ulothrix
Acrosiphonia arcta
Chaetophora incrassata
Cladophora aegagropila
Cladophora fracta
Cladophora glomerata
Cladophora rupestris
Monostroma balticum
Monostroma grevillei
Spongomorpha aeruginosa
Lithoderma
Chorda filum
Dictyosiphon chordaria
Dictyosiphon foeniculaceus
Ectocarpus confervoides
Ectocarpus siliculosus
Elachista fucicola
Eudesme virescens
Fucus radicans
Fucus vesiculosus
Halosiphon tomentosus
Leathesia difformis
Pilayella littoralis
Pseudolithoderma
Scytosiphon lomentaria
Sphacelaria arctica
Sphacelaria radicans
Stictyosiphon tortilis
Vaucheria
Nitellopsis
Tolypella
Chara aspera
Chara baltica
Chara braunii
Chara canescens
Chara globularis
Chara horrida
Chara tomentosa
Chara virgata
Nitella flexilis
Nitella hyalina
Nitella opaca
Nitella walhbergiana
Marchanthiophyta
Scorpidium
Climacium dendroides
Calliergon cordifolium
Calliergon megalophyllum
Drepanocladus aduncus
Drepanocladus sordidius
Fissidens fontanus
Fissidens osmundoides
Fontinalis antipyretica
Fontinalis dalecarlica
Fontinalis hypnoides
Hygrohyphum luridium
Leptodictyum riparium
Oxyrrhynchium speciosum
Platyhypnidium riparioides
Racomitrium canescens
Warnstorfia trichophylla
Alisma plantago aquatica
Alisma wahlenbergii
Bolboschoenus maritimus
Butomus umbellatus
Callitriche cophocarpa
Callitriche hermaphroditica
Callitriche palustris
Ceratophyllum demersum
Ceratophyllum submersum
Crassula aquatica

Elatine hydropiper
Elatine orthosperma
Elatine triandra
Eleocharis acicularis
Eleocharis mamillata
Eleocharis palustris
Eleocharis palustris var *lindbergii*
Eleocharis parvula
Eleocharis uniglumis
Eleocharis uniglumis suniglumis
Elodea canadensis
Equisetum fluviatile
Glyceria fluitans
Glyceria maxima
Hippuris tetraphylla
Hippuris vulgaris
Hippuris lanceolata
Hydrocharis morsus ranae
Iris pseudoacorus
Isoetes echinospora
Isoetes lacustris
Lemna minor
Lemna trisulca
Limosella aquatica
Myriophyllum alterniflorum
Myriophyllum sibiricum
Myriophyllum spicatum
Myriophyllum verticillatum
Najas marina
Najas tenuissima
Nuphar lutea
Nymphaea candida
Nymphaea alba
Persicaria foliosa
Phragmites australis
Potamogeton alpinus
Potamogeton berchtoldii
Potamogeton compressus
Potamogeton friesii
Potamogeton gramineus
Potamogeton gramineus x *perfoliatus*
Potamogeton lucens
Potamogeton natans
Potamogeton obstusifolia
Potamogeton perfoliatus
Potamogeton praelongus
Potamogeton pusillus
Potentilla palustris
Ranunculus circinatus
Ranunculus confervoides
Ranunculus peltanus ssp *peltatus*
Ranunculus peltatus ssp *baudotii*
Ranunculus reptans
Ruppia cirrhosa
Ruppia maritima
Sagittaria natans
Sagittaria sagittifolia
Sagittaria sagittifolia x *natans*
Schoenoplectus lacustris
Schoenoplectus maritimus
Schoenoplectus tabernaemontani
Sparganium angustifolia
Sparganium emersum
Sparganium gramineum
Sparganium natans
Stratiotes aloides
Stuckenia filiformis
Stuckenia pectinata
Stuckenia vaginata
Subularia aquatica
Typha angustifolia
Typha latifolia
Utricularia australis
Utricularia intermedia
Utricularia minor
Utricularia vulgaris
Zannichellia major
Zannichellia palustris
Zostera marina

European Regional Development Fund

European Union

Interreg

Central Baltic

KIK

KESKKONNAINVESTEERINGUTE KESKUS

SYKE

Turun yliopisto

University of Turku

UNIVERSITY OF TARTU

Helsinki-Uusimaa Regional Council

VARSINAIS-SUOMEN LIITTO

EGENTLIGA FINLANDS FÖRBUND

REGIONAL COUNCIL OF SOUTHWEST FINLAND

BEF ESTONIA

SYKE.FI/PROJECTS/PLAN4BLUE