Environmental status and modelling effort in Vejle Fjord

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1 General Introduction

Eelgrass (*Zostera marina* Linnaeus, 1753) is a common marine angiosperm in the northern temperate hemisphere (Krause-Jensen et al. 2011). Historically, eelgrass covered 1/7 of all shallow Danish marine waters (Petersen 1901; 1914), however, due to worldwide wasting disease, the Danish eelgrass population has been greatly reduced (Rasmussen 1977) and further decline has been documented mainly due to anthropogenic eutrophication (Hauxwell et al. 2003; Krause-Jensen et al. 2000; Valdemarsen et al. 2010). Despite that nutrient loadings in Danish estuaries has been reduced the past 20-25 years eelgrass has failed to recover and even shows further decline in some areas (Flindt et al. 2016).

Seagrasses are considered essential in achieving healthy and stable estuarine and coastal ecosystems (Flindt et al. 2016) due to the numerous ecosystem services they provide. This includes growth related nutrient retention, improving water quality, trapping of suspended particles, nursery ground for a number of juveniles fauna species, acts as a carbon sink and help increase biodiversity (Duffy 2006; Hemminga and Duarte 2000; Orth et al. 2006; Terrados and Duarte 2000). Therefore, seagrasses were introduced as a key element in the European Union Water Framework Directive (WFD) and are used in defining the ecological status of transitional and coastal waters. WFD was introduced in 2000 and based on biological indicators (e.g., phytoplankton and benthic flora and fauna) the directive aims to achieve good ecological status (GES) in rivers, lakes and coastal waters.

According to the WFD the ecological status must be expressed as the ratio between the actual level of a biological indicator and the reference level of the biological indicator (*Ecological Quality Ratio*, EQR). Reference levels should be based on the level of the biological indicator in an undisturbed ecosystem with “no or only minor” anthropogenic influence. However, due to anthropogenic eutrophication, no such “undisturbed” water bodies are available, and the reference levels are instead based on historical records, modelling or expert judgements. From the degree of which the actual level differs from the reference level the WFD defines five “*ecological status classes*”: high, good, moderate, poor and bad status (Krause-Jensen et al. 2005).

In Denmark, the depth limit of eelgrass (i.e., the deepest depth where 10% coverage of eelgrass is present) is used as the key *Biological Quality Element* and is yearly measured in coastal waters and estuaries (Greve et al. 2005). Although the depth limit is a useful indicator of light climate related growth conditions of eelgrass at the sampling points it is only zero- or one-dimensional information and it does not provide spatial resolution on eelgrass coverage and production (Flindt et al. 2016). But, what does it mean that this parameter is zero/one dimensional? In relation to the WFD and in simple terms, it means that if the depth limit for a certain water body is set at 5m to achieve good ecological status (GES), then such water body should have at least one area (does not matter how big) with 10 % coverage of eelgrass at 5m. This indicator does not consider whether the entire fjord is covered with eelgrass at 5m depth or if only a very local area of the fjord has one patch with 10% coverage of eelgrass. On the other hand, the presence and absence of eelgrass are monitored through the NOVANA stations, which are very limited to only one or a few transects for each water body. In this case, if no eelgrass is found at 5 m depth in these transects, the GES is not achieved. From an ecological perspective, the presence of eelgrass at one single location (even if at 5 m), or spread all over the fjord to 5 m depth, has very different ecological implications, and provide the water body with different resilience levels. The ecosystem services provided by eelgrass are related to the area where eelgrass is spread. The higher area of distribution, the higher buffer capacity (capability to trap nutrients and carbon for longer periods) eelgrass provide to the system.

In Denmark, eelgrass recovery is still lacking even in areas where the light climate is supporting eelgrass growth (Flindt et al. 2016; Valdemarsen et al. 2014). Some of the parameters that prevent eelgrass to spread over areas with sufficient benthic light availability are: sediment characteristics (the sediment is too muddy due to the many years of high nutrient load), the lack of parental eelgrass beds (the reproduction success in *Z. marina* is very low, as well as the stock population), the presence of other colonizers (ex. sand worm *A. marina* acts as destabilizer, displacing the eelgrass recovery) etc (Flindt et al 2016). It is therefore that a good understanding of present ecological conditions and area specific data is needed to evaluate future scenarios (ex. reduction scenarios) in a specific water body. In relation to the WFD, if the focus remains only on achieving good light availability for eelgrass at the sea bed, we could assume that all areas with such levels of light will be covered by eelgrass, this will likely assume a much higher eelgrass livestock than existing, hence a much higher buffer capacity than existing and therefore will wrongly estimate future scenarios. For a better understanding of eelgrass recovery, spatial information of eelgrass coverage and area-specific production is essential (Benson et al. 2013; Orth et al. 2012). When developing tools to assess the environmental conditions in entire marine ecosystems it is essential to include area-based data on ex. eelgrass biomass and coverage along with growth and loss processes (Flindt et al. 2016).

To enhance measured field data computer modelling can be an important add-on in providing the spatial resolution needed in proper ecosystem management. For about 40 years computer models have been used in describing and analyzing marine- and freshwater systems (Fath et al. 2012; Janssen et al. 2015). Initially, mechanistic modelling was used as “water quality models” to describe plankton- and nutrient dynamics and dissolved oxygen conditions. With technical development and increased computational power mechanistic models evolved into complex and dynamic models simulating whole ecosystems with increasing complexity of forcing functions, auxiliary functions, state variables etc. In other words, the model not only can simulate the transfer of nutrients from functional groups (e.g. primary producers), but can combine them with other feedback mechanisms and/or limitations quantified through scientific studies (e.g. eelgrass should not be able to grow if the sediment is too muddy, or the presence of A. marina is over a certain threshold). In addition, the spatial and temporal resolutions were improved. In model development the complexity of the model must be carefully evaluated. Depending on the specific need and use of the model, often spatial and temporal resolution are more important that introducing additional state variables (key elements, e.g.. mussel biomass in the system) into the model. High spatial resolution is especially important in modeling parameters growing in patches (e.g., benthic primary producers such as seagrasses) and along step bathymetric slopes where e.g., one grid cell may cover more than one depth interval. There is always tradeoff between the spatial resolution of the model and the extension and complexity of the same. For example, in our case we would like to investigate the fate of eelgrass in a nutrient decreasing scenario, however, we would like to run our model for the entire Nordlige Bælthav, so that the relative differences from the areas are in focus and so we could take relatively more actions in those area with the higher impacts. In this setup we will typically sacrifice some spatial resolution in our model, e.g. the definition of the shallow areas will be simplified to gain the necessary computational power to run correctly the general hydrodynamics for the entire area. But what does this mean? When reducing the spatial resolution of the shallow areas we will underestimate or overestimate the potential of eelgrass (depending on our choices). If we cluster in a single calculation cell the areas between 0 and 5 m depth and state an average depth of 2,5m to run our model, we will overestimate the eelgrass biomass because there will be a large area withing your one grid cell that in reality is at >3m depth and does not support enough light to develop eelgrass biomass. On the other hand, if we decide that this grid should be modelled as 3,5 m, we will underestimate the potential in our model (due to, in reality, there is much more productive area than we consider in the model). Models are always a simplification of reality, hence they need to be used to the correct purpose, in the present example, to see the relative impacts in areas that are very large. In simple terms to simulate local changes generated by local actions, the models have to be downscaled. There you can use your computational capacity to run smaller grids (higher accuracy of e.g. local bathymetry compared to the reality) while sacrificing the large overall area for calculations. In this case, the eelgrass coverage should be much more accurate, however not totally comparable to the results found in the large model. Temporal resolution is important in the advection and dispersion of solutes, particles and organisms such as nutrients, phytoplankton and other 3D parameters suspended in the water column (Fraysse et al. 2013). The processes run in mechanistic models occurs withing different seasonalities and different day lengths etc. The different processes therefore need to be included at the correct temporal resolution. For instance, the primary production growth process can be somehow more simplified in the number of time steps than the sedimentation of a particle, since both processes in reality occurs at a very different temporal speed. The fist process is much slower than the second, and this differences and calculation accuracy needs to be weighted in the models. At present, modelling is used in a variety of different fields such as assessing effect of climate change (Meier et al. 2011a; Neumann 2010), effect from measures against eutrophication (Thieu et al. 2010), and eutrophication status of systems (Nobre et al. 2005) as well as simulating the impact from different stressors on eelgrass recovery (Canal-Vergés et al. 2014; Kuusemäe et al. 2016; Kuusemäe et al. 2018). Dynamic mechanistic models are very useful in scientific research as they allow for applying different pressure to an ecosystem and studying the effects in high spatial and temporal resolution (Nobre et al. 2005). In addition, models are useful tool in testing scientific hypothesis (Jørgensen and Fath 2011) prior to e.g., extensive field campaigns. Dynamic mechanistic models have been applied to both the Baltic- (Meier et al. 2011b; Neumann and Schernewski 2005) and the North Sea (Lenhart et al. 2010; Thieu et al. 2010) to investigate effects from nutrient loading on the ecological status of the systems. In the Danish WFD work models have been used to calculate the maximum allowable input of N and P (N-MAI and P-MAI) in order to achieve and maintain good ecological status (GES) in the Danish WFD water bodies (Erichsen et al. 2020). By comparing the calculated MAIs and the present nutrient loadings to Danish water bodies nutrient reduction scenarios were suggested (Erichsen et al. in prep-a) and non-coupled 30% N and P reduction scenarios were applied to the Danish River Basin Management Plans 2021-2027 (RBMP 2021-2027). The MAIs defining the need for nutrient reduction is based on chlorophyll -*a* and the depth limit of eelgrass, however, as introduced above, the eelgrass depth limit does not provide area-based information about coverage or production and thus, the effect of such nutrient reductions should be evaluated using an area-based approach (Canal-Vergés et al. 2021) (e.g., geographical information systems, GIS). Therefore, our focus was to investigate not only what is the MAI to reach 10% eelgrass coverage at at least one location withing the model, but also, what is the overall status and most important overall potential of the systems given the current status and the potential reduction. This imply an area analyses of the different modelled parameters and results of the model. Here we can elaborate on the impact of the different parameters on eelgrass development as well as to plan or discuss action plans for concrete location and problems.

In short, Geographical Information Systems, or GIS, is a georeferenced database that consists of three main elements: a computer system, a GIS software and spatially referenced or geographical data. GIS can be used to add value to a specific dataset such as creating figures, tables or maps from a raw model output. In addition, GIS allows for viewing and organizing datasets efficiently, merging data with other datasets and creating new datasets by queries performed on existing data. The spatial referencing of the data is fundamental and GIS data should always have a georeferenced entity type (e.g., lines, points or polygons). Thus, a row in a GIS data table always refer to a geolocation (line, point or polygon) in which different information about the location can be stored. In GIS, functionalities such as the ability to transform spatial data from one entity type the another (e.g., interpolation between point data creating area-based polygons) are vital in performing spatial analysis which is a key element that renders GIS as a valuable tool in ecosystem analysis.

In this report we a status of modelling performed in Vejle Fjord and introduce the mechanistic model complex developed for Vejle Fjord during both the Danish WFD work and Velux project “Sund Vejle Fjord”. In addition, we present two GIS tools developed specifically to provide ecosystem analysis in high spatial resolution with emphasis on eelgrass recovery potential, gradient analysis of benthic- and pelagic production and nutrient dynamics using either model- or field data as input.

2 Models and supporting GIS-tools in Vejle Fjord

2.1 Introduction

In this section two mechanistic models covering Vejle Fjord (VF) are presented, and the spatial resolution of the model domains are compared and evaluated. Both models are introduced using a baseline scenario to evaluate the bathymetric solution of the two Vejle Fjord models. Having sufficient resolution on the model bathymetry is especially important when simulating complex and dynamic systems on steep bathymetric slopes such as Vejle Fjord. Vejle fjord has steep slopes starting close to the shore.Simplified, this means that the fjord does not have a high area with shallow depths, compare to the higher depths. In the WFD, they estimate that in order to achieve GES in Vejle Fjord, eelgrass depth limit should be 5,7 and 7,3 m in the inner and outer fjord (even if eelgrass is only found at 1,9 and 2,6 m for the inner and outer fjord at present), respectively. Given the low total area under 7 m in the fjord, it is imperative that these shallow depth ranges are highly defined in the model, so that each of the shallow depth intervals is adjusted as much as possible to the reality, to calculate the correct buffer capacity of benthic primary producers in the fjord. Furthermore, this refined bathymetry will as well facilitate the visibility of local changes to the system.

The first VF model (VF_{RBMP}) was developed as part of the marine modelling complex used in the River Basin Management plans 2021-2027 (WFD). The VF_{RBMP} model is a part of a larger model (Nordlige Bælthav model, NBH) covering the entire Nordlige Bælthav, from north of Funen enclosed by Jutland to the west and Zeeland and Kattegat to the east. Due to the large scale of this model, some spatial resolution has been sacrificed in local areas such as Vejle Fjord. Thus, the horizontal mesh in the Vejle Fjord model domain in NBH consists of only 2367 finite element polygons (Figure 1, A) in which calculations occur.

The second VF model (VF_{HiRes}) has been developed in connection to the Velux project “Sund Vejle Fjord”, as a high-resolution estuary model covering Vejle Fjord. When covering a smaller area compared to the NBH model it is possible to enhance spatial resolution and thus, the horizontal mesh in the VF_{HiRes} model consists of 13930 finite elements (Figure 1, B) enabling improved resolution on bathymetry and dynamic item calculation. High spatial resolution is especially important in modeling parameters growing in patches (e.g., benthic primary producers such as seagrasses) and along step bathymetric slopes where e.g., one grid cell else may cover more than one depth interval.

In addition, the methods behind two GIS-tools (Canal-Vergés et al. 2016; Flindt et al. 2016; Lees 2021) that are applicable to different water bodies and used in the RBMP work are briefly introduced. The combined model- and GIS-tool results from the RBMP work in Vejle Fjord is presented in section 3.

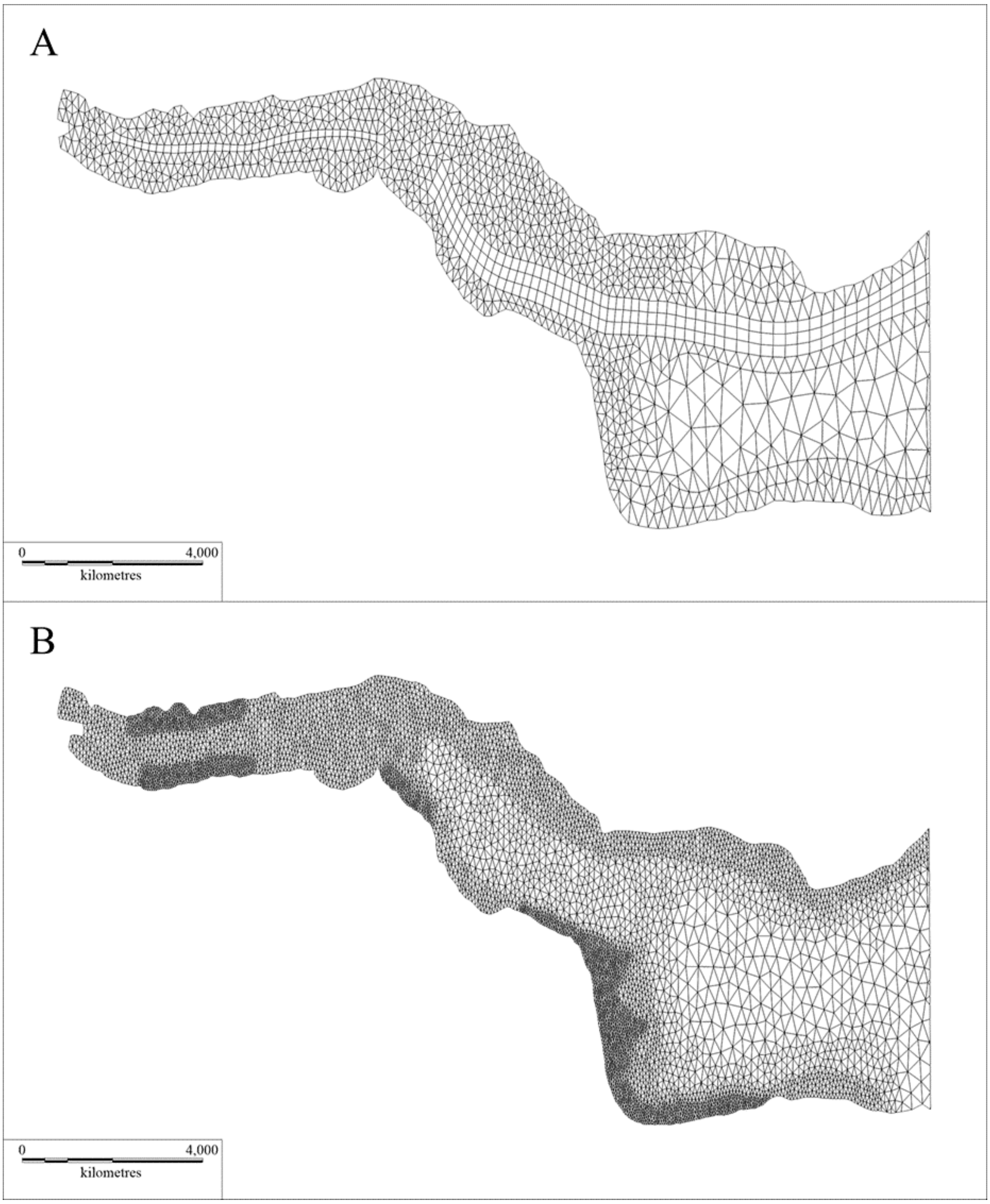

Figure 1. *Vejle Fjord, Denmark*. Model domain of the VF_{RBMP} (A) and VF_{HiRes} (B) models showing the differences in number and spatial resolution of finite elements polygons. Scalebar is 4 kilometres.

2.2 Methods

2.2.1 The mechanistic model complex

The mechanistic models consist of three modules: 1) a hydrodynamic module (HD), 2) an advection/dispersion module (AD) and an aquatic ecosystem module (AEM or ECOlab). The HD module computes physical parameters such as water levels, current speed and direction, salinity and temperature. The AD module is a transport module and computes the advection and dispersion of biochemical components such as particulate and dissolved nutrients. The numerical MIKE solver ECOlab AEM module computes changes in concentrations of biochemical components due to various ecological processes such as growth and loss processes (Erichsen and Birkeland 2019).

2.2.2 Details on the AEM module

The AEM module comprise three compartments: pelagic- and sediment compartment and benthic production (Figure 2).

The pelagic compartment simulates the concentrations of phytoplankton, zooplankton, detritus, dissolved organic matter and dissolved oxygen content of the water phase. Phytoplankton growth is modelled as result of phytoplankton primary production minus losses. The main factors controlling phytoplankton production are nutrients and light availability along with temperature. Loss processes are mainly controlled by respiration, grazing and sedimentation. The nutrient uptake of phytoplankton is controlled by Monod kinetics where nutrients are taken up into internal pools in the algal cells as a function of the ambient nutrient concentration. The post-uptake phytoplankton growth is then controlled by Droop kinetics (Droop 1968) where growth is a function of the intracellular nutrient concentration. To differentiate between seasonal changes in algal characteristics the AEM models phytoplankton in three functional groups: 1) silicate dependent diatom state variable, 2) flagellate state variable and 3) colony-forming cyanobacteria state variable. The diatom state variable introduces non-motile low light dependent algal cells which rely on water turbulence to prevent sedimentation. The flagellate state variable introduces neutrally buoyant algal cells to the model. The cyanobacteria state variable introduces N_2 fixing algal cells that inhabit brackish waters (< 10-12 psu). Additionally, these algal cells show a strong growth response with increasing temperatures and can aggregate in large numbers in surface waters.

The phytoplankton concentration is reduced by grazing and decomposition and is either transformed into the zooplankton or detritus state variable (pools). Detritus is described as particulate and dissolved organic matter (C, N and P) and is either subject to sedimentation or bacteria driven remineralization processes (microbial loop). Hence, remineralization of labile organic matter reintroduces otherwise “lost” N and P to the water column several times. These high turnover rates support additional phytoplankton growth influencing the light climate.

In addition to detritus from phytoplankton loss processes C, N and P as dissolved organic matter (DOM) are also introduced in the system by land-based runoff. In the models, DOM is divided into labile dissolved organic matter (LDOM) and relatively refractory colored dissolved organic matter (CDOM). Thus, three states of organic matter are computed in the AEM module: detritus, LDOM and CDOM.

The nutrient input in the AEM module can be separated into external sources (e.g., land-based runoff areas such as rivers, direct discharges such as wastewater treatment plants and atmospheric deposition) and internal sources (e.g., sediment fluxes and mineralization of organic matter produced in the water column). Additionally, pelagic recycling is also included in the module where nutrients are recycled in the water column and sediments due to heterotrophic activity. Finally, N_2 fixation from cyanobacteria constitute an important N source in the larger regional models. However, since large occurrences of cyanobacteria only rarely are present in the local- and estuary model areas N_2 fixation is not applied to those models. Hence, N_2 fixation only indirectly affect the local- and estuary models through the boundary conditions.

The pelagic compartment is coupled with a two-layer sediment compartment through several processes such as sedimentation, filtration, nutrient uptake by benthic plants and macroalgae, bioturbation, mineralization, resuspension and predation. These processes accounts for the exchange of solutes, particles and organisms between the two compartments. A fraction of the internal nutrient source in then pelagic compartment, as mentioned above, is a result of mineralization of organic matter in the sediment. This internal nutrient loading is dependent on the size of the C, N and P pools in the sediment as well as bottom oxygen concentrations, water temperature and bottom water exchange. Organic C, N and P is released to the sediment pore water by the degradation of the C, N and P pools while also, a small fraction is immobilized. The immobilization of organic C, N and P is driven by the C:N ratio and increases with a higher C:N ratio, where a low C:N ratio indicate higher lability of the organic pool. While the degradation is realized by utilizing oxygen or nitrate (NO_3) the rate degradation is dependent on the oxygen- and/or NO_3 availability as well as the C:N ratio in the sediment. The sediment compartment also computes denitrification of N_2 and the binding of inorganic P to oxidized iron (Fe^{+++}) in an oxidized sediment. Additionally, inorganic P is released to the pore water when the sediment is reduced due to sediment oxygen depletion. The AEM module ensures integration of the pelagic and sediment compartments and allow for sediment/pelagic nutrient exchange and thus, the sediment may act either as a sink or a source of inorganic nutrients to the water phase. This integration is introduced in both regional, local and estuary models.

Besides phytoplankton introduced in the pelagic compartment the benthic production compartment introduces four important benthic primary producers: Eelgrass (*Zostera marina*), annual opportunistic macroalgae (e.g., filamentous brown algae and *Ulva* sp.), perennial macroalgae (e.g., *Fucus* sp.), and benthic diatom microalgae. As with phytoplankton the benthic primary production is a result of water temperature, benthic light availability and nutrients. Additionally, eelgrass and perennial macroalgae needs appropriate sediment and substrate conditions, respectively, in order to grow. In the models, sediment conditions are related to sediment bulk density. Since eelgrass is a flowering plant with roots and rhizomes eelgrass take up inorganic nutrients from the sediment pore water and from the water column by the leaves. Thus, If the nutrient concentration in the pore water is sufficiently high this would allow for eelgrass growth even when the inorganic nutrient concentration in the water phase is low. This would also apply for microbenthic diatoms growing on the sediment however, opportunistic and perennial macroalgae can only take up nutrients from the water phase. Benthic growth is regulated by the internal N and P pools and where separate N and P pools are used for each benthic primary producer. The models allow for accumulation of internal nutrients and may drive growth in seasons where the external nutrient loading in surface waters is depleted. Loss in benthic primary production includes respiration, grazing, decay and leaf shedding. Lost benthic primary production is a source of organic matter to the water phase and the sediment where inorganic and organic nutrients are returned to the internal N and P pools through mineralization.

The biochemical models include > 50 state variables which are about equally divided between pelagic and benthic compartments. While the variables attached to the benthic compartments are fixed to the seabed or sediment surface advection and dispersion is introduced to the pelagic state variables due to the movement of water. Finally, a mass-balance module registers the transport, size and exchange of the organic matter and nutrient pools in the different compartments ensuring mass conservation in the models.

This section (2.2.2) is a short summary of a technical note (Erichsen and Birkeland 2019) on the biochemical model (AEM module) which is a part of the scientific documentation of the Mike model complex developed by DHI.

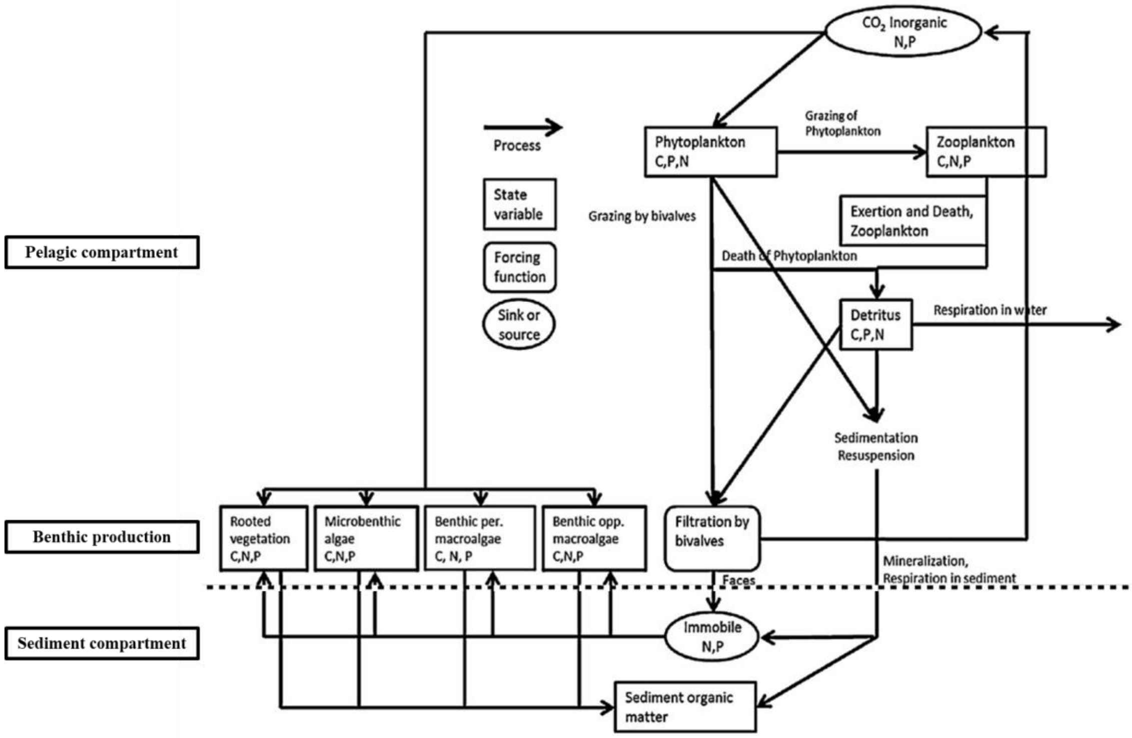

Figure 2. The three compartments of the AEM (or EcoLab) module showing the main state variables and C, N and P fluxes (Kuusemäe et al. 2016).

As eelgrass is the key biological indicator used to assess the environmental conditions in the different water bodies the AEM module has been continuously enhanced by introducing different eelgrass stressors to the module that inhibits eelgrass growth. This allows eelgrass growth conditions to be influenced not only by benthic light and nutrient availability but also by sediment quality, resuspension frequency, wave and current induced bed shear stress, ballistic stress from macroalgae and burial of seeds and seedlings by lugworms. This ensures a dynamic growth expression of eelgrass by several parameters in high resolution and helps to explain why eelgrass has not recovered in areas that supports sufficient benthic light availability (i.e., depth limit of eelgrass).

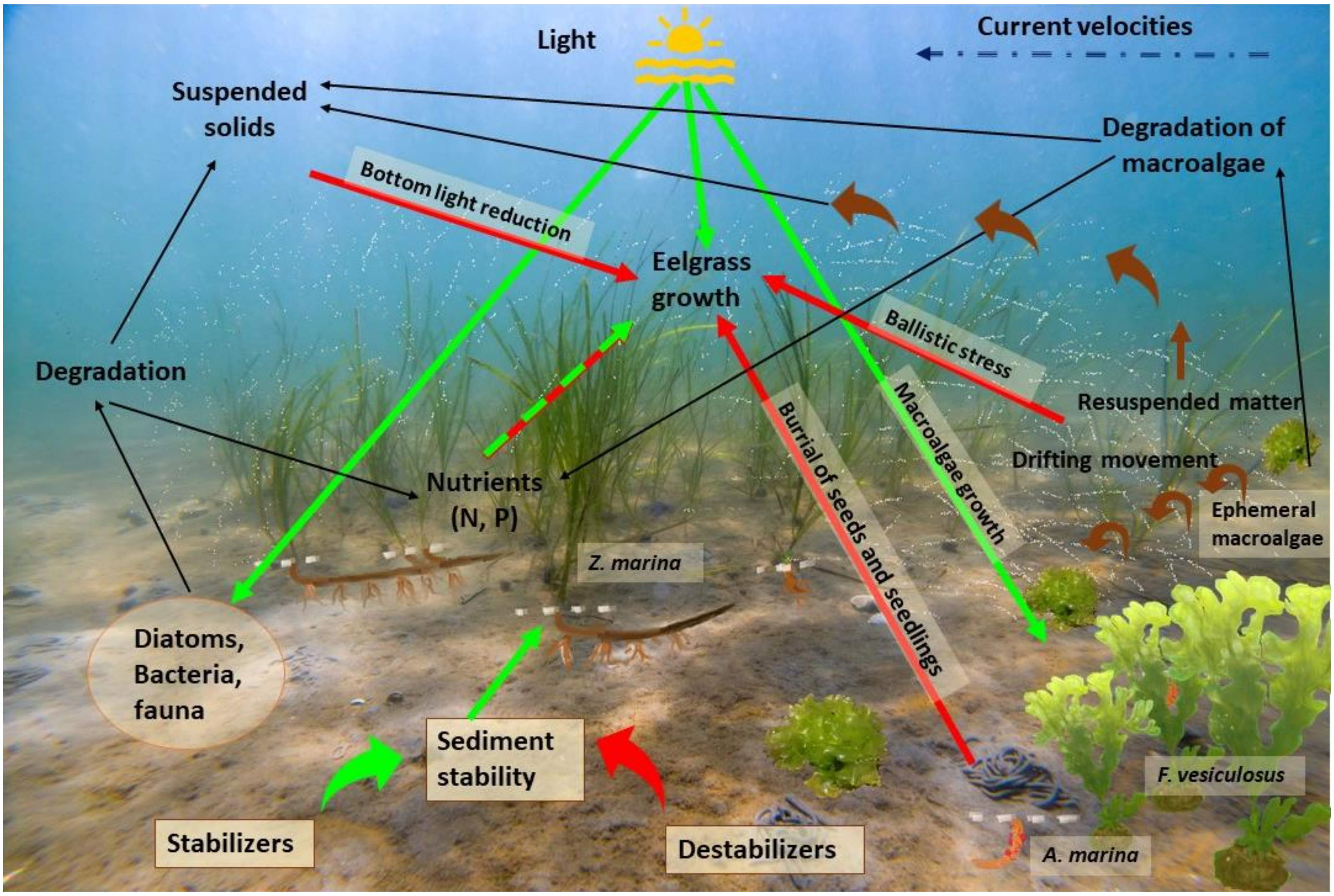

Figure 3. Main stressors inhibiting eelgrass growth and distribution. Green arrows indicate positive influence and red arrows negative influence. Red-green dashed arrow shows positive and negative effects on eelgrass growth (Kuusemäe et al. 2016).

2.2.3 GIS model tools

The GIS analysis of the model results harvested from the VF_{RBMP} and VF_{HiRes} models was performed using two GIS tools. Figure 4 help to illustrate the data flow where raw model result files are passed through the GIS tools performing ecosystem analysis in high spatial and temporal resolution.

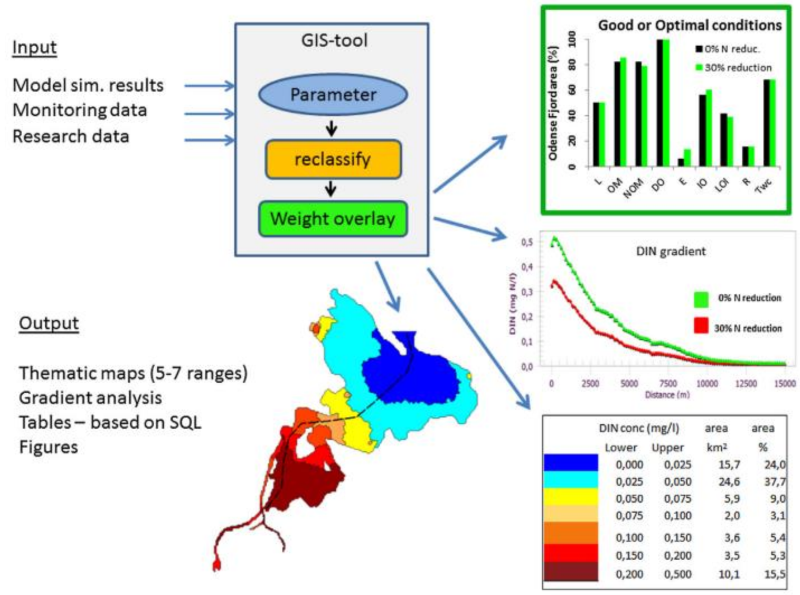

Figure 4. Illustration of the data flow from model results or field data, the processing of data either by using the data directly to produce gradient analysis, or reclassifying data using weighted overlays and create outputs as graphs, gradient and thematic maps and tables based on SQL (Flindt et al. 2016).

Tool 1 Identifies the eelgrass reestablishment potential by evaluating the impact of different eelgrass stressors affecting eelgrass growth. These stressors are discussed in detail in Flindt et al. (2016). To assess the importance of the different stressors this GIS-tool combines results from mechanistic models and monitoring data to predict the areas with potential for natural eelgrass recovery. In this report, we have used only modelled results as input for the GIS-tool.

The different data layers (Table 2) are imported into GIS, assuming that the specific data layer represents the parameter for each model grid. Overall, the different data layers include general ecological data as well as processes that are critical for eelgrass recovery. The GIS model gathers the information from the various data layers, and hence, accumulates multiple stressors into areas with potential for natural eelgrass recovery. The input layers are reclassified into 5 categories according to their impact on the eelgrass recovery process: 1) Optimal recovery, 2) Good recovery, 3) Threshold for recovery, 4) Poor recovery, and 5) Very poor recovery. These 5 categories are explicitly defined for each parameter included in the GIS-tool and are based on scientific knowledge (see Flindt et al. (2016) for more details). The thresholds for individual stressors and how they are defined are summarized in Table 1. The data included in the GIS-analysis is shown in Table 2. The present GIS tool includes two minor changes with respect to that described by Flindt et al. (2016). Lugworm (*Arenicola marina*) was not included in these analyses (due to lack of data), and the DIN categories have been adjusted due to improved knowledge supported by Canal-Vergés et al. (in prep.). The changes in DIN categories are, however, minor.

This GIS tool provides with two results, areas with potential for vegetative growth and areas with potential for sexual reproduction. Vegetative growth is calculated after the weighted overlay of all the ranked environmental conditions, where light availability and eelgrass conditions weighted 20 % and all other factors weighted 10 % each. The necessary conditions for sexual reproduction are more restrictive than those for vegetative growth. Therefore, the tool assumes that there cannot be successful sexual reproduction (survival of seedlings) in areas with very poor critical share stress, sediment conditions (LOI), resuspension, abundances of opportunistic or non-opportunistic macroalgae as well as in areas with very poor or poor oxygen and light availability conditions. As stated, if all environmental conditions are correct, bare sediment areas can be populated by seedlings even with no neighboring eelgrass beds are present, hence areas with very poor eelgrass densities were not zeroed out (Flindt et al. 2016).

Table 1. The individual stressors and their thresholds for obtaining optimal, good, restricted, poor and very poor recovery for eelgrass (Flindt et al. 2016).

		Recovery				
Parameter (layer)	Unit	Very poor	Poor	Restricted	Good	Optimal
T_{wc}	N m^{-2}	>1	0,7-1	0,5-0,7	0,2-0,5	0-0,2
Sediment LOI	%	> 10	5-10	2-5	1-2	0-1
DIN*	µg N l^{-1}	> 150	75-150	40-75	25-40	0-25
Resuspension	Frequency	> Daily	Daily	Monthly	Biannual	< Biannual
Benthic light	µE m^{-2} s^{-1}	0-100	100-200	200-300	300-400	> 400
O_2 limitation	Period	3 Week^{-1}	2 Week^{-1}	Weekly	Monthly	< Montly
Opp. Macroalgae	gC m^{-2}	> 26	13	10	6	< 2
Non-opp. Macroalgae	gC m^{-2}	> 26	13	10	6	< 2
Lugworm	g WW m^{-2}	>50	40	25	10	<9
Eelgrass	gC m^{-2}	< 3	< 7	< 14	< 28	> 28

Tool 2 is a combined GIS- and a visualization model performing gradient analysis of nutrients, marine primary production and growth limitation by nutrient availability. Here, the water body is split into several sub-polygons along the eutrophication gradient. The number of sub-polygons is determined by water body specific properties such as bathymetry and hydrology. In Vejle Fjord we used 5 sub-polygons to analyze model results along the eutrophication gradient (Figure 5).

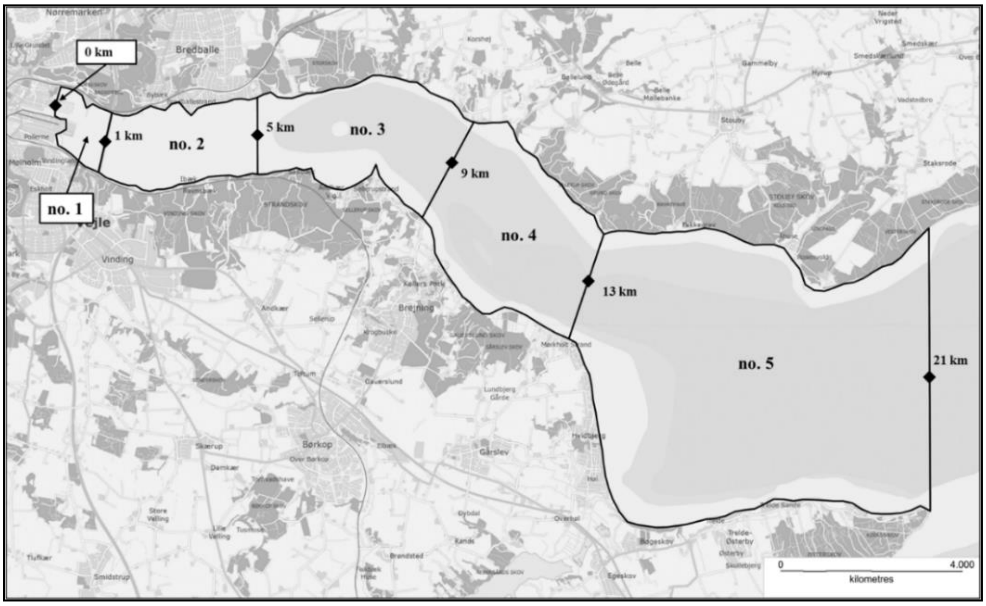

Figure 5. *Vejle Fjord, Denmark*. The eutrophication gradient is shown as polygons numbered from 1 to 5 and as distance from the inner-most part of the fjord measured at the black diamonds. Scalebar is 4 kilometers.

The extracted parameters (Table 2) was imported into the tool as separate GIS layers and were compiled into a master table containing all physical (bathymetry, area, volume, sub-polygon no. etc.) and ecological parameters (biomasses, production rates, nutrient concentrations etc.) for every finite element polygon (i.e., 2,367 and 13,930 rows for VF_{RBMP} and VF_{HiRes}, respectively). From within each sub-polygon the parameters were extracted from the master table and recalculated as area- or volume specific values for every depth interval (0.5 m) and thus creating a depth-specific mass budget for each sup-polygon. The mass-budgets enables for extracted of vertical profiles and recalculations of parameter values into mean values covering the entire sub-polygons.
The visualization module was built in the open-source software R and designed to use the mass-budgets as an input, make recalculations and create specific graphs and tables as the output. The model was built to ensure methodological consistency on the recalculation and visualization of the mass-budgets across different water bodies and with a varying number of gradient polygons. The model structure consists of three sections compiled into a single R function: 1) data import, 2) mass-budget reordering and recalculation, 3) creating and exporting graphs and tables.

Table 2. List of ecological parameters extracted from the Mike model result files and used in the GIS analysis.

Description	ID	Unit	Timeframe	Statistical calculation
Eelgrass biomass, C	EC	g C m^{-2}	GS	Max
Eelgrass biomass, N	EN	g N m^{-2}	GS	Max
Eelgrass biomass, P	EP	g P m^{-2}	GS	Max
Opportunistic macroalgae biomass, C	BC1	g C m^{-2}	GS	Max
Opportunistic macroalgae biomass, N	BN1	g N m^{-2}	GS	Max
Opportunistic macroalgae biomass, P	BP1	g P m^{-2}	GS	Max
Perennial macroalgae biomass, C	BC2	g C m^{-2}	GS	Max
Perennial macroalgae biomass, N	BN2	g N m^{-2}	GS	Max
Perennial macroalgae biomass, P	BP2	g P m^{-2}	GS	Max
Benthic diatom biomass, C	BDC	g C m^{-2}	GS	Max
Benthic diatom biomass, N	BDN	g N m^{-2}	GS	Max
Benthic diatom biomass, P	BDP	g P m^{-2}	GS	Max
Phytoplankton biomass, C	PC	g C m^{-3}	GS	Max
Phytoplankton biomass, N	PN	g N m^{-3}	GS	Max
Phytoplankton biomass, P	PP	g P m^{-3}	GS	Max
Chlorofyll-*a* concentration	CH	g chl *a* m^{-3}	GS	Max
Dissolved inorganic N	DIN	g N m^{-3}	GS	Mean
Dissolved inorganic N, bottom value	DIN_b	g N m^{-3}	GS	Mean
Dissolved inorganic P	DIP	g P m^{-3}	GS	Mean
Dissolved inorganic P, bottom value	DIP_b	g P m^{-3}	GS	Mean
Benthic light intensity	Io_b	E m^{-2} d^{-1}	GS	Mean
Secchi depth	SD	m	GS	Mean
Water temperature	T	°C	GS	Mean
Wave and current induced bed shear stress	T_{wc}	N m^{-2}	y	Max
Sediment wet bulk density	SedDens	g WW m^{-2}	GS	Mean
Sediment loss of ignition, sediment layer 1	LOI_1	%	GS	Mean
Sediment loss of ignition, sediment layer 2	LOI_2	%	GS	Mean
Time since last resuspension event	Resusp	no.	y	Mean
No. of days with O_2 conc. < 1.0 mg l^{-1}	SumDO1_1	d	1/3-2016	Accumulated
No. of days with O_2 conc.< 1.0 mg l^{-1}	SumDO1_2	d	1/10-2016	Accumulated
No. of days with O_2 conc.< 4.0 mg l^{-1}	SumDO4_1	d	1/3-2016	Accumulated
No. of days with O_2 conc.< 4.0 mg l^{-1}	SumDO4_2	d	1/10-2016	Accumulated
Eelgrass production	SumPREC_1	g C m^{-2}	1/3-2016	Accumulated
Eelgrass production	SumPREC_2	g C m^{-2}	1/10-2016	Accumulated
Opp. macroalgae production	SumPRBC1_1	g C m^{-2}	1/3-2016	Accumulated
Opp. macroalgae production	SumPRBC1_2	g C m^{-2}	1/10-2016	Accumulated
Per. macroalgae production	SumPRBC2_1	g C m^{-2}	1/3-2016	Accumulated
Per. macroalgae production	SumPRBC2_2	g C m^{-2}	1/10-2016	Accumulated
Benthic diatom production	SumPRBDC_1	g C m^{-2}	1/3-2016	Accumulated
Benthic diatom production	SumPRBDC_2	g C m^{-2}	1/10-2016	Accumulated
Phytoplankton production	SumPRPC_1	g C m^{-2}	1/3-2016	Accumulated
Phytoplankton production	SumPRPC_2	g C m^{-2}	1/10-2016	Accumulated

Note. The units are the initial units predefined by the Mike model. Some parameters e.g., benthic light intensity, were converted into other units during dataprocessing in the GIS-tool. For sumDOX and SumPRXX parameters the SumX_1 was substracted from SumX_2 in order to get accumulated values in the growth season.

2.3 Results

By introduction enhanced spatial resolution to the model domain in VF_{HiRes} apparent improvements in depth characteristics are seen in the target depth area defined by the eelgrass depth limits (Table 3). VF_{HiRes} shows a large gain in accumulated shallow area at depths between 1-5 m compared to the VF_{RBMP} model. Especially sub-polygons 1, 3, 4 and 5 shows an apparent signal (Figure 6). When considering the entire depth range of the fjord the area at depth from 0-5 m constitutes 14.7 and 15.1 km^2 for VF_{RBMP} and VFHiRes, respectively. Thus, by introducing a higher resolution on the model bathymetry, the shallow area in the fjord increases by 0.4 km^2 (or about 3%).

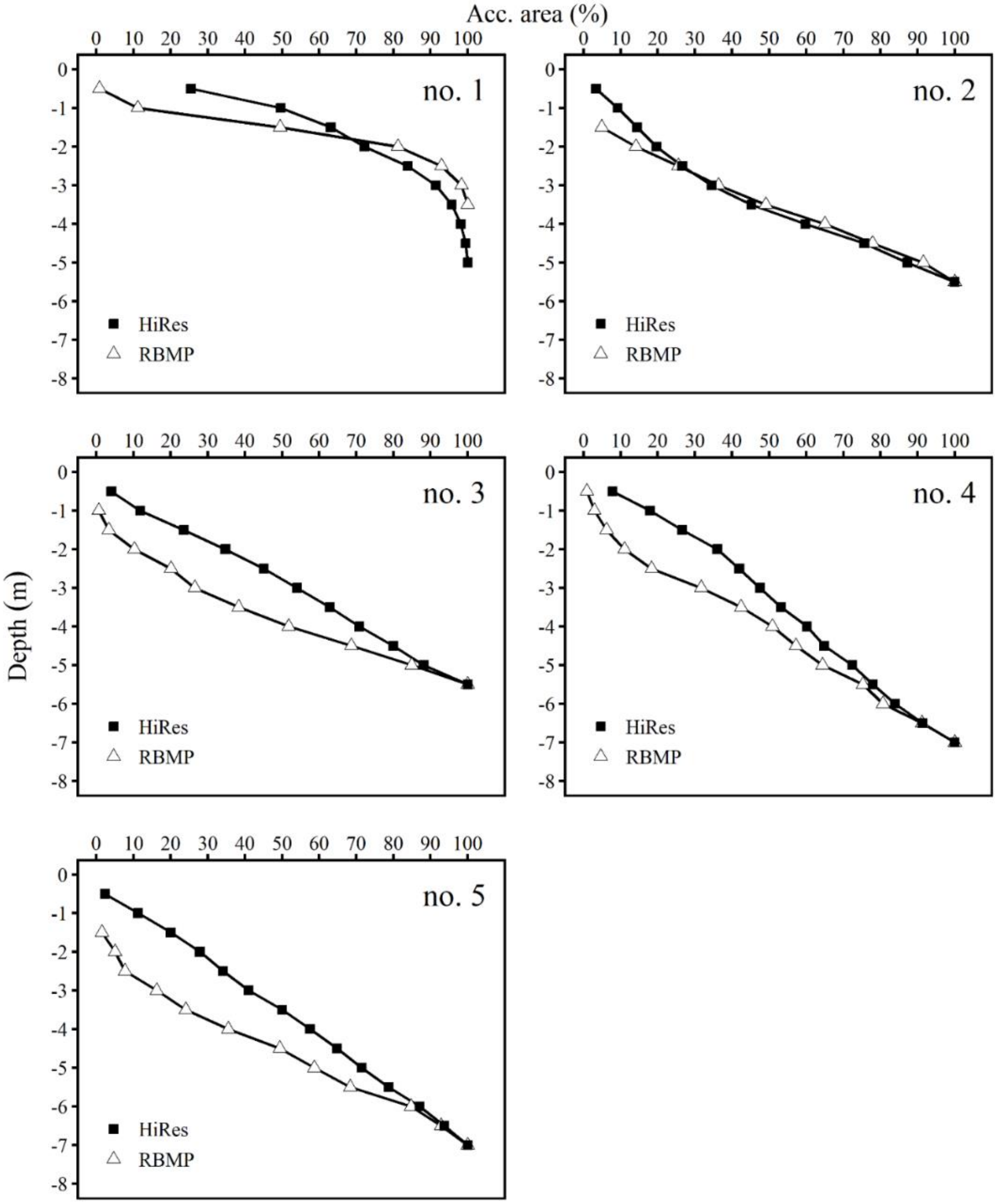

Figure 6. *Vejle Fjord, Denmark*. Hypsographs showing the accumulated area as a function of depth. Only the area within the target depth limit presented in Table 3 is included in the figure. From within each sub-polygon the baseline scenario depth characteristics of the two Vejle Fjord models are compared. $VF_{\_RBMP}$ (open triangles) and VF_{HiRes} (closed squares).

2.4 Discussion and conclusions

The inner part of Vejle Fjord is shallow, while the outer part has extended deep areas with small scattered shallow areas near the coast. Moving from the shores the depth increases rapidly due to step bathymetric slopes. Modeling this type of water body requires a very refined bathymetric resolution to catch the shallow areas on the bathymetric slopes. For use in the GIS-tools, the finite elements grid cells are pooled in depth intervals (e.g., 0.5 m intervals) based on the model bathymetry and thus, by using a coarse resolution grid cells may cover several contour lines along a bathymetric slope but only contain a single depth value rendering the model bathymetry inaccurate. As shown, the VF_{HiRes} model provides with an improved bathymetric solution compared to the VF_{RBMP} model. This is especially important as the main focus here is to model the effect of different scenarios on the productive shallow areas of Vejle Fjord. In addition, with increased shallow area the potential for benthic production also increases rendering the model more sensitive to local changes in nutrient loadings.

3 Vejle fjord River Basin Management Plan - reduction scenarios

When preparing the Danish River Basin Management Plans 2015-2021 (RMBP 2015-2021), DHI and Aarhus University (AU) developed a number of mechanistic (DHI) and statistical (AU) models that were used for calculating chlorophyll-a target values defining the threshold (GM) between 'Good Ecological Status' (GES) and 'Moderate Ecological Status'. The models were also used for calculating Maximum Allowable Inputs (MAIs) of total nitrogen (N) from Danish catchments based on chlorophyll-a threshold values and a proxy for eelgrass depth limit. Hence, the aim was both model development and development of a method for calculating the MAIs.
On a national scale the outputs of the mechanistic modelling was used to evaluate the effects of Danish nutrient loading reductions. In this section we present the results from a combined modeling- and GIS effort performed on Vejle Fjord evaluating the effect of non-coupled 30% nutrient reduction scenarios (30% reduction of Danish N and P land-based loadings).

3.1 Methods

For this section, the GIS model tools already presented (section 2.2.3) was applied to model results harvested from three different scenarios simulated by the VF_{RBMP} model (section 2.2.1). The model scenarios include 1) a baseline (present status), 2) a 30 % reduction of land-based nitrogen (N-30%) and 3) a 30% reduction of land-based phosphorous (P-30%).

3.2 Results

As a starting point for the analysis, all data from Table 2 have been lumped and assessed in sub-polygons, where the aim was to perform environmental gradient analysis. Figure 5 shows the sub-polygons for Vejle Fjord, and as can be seen from the figure, Vejle Fjord includes five sub-polygons. Instead of a standard areal GIS data analysis, the assessment sub-polygons allow for illustrations of the eutrophication gradient both vertically and horizontally from the innermost part of the estuary towards the open boundary.

3.2.1 Depth characteristics

This section aims to assess the effects of nutrient reductions in Vejle Fjord, with a specific focus on the spatial impact concerning eelgrass recovery. To assess the spatial impact, we need to consider the depth characteristics of the estuary. Hence, a hypsographs was extracted for the fjord. A hypsograph describes the depth distribution within a defined gradient polygon. Figure 7 shows the hypsographs of the five sub-polygons in Vejle Fjord where the inner sub-polygons 1 and 2 are very shallow with close to 100% of the area being shallower than 5 meters, however, the inner fjord polygons are close to the main nutrient source and are highly eutrophicated. The depth gradually increases with distance from the inner fjord where about 25%, 15% and 10% of the sub-polygon area in polygons 3, 4 and 5, respectively, is shallower than 5 meters. Thus, only small stretches along the coast in Vejle Fjord is sufficiently shallow to support benthic vegetation while also providing with sufficiently low nutrient concentrations.

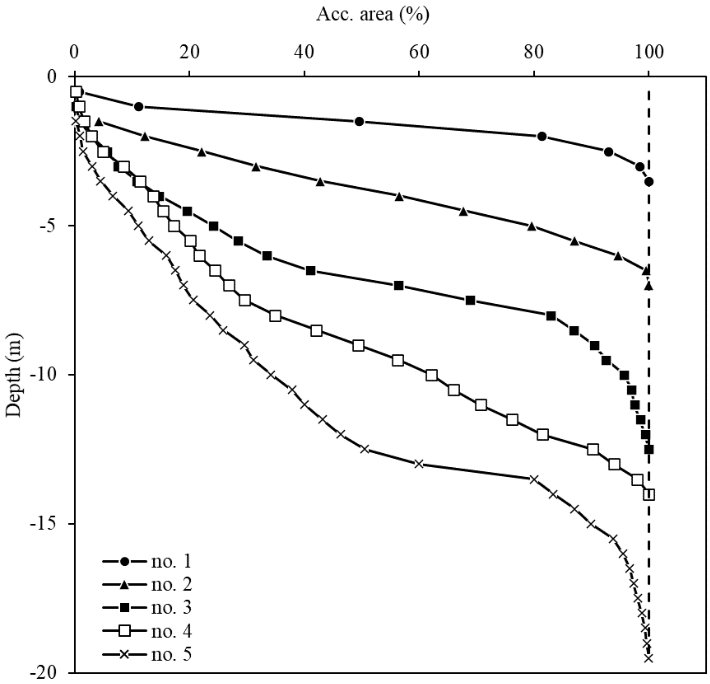

Figure 7. *Vejle Fjord, Denmark*. Hypsograph of the five gradient polygons presented in Figure 5. The hypsograph is based on the VF_{RBMP} model bathymetry.

The depth characteristics are essential and can illustrate the potential for eelgrass coverage (spatial distribution), and hence, supplement the intercalibrated indicator 'main eelgrass depth limit', which is one of the indicators used for assessing ecological status in Danish marine water bodies.
In Vejle Fjord, the first three sub-polygons correspond to the inner part (water body no. 123) of the estuary, and the two last sub-polygons correspond to the outer part (water body no. 122). The data for reference, target and status eelgrass depth limits are included in Table 3.

Table 3. Summary of the area (in %) in Vejle Fjord supported by eelgrass depth limits (*DL*) in a reference situation ($DL_{reference}$), at the threshold between good and moderate ecological status (GES) (DL_{target}) and at the observed depth limit (DL_{status}). The data is split into the inner part (sub-polygons 1-3, water body ID 123) and the outer part (sub-polygons 4-5, water body ID 122) (data from (Timmermann et al. 2020)).

Waterbody ID	Sub-polygon ID	Area (km^2)	DL_{status} (m)	DL_{target} (m)	$DL_{reference}$ (m)	DL_{status} (%)	DL_{target} (%)	$DL_{reference}$ (%)
Inner part (no. 123)	1	1.4				78	100	100
	2	5.5	< 1.9	< 5.7	< 7.6	11	91	100
	3	9.2				3	30	62
Outer part (no. 122)	4	11.5	< 2.6	< 7.3	< 9.9	5	28	61
	5	42.5				2	20	34

Note. The eelgrass depth limits are different in the inner part (no. 123) from the outer part (no. 122)

In summary, both reference and target depth limit support substantial areas in sub-polygons 1 and 2, whereas the target depth limit support much smaller areas than reference values in sub-polygons 3-5. Finally, evaluating the status depth limit, the supported spatial coverage is about 78%, 11%, 3%, 5% and 2% in sub-polygons 1-5, respectively (Table 3).

3.2.2 Pelagic versus benthic primary production

In the present report, we evaluate the effects of nutrient load reductions by an ecosystem-based approach. Here we analyze the changes in pelagic primary production to benthic primary production based on the existing biomasses of, e.g., eelgrass and macroalgae. The changes are primarily based on the simulated environmental conditions performed in the N-30% reduction scenario, as shown below in Figure 8. In the present state (baseline) almost no areas in the inner part of Vejle Fjord shows DIN concentrations below 75 µg N l^{-1} in the growth season (Figure 8). In the outer part, the DIN concentrations decrease to between 50-75 µg N l^{-1} and to between 25-50 µg N l^{-1} close to the fjord boundary. In the N-30% reduction scenario extended areas in Vejle Fjord are getting released from high DIN concentrations with DIN ranging between 50-75 and 25-50 µg N l^{-1} in sub-polygons 3-4 and 5, respectively.

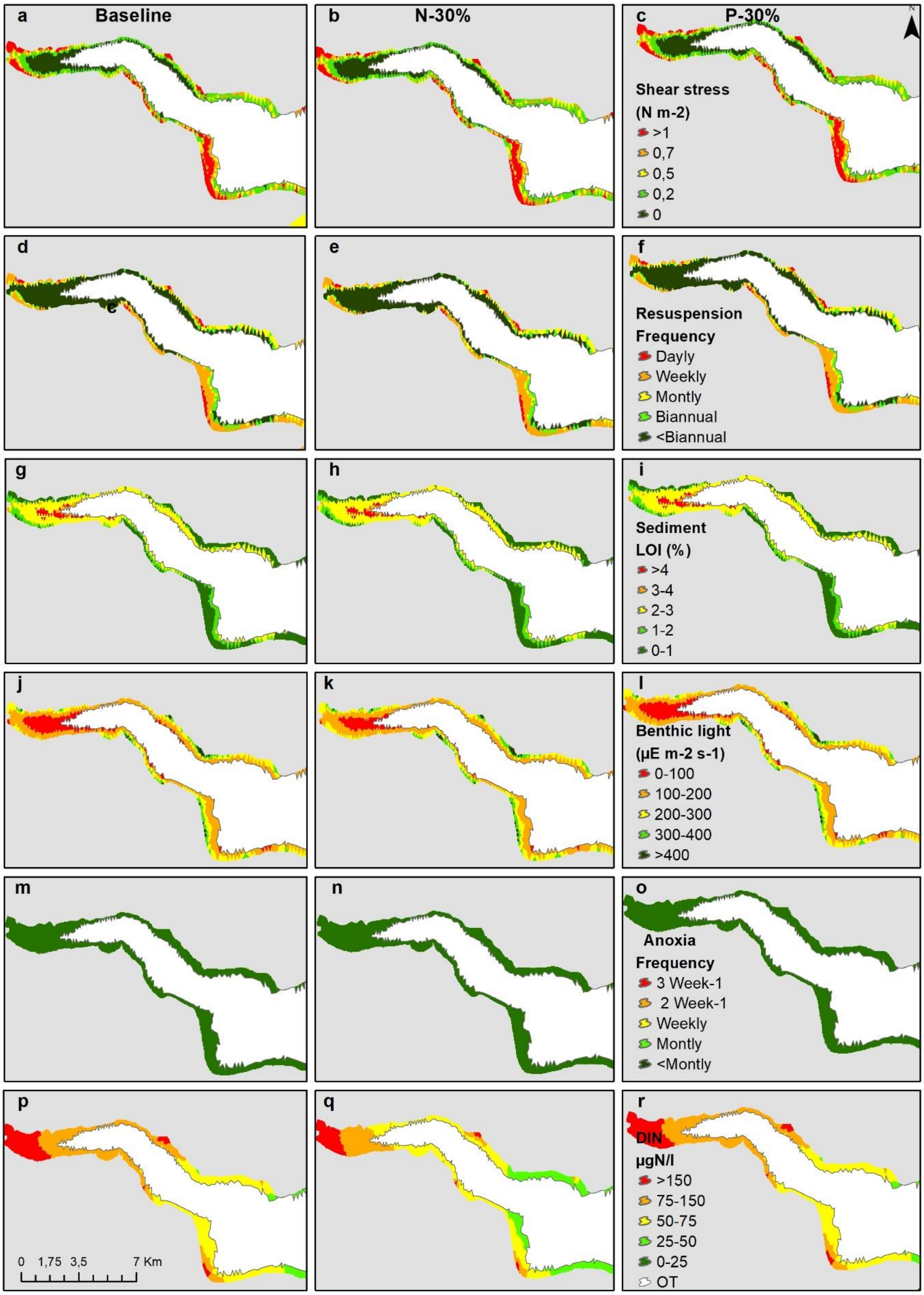

Figure 8. *Vejle Fjord, Denmark*. Abiotic reclassified parameters included in the GIS hotspot analyses for areas with vegetative and sexual reproduction potential for Vejle fjord baseline, N-30% and P-30% scenarios. a) Shear stress induced by waves and current (Twc), d) Resuspension frequency, g) Sediment stability and uprooting of eelgrass seedlings (LOI), j) Benthic light intensity, m) Frequency of anoxia, p) Dissolved inorganic Nitrogen (DIN). * OT areas with depths out from targeted areas (WFD) (Table 3).

Ecosystems subject to eutrophication are characterized by increased pelagic primary production, resulting in increasing pelagic biomass, increased sedimentation and deteriorated light conditions at the seabed and consequently reduced biomass and productivity of benthic vegetation and benthic diatoms. These processes are well known (Andersen and Conley 2009; Chislock et al. 2013; Flindt and Kamp-Nielsen 1997; Flindt et al. 1999; Neto et al. 2008).

When the rooted benthic vegetation is lost, the re-establishment is difficult and associated with substantial time lag. This goes for all ecosystems, including shallow-water ecosystems. In Figure 9 and Figure 11, we evaluate the ecosystem response to nutrient reductions in the different sub-basins in Horsens Fjord described by the response in benthic to pelagic primary production.

The eelgrass primary productions (Figure 9, A) increase from zero g C m^{-2} GS^{-1} in sub-polygon no. 1 to just below 20 g C m^{-2} GS^{-1} in sub-polygon no. 6 and are related to the decreasing DIN concentration from sub-polygon no. 1 towards sub-polygon no. 6. There is almost no difference in eelgrass primary production between the baseline and the P-30% scenario. In the N-30% reduction scenario, DIN concentration decreases in the inner part of the estuary and the eelgrass primary production increases slightly in sub-polygons no. 3-4.

With respect to the opportunistic macroalgae (Figure 9, B), the primary production is around 30 g C m^{-2} GS^{-1} in the inner fjord decreasing to just above 15 g C m^{-2} GS^{-1} in the outer fjord where production becomes growth limited by the low DIN concentrations. A peak in production is, however, seen in sub-polygons no. 3-4 at about 25 g C m^{-2} GS^{-1}. Again, we see almost no response to P-reductions. In contrast, the N-30% increases the opportunistic macroalgae production in sub-polygons 1 and 2 and with a slight decrease in primary production in sub-polygons 3-5.

As was seen for eelgrass and opp. macroalgae production almost no change in phytoplankton production is seen between baseline and the P-30% reduction scenario (Figure 9, C). The primary productions start at almost 200 g C m^{-2} GS^{-1} in sub-polygon no. 1 and increases to about 250 g C m^{-2} GS^{-1} in sub-polygon 2. The phytoplankton production gradually decreases throughout sub-polygons 3-4 to about 70 g C m^{-2} GS^{-1} in sub-polygon 5. In the N-30% scenario, the primary production is reduced by about 50 g C m^{-2} GS^{-1} in sub-polygons 1-3 and about 25 g C m^{-2} GS^{-1} in the outer sub-polygons 4 and 5.

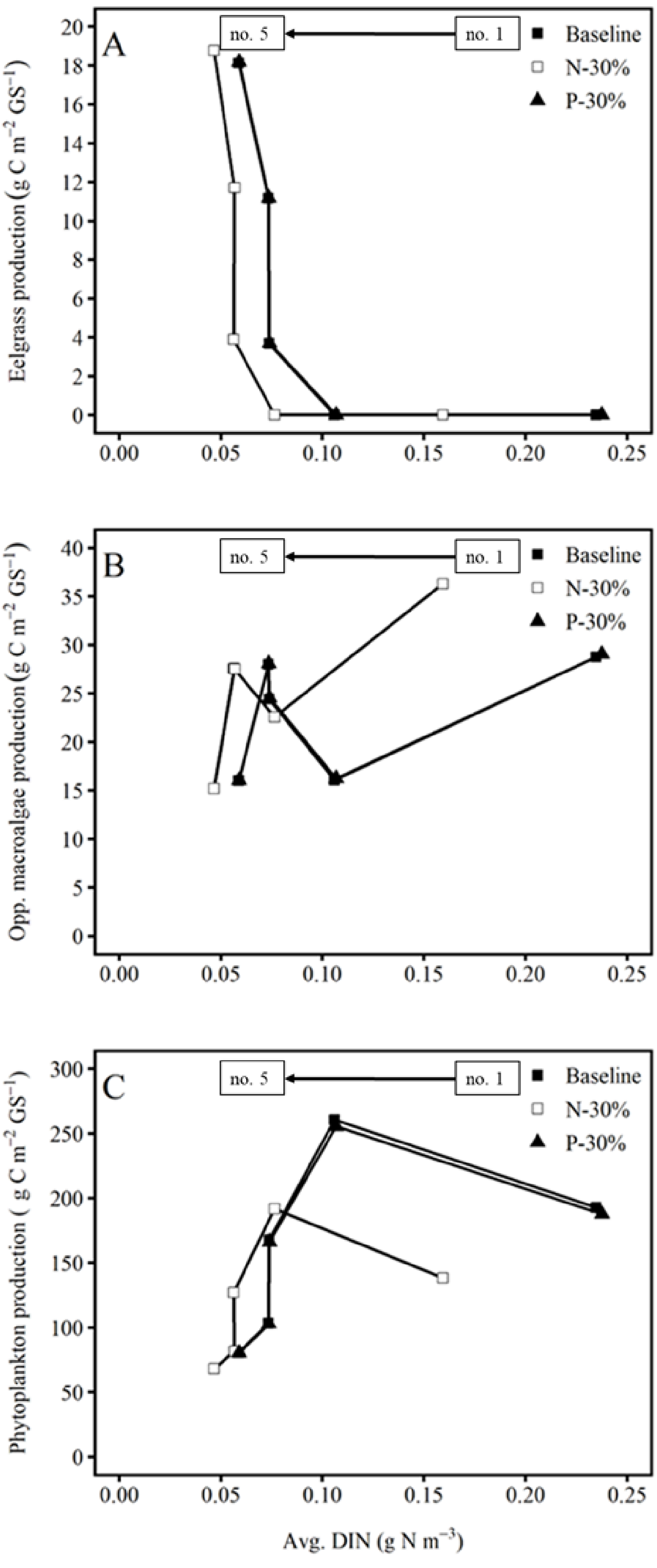

Figure 9. *Vejle Fjord, Denmark*. Area-specific growth season production of eelgrass, opportunistic macroalgae and phytoplankton as a function of Avg. DIN along a eutrophication gradient simulated by three model scenarios: Baseline (solid squares), 30% nitrogen reduction (open squares) and 30% phosphorous reduction (solid triangles). Each point on the graph represents a gradient polygon (1 to 5) moving from right to left. Reference Figure 5 for distances.

The results in Figure 9 are strongly influenced by the initial biomasses in the baseline or the scenarios, respectively. The modelled biomass of the different benthic vegetation components (eelgrass, opportunistic macroalgae and the perennial macroalgae) is shown in Figure 10.

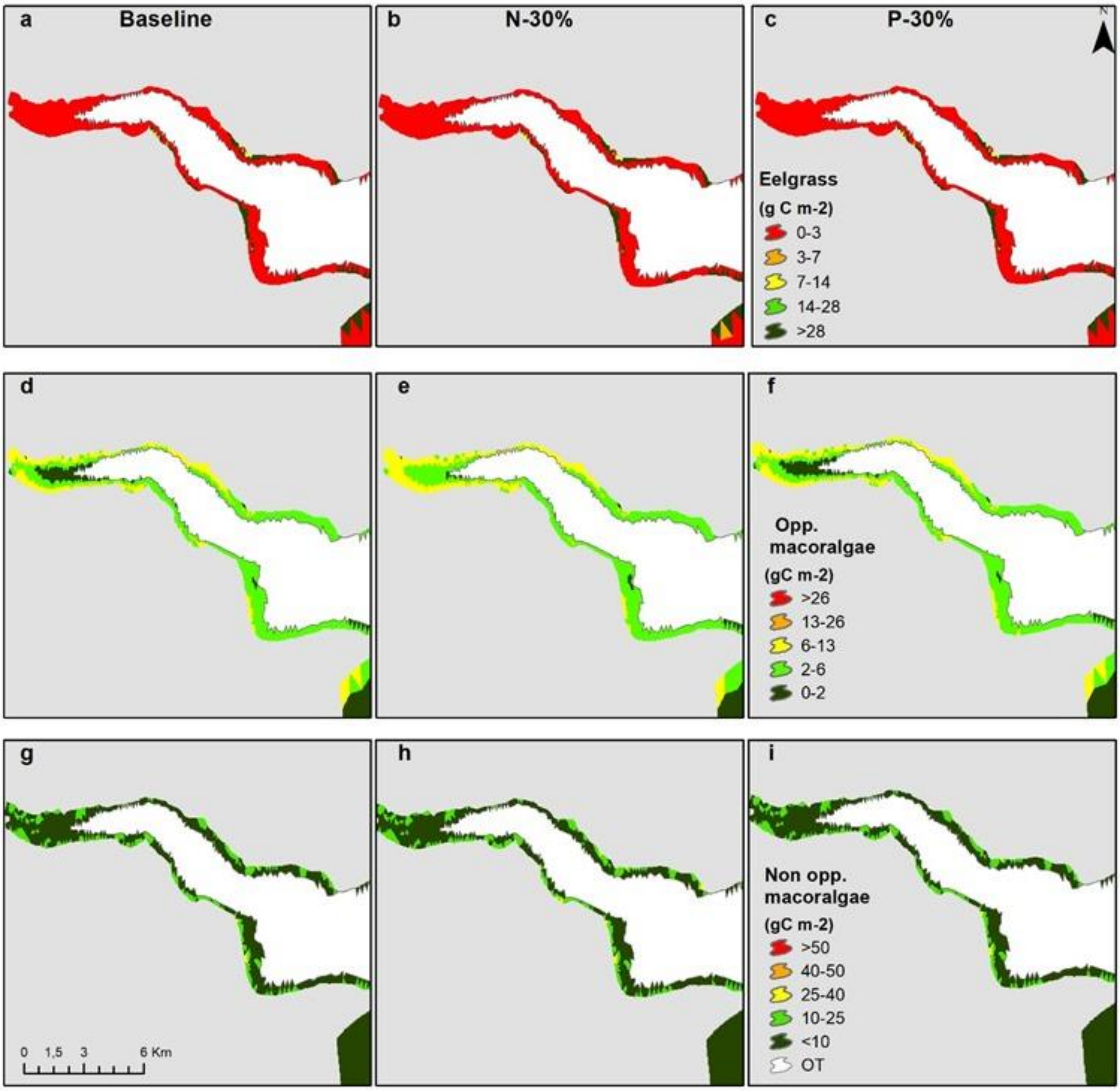

Figure 10. *Vejle Fjord, Denmark*. Biotic reclassified parameters included in the GIS-hotspot analyses for areas with vegetative and sexual reproduction potential for Vejle Fjord baseline, N-30% and P-30% scenarios. a) Concentrations and eelgrass coverage, d) Opportunistic macroalgae, g) non-opportunistic mobile macroalgae. * OT areas with depths out from targeted areas (WFD) (Table 3).

The initial biomasses of eelgrass in Vejle Fjord are relatively low (Figure 10) with a few narrow stretches (model grid cells) in sub-polygons 3-5 and no to very low initial biomass in sub-polygons 1 and 2. The initial biomass of opportunistic algae covers a larger part of Vejle Fjord along the coast, and the perennial macroalgae have initial biomass along the coast but in a much narrower stretch than the opportunistic algae (Figure 10). Hence, in the N-30% reduction scenario, growth conditions for benthic vegetation are improved (less phytoplankton primary production and improved light conditions), but as the opportunistic algae are already present in larger biomasses, they initially benefit the most. Over time we will expect this to change when eelgrass slowly takes over. But the overall primary productions move from pelagic production towards benthic production, which is desirable from an ecosystem point of view.

This is illustrated in Figure 11. This figure shows the overall growth season primary production divided into pelagic (WC prod.) or benthic (Benthic prod.) production, respectively. The benthic production consisting of the summed production of eelgrass, perennial macroalgae species, opportunistic macroalgae species and benthic diatoms. The pelagic production was given solely by phytoplankton. The figure shows that the pelagic production is accountable for 70-80% of the total production in the innermost part of the estuary decreasing to about 60% in the outer part. The benthic production accounts for about 20-30% of the total production in the inner part increasing to about 40% in the outer part of the estuary. Almost no effect is observed in the P-30% scenario, but the primary productions are forced towards more benthic production in the N-30% scenario. Here, about 10% of the total primary production in sub-polygons 1-4 and slightly lower in sub-polygon 5 has shifted from pelagic to benthic production.

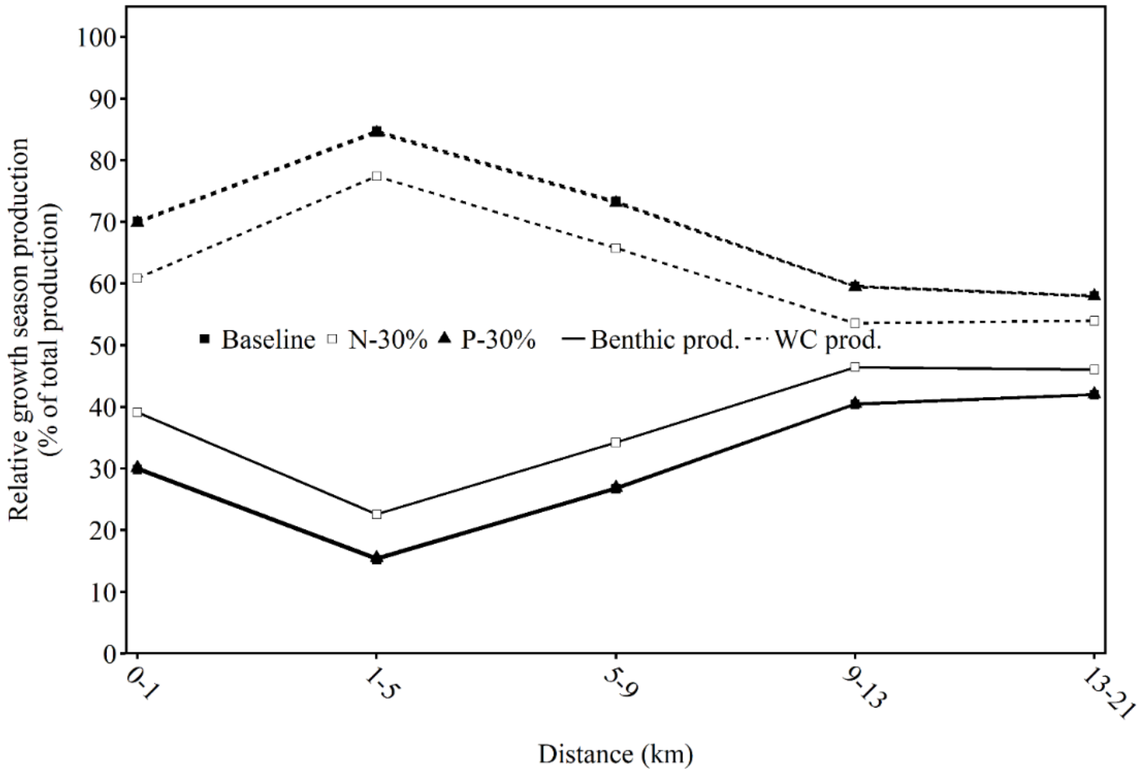

Figure 11. *Vejle Fjord, Denmark*. Growth season benthic (solid lines) and water column (stippled lines) production along a eutrophication gradient simulated by three model scenarios: Baseline (solid squares), 30% nitrogen reduction (open squares) and 30% phosphorous reduction (solid triangles). Production is shown as a relative (%) to the total production (benthic and water column production) in each of the eutrophication gradient polygons.

Overall, the primary production moves from the pelagic production towards benthic production in scenario N-30%, while no major changes in the system can be observed in scenario P-30%. In general terms, these changes from pelagic to benthic habitats are stronger/more visible towards the inner most parts of the estuary, where the eutrophication gradient increases.
Vejle Fjord has relatively few shallow areas and a deep central part (see the hypsograph in Figure 7), resulting in a relatively much smaller area suited for benthic vegetation compared to other Danish estuaries. This reduces the impact of N-30% and explains why the relative change from pelagic to benthic primary production is not more pronounced.

3.2.3 Eelgrass recovery potential

There are a number of stressors affecting the natural recovery and transplantation success for eelgrass in the majority of Danish estuaries (see (Flindt et al. 2016; Petersen et al. 2021) for a more detailed description). Based on model results the different stressors are ranged in classes. The threshold limits behind the different classes are shown in Table 1, and in the following, a GIS- analysis is carried out to identify areas where vegetative growth and sexual reproduction can be expected and where potential restoration can be expected to accelerate those processes (Flindt et al. 2016).

The data in this section is shown as percentage of the targeted depth limit area (Table 3, WFD). As an example, the targeted depth limit for the inner part of Vejle fjord is 5.7 m. Hence, for the inner fjord, the accumulated are from 0 to 5.7 m depth represents the 100%. If let's say light availability is good or very good in 10 % of the area, we are stating that only 10 % of the targeted depth limit area (and not of the entire fjord) has enough light to support eelgrass growth. In the rest of this report, we will refer to the targeted depth limit area as targeted area.

In Figure 8, the abiotic parameters (Twc, LOI, resuspension, bottom light, O_2 and DIN) affecting eelgrass recovery are shown as 2D plots, and the data coverage are summarized in Figure 12. In Figure 12, the data is structured in the area with bad or very bad conditions (red frame), the area with threshold conditions (yellow frame) and the area with good and very good conditions (green frame), and hence, the sum of the tree images in Figure 12 for each of the parameters equals 100% of the area (this area correspond to the accumulated area within the targeted depth limit for each fjord section).

In contrast to other Danish estuaries, Vejle Fjord has steep slopes and becomes deep, relatively close to the coast. Hence, the areas supporting eelgrass growth connected with the N- and P-reduction scenarios, as well as the total area with depth intervals shallower than target depths (WFD), show only little improvements. Thus, Vejle Fjord shows little potential eelgrass coverage at the target depth limit (< 30% in sub polygons 3-5, Table 3) while all other parameters (Table 1) were suitable.

The summary of abiotic and biotic parameters (Figure 12) clearly show that especially bottom light and present eelgrass biomass coverage keep the area in Vejle Fjord at target depth limit or shallower in bad or very bad conditions (more than 80%). The abiotic parameters LOI and DIN concentrations keep the estuary at threshold conditions, whereas Twc, resuspension, O_2, opportunistic and perennial macroalgae support eelgrass growth in large areas (more than 80%).

We see no, or minimal effects of reducing P-loads (P-30%) and very little improvements in the N-30% scenario. DIN and light conditions are improved in less than 15 and 5 % of the total targeted area respectively.

In Figure 10, we observe that opportunistic and perennial macroalgae are present along most of the coastline, but only few narrow stretches suggest eelgrass biomass, and in the innermost part of the estuary (sub-polygons 1-2), no eelgrass biomass exists at all.

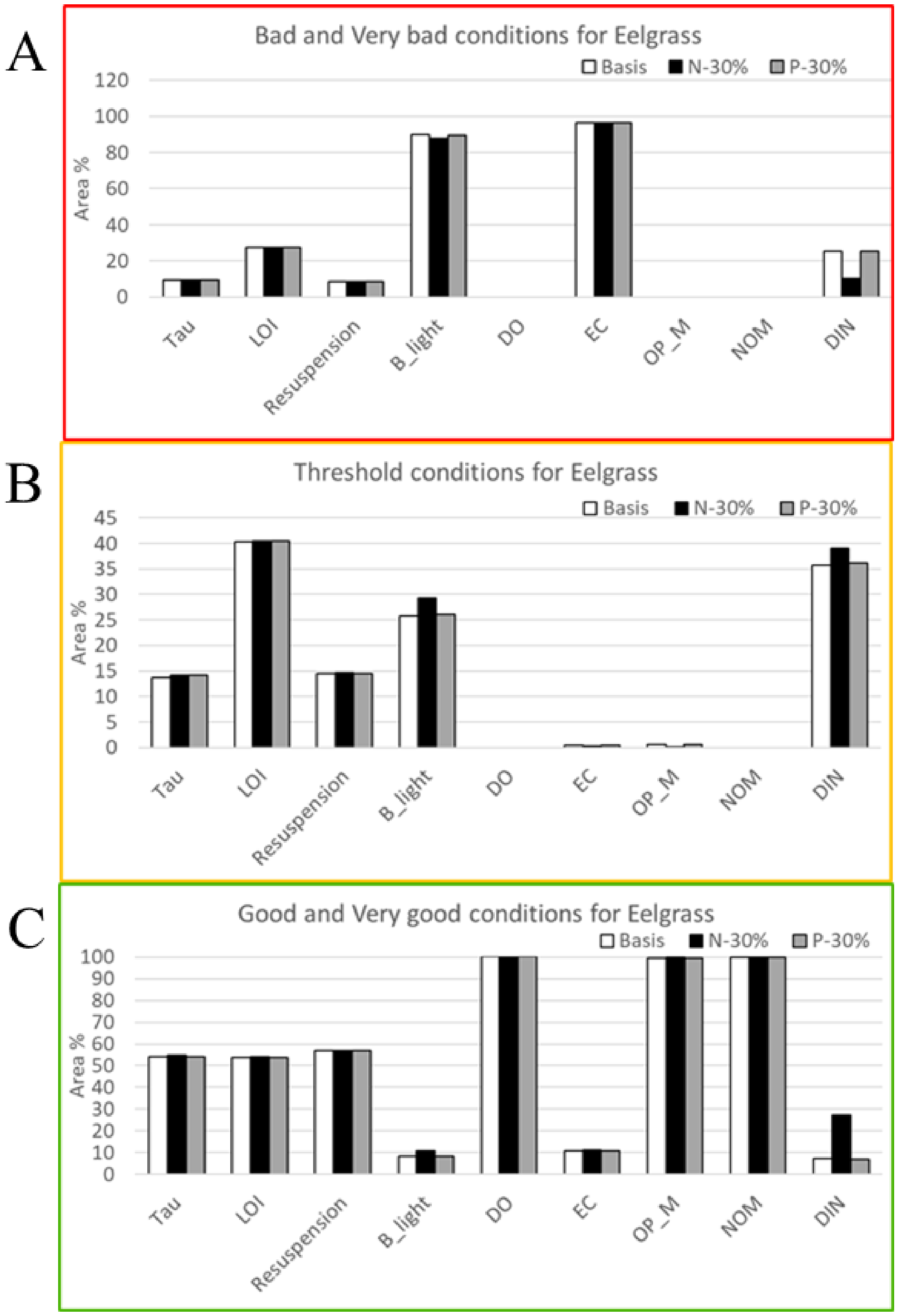

Figure 12. *Vejle Fjord, Denmark*. Parameter contributions to the GIS tool in Vejle fjord under baseline, N-30% and P-30% reduction scenarios. Tau: wave and current stress. LOI: Sediment organic content, Resuspension: frequency of resuspensions, B_light: benthic light availability, DO: anoxia frequency, EC: Eelgrass carbon, OP_M: Opportunistic macroalgae, NOM: non-opportunistic macroalgae, DIN: Dissolved inorganic nitrogen. A) Proportion of the targeted area (%) with bad or very bad conditions for eelgrass growth depending on the modelled parameters Tau, LOI, Resuspension, B_light, DO, EC, Op_M, NOM, and DIN. B) WFD target area (%) with threshold conditions for eelgrass growth depending on the modelled parameters Tau, LOI, Resuspension, B_light, DO, EC, Op_M, NOM, and DIN. C) WFD target area (%) with good or optimal conditions for eelgrass growth depending on the modelled parameters Tau, LOI, Resuspension, B_light, DO, EC, Op_M, NOM, and DIN.

What is clear from Figure 12 is also evident in Figure 13, where we observe slight improvement (increased area) supporting vegetative growth and sexual reproduction, but as summarized in Figure 14 and Figure 15, the potential vegetative growth and sexual reproduction areas are low, ~10 % of the existing areas under targeted depth. Reduction scenarios improvements are minor, less than 1 % of the total targeted area. The improvements in sexual- and vegetative reproduction by reducing N by 30% shows the largest signal in sub-polygon 3 (Figure 16 and Figure 17) as vegetative- and sexual reproduction improves by about 4% and

7%, respectively. The other sub-polygons only show little or no improvements in vegetative- and sexual reproduction following the N reduction. Note that the target depth limits increase from 5.7 m in the inner part to 7.3 m in the outer part (sub polygons 4 and 5). This can explain the lack of signal in sub-polygons 4 and 5.
However, it is important to remember that especially Vejle Fjord will benefit from N-reductions in neighboring countries which is not assessed in the present report (Erichsen et al. in prep-a; in prep-b).

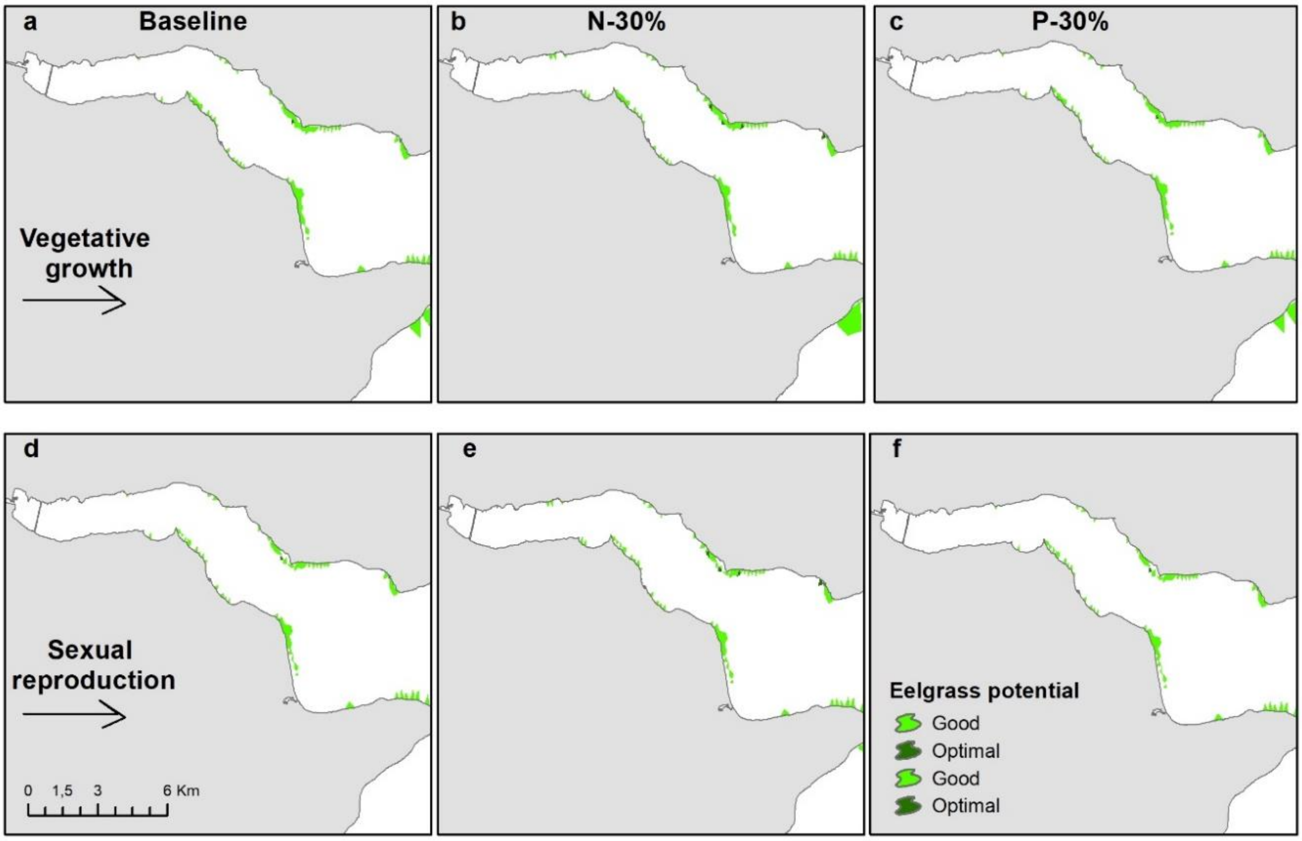

Figure 13. *Vejle Fjord, Denmark*. GIS predicted locations with potential for eelgrass reestablishment at baseline (left), after 30% N-reduction (middle column) and after 30% P-reduction in Horsens fjord. Locations with potential for eelgrass transplantation actions are shown in the upper row, while areas with potential for eelgrass seedlings survival areas shown in the lower row.

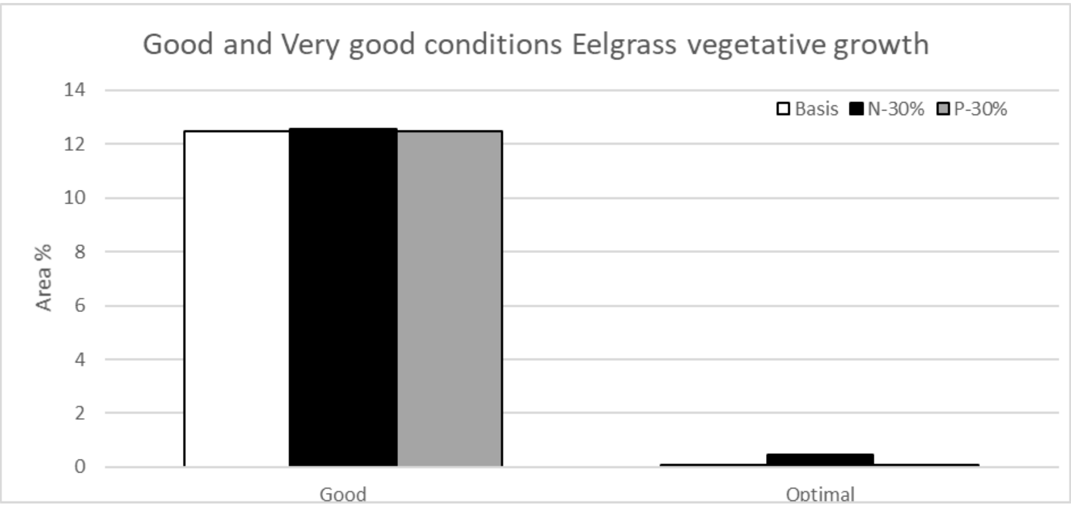

Figure 14. *Vejle Fjord, Denmark*. GIS predicted accumulated proportion of the targeted area (%) with potential for eelgrass reestablishment by vegetative growth in the baseline, after 30% N-reduction and after 30% P-reduction scenarios in Vejle fjord.

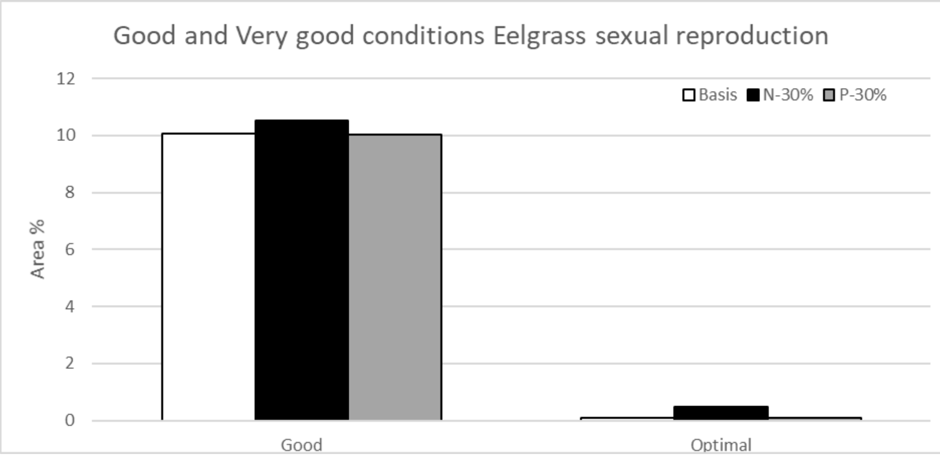

Figure 15. *Vejle Fjord, Denmark*. GIS predicted accumulated proportion of the targeted area (%) with potential for eelgrass reestablishment by sexual reproduction in the baseline, after 30% N-reduction and after 30% P-reduction scenarios in Vejle fjord.

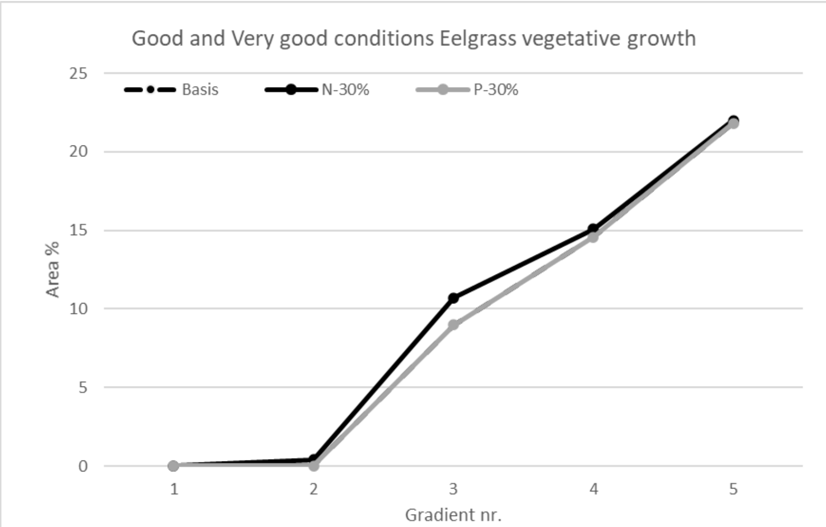

Figure 16. *Vejle Fjord, Denmark*. GIS predicted accumulated proportion of the targeted area with potential for eelgrass reestablishment by vegetative growth in the baseline, after 30% N-reduction and after 30% P-reduction scenarios, calculated along a eutrophication gradient in Vejle Fjord.

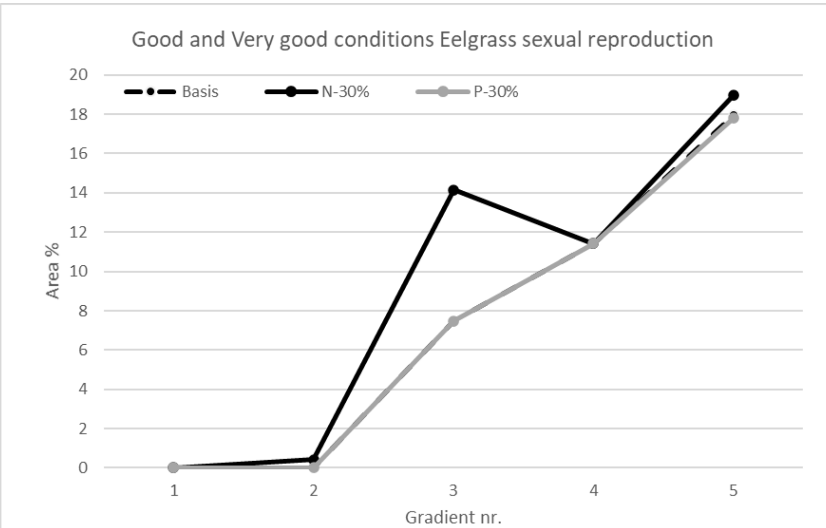

Figure 17. *Vejle Fjord, Denmark*. GIS predicted accumulated proportion of the targeted area with potential for eelgrass reestablishment by sexual reproduction in the baseline, after 30% N-reduction and after 30% P-reduction scenarios, calculated along a eutrophication gradient in Vejle Fjord.

3.3 Conclusion

As stated in the introduction to this report, the overall aim of the present study is to evaluate the effects of nutrient reductions in shallow water areas regarding the potential for eelgrass recovery, including an assessment of the abiotic and biotic processes prohibiting eelgrass recovery and to relate the study results to the estimated maximum allowable nutrient inputs described in Erichsen et al. (in prep-a).

The most essential is that the nutrient reduction provide new benthic area with sufficient light intensity to support further nutrient immobilization by benthic production. The overall conclusion from the analysis shows that the suggested 30% N-reduction in Vejle Fjord is insufficient to increase the area supporting vegetative eelgrass growth. In this report, the natural recovery predicted withing the areas with eelgrass vegetative growth potential are subjected to the already existing eelgrass presence (existing eelgrass beds). Which has not been monitored, nor evaluated in the present report. If eelgrass starts to recover or is actively transplanted/restored in new areas, the ecosystem services provided by the growth will support further improved environmental conditions. According to Erichsen et al. (in prep-a), the suggested reductions might support the depth limit based on light availability, but the environmental improvements are not sufficient to support sexual-based recovery where seeds germinate and initiate vegetative growth to form eelgrass patches. In this process individual seedlings are very vulnerable and exposed to the mentioned stressors (Valdemarsen et al. 2010), while more self-protection is supporting mature patches at deeper waters. To compensate for the higher loss rates of shoots during the recovery process a higher light intensity is needed. An example is Odense Fjord, where SDU have performed transplantations at 2.5 m’s depth in the outer part. Although natural patches are growing at more than 3 m’s depth the net results of the transplantation activity were a slowly decline in shoot densities, due to the light intensity not supporting sufficient shoot production to compensate for the losses (unpublished results). It is these results that together with literature data provided the relative high threshold for benthic light intensities in Flindt et al. (2016). These results suggest that considerably more reductions are needed to allow for a self-sustained reproduction of eelgrass in those three estuaries.

Based on the present analysis, we conclude that N-30% will improve the water quality in Vejle Fjord, allowing successful restoration in specific areas. However, this will only work if the reductions from Erichsen et al. (in prep-a) are implemented. This is supported by experiences from multiple transplantation attempts in different estuaries – here we only see some success when transplanting eelgrass in the outer part of the systems, far from where the nutrient loading typically is released.

Hence, we conclude that the reductions suggested in Erichsen et al. (in prep-a) are needed, but we cannot conclude that they are sufficient to ensure a self-maintained eelgrass propagation and growth in Vejle Fjord, why they might be a minimum requirement to ensure GES based on summer chlorophyll-a and light availability alone.

As stated in the introduction to this report, we have evaluated impacts based on Danish nutrient reductions alone. The nutrient reductions applied are general reductions (30% reductions) and not the specific reductions suggested in Erichsen et al. (in prep-b), as the specific MAIs are still not known. Also, the present study does not evaluate additional impacts from nutrient reductions from neighboring countries. Nutrient reductions from neighboring countries will enhance the effects from Danish nutrient reductions, especially in the more open estuaries.

In Vejle Fjord, almost no improvements occur in the N-30% scenario, and almost no improvements are modelled in the P-30% scenario. However, the abiotic parameter DIN improves in N-30%.

Good and very good conditions are only observed in the outer and northern part of the estuary, whereas a larger part of the estuary, including sub-polygon no. 3 and to some extent also areas in sub-polygon no. 2, reaches threshold conditions. Hence, N-30% imposes improvements, but as the reductions reported in Erichsen et al. (in prep-a) are smaller than N-30%. Potential effects from reductions in neighboring countries are not included in the present analysis, but even though we do not expect the estuary to restore eelgrass meadows by itself through vegetative growth or sexual reproduction without any restoration measures implemented.

4 Overall Conclusions

In the current Danish RBMP, the eelgrass target depth for the inner and outer fjord is 5,7 and 7,3 m for the inner and outer fjord. However, the current eelgrass depth limit found in the fjord only reach to 1,9 and 2,6 m. Even if we do not consider eelgrass at all, our analyses shows that the current status of the fjord is not good. Analyzing RBMP model results in an area-based approach shows that Vejle fjord bad quality status is not only related to the lack of eelgrass recovery, but also the strong eutrophication the fjord is subjected to and the accumulated impacts in other element of the fjord (e.g., organic sediments, lack of hard substrate, lack of primary producers etc.) resulting from the many years of sustained eutrophication. Our analyses shows, a gradient across the fjord, where on the contrary of the RBMP (where outer fjord is shown to have worst condition than the inner), the inner fjord is in worst environmental status than the outer fjord. This is partly due to their relative differences with the reference status used for the inner and outer fjord. However, considering the fjord as one water body we can see a clear eutrophication gradient, which should be assessed more detail (and solved) before any improvements can be seen. Among others, the lack of standing livestock of primary producers with longer retention time (eelgrass and perennial macroalgae) in combination with the small area where light availability at the bottom is enough to allow the growth for this mentioned species leads to a very limited buffer capacity in the fjord at present.

The RBMP shows some improvements with a nitrogen reduction scenario, however those are uneven along the mentioned eutrophication gradient and in any case are insufficient to improve the fjord status to GES, even if we only focus on the increase of light availability (eelgrass depth limit). The analyses shows that other parameters such as too high exposure in bare sediments (inducing resuspension), too much organic sediment in the seabed (mud), and excessive levels of nitrogen and phosphorus (promoting ephemeral species such as phytoplankton and opportunistic species) does not favor either the overall improvements of the fjord. Our field studies corroborate these findings and suggest that this is not even a worst-case scenario, since we also find additional stressors not even considered in the model. Some of those are overpopulation of green crabs (*C. maenas*), lack of fish (leading to a lack of top to bottom predatory control), lack of stable substrate (no stones to provide anchor to perennial macroalgae species), lack of filter feeders (low presence of effective filter feeders such as blue mussels) etc.

Increasing the livestock of perennial primary production species will increase the overall buffer capacity of the system further increasing the recovery potential as well as making nutrient reductions more efficient that otherwise are likely to fail. Regarding the used model in RBMP, our analyses show that for Vejle fjord, due to its specific bathymetry (steep slopes), there is an overall oversimplification of the benthic areas (for the preset discussion below 5,7 and 7.3 m in the inner and outer fjord). The model developed by Sund Vejle fjord, has corrected this spatial definition issue. This correction has resulted on a large area gain of benthic areas, hence a larger area to potentially increase the buffer capacity to the system. While we expect that these changes will have a rather small effect in the overall fjord (due to the proportionally large deep areas), we expect to see a change at local scale and along the gradient of eutrophication. With this model we should be able to see the effects of local action or changes of loads in the near field. We should also be able to see the extent to which these changes get diluted towards higher depth (and no longer be a part of the overall fjord change). Finally, this model should allow us to be much more precise when determining the present buffer capacity of the fjord, as well as the potential gained by changes simulated (model scenarios). We expect these changes to be more effective if positive or visible if negative than those modelled in the RBMP. At last, in the RBMP, the loading of the fjord was pooled together, so that the model was loaded with different levels of TN/TP according to the existing seasonality linked to the different sources. In the present model, we can separate the sources, respecting its individual seasonality and position (in the eutrophication gradient along the fjord) to estimate relative impacts to the local and overall areas. This will give us an impression of the relative impact of the different nitrogen and phosphorus sources both on time (seasonality) and on space (eutrophication gradient along the fjord). These kind of analyses, open as well the possibility (in the future), for optimizing when possible, the source placement or seasonal patterns to minimize local impacts, which will improve the overall system not only with the direct effect, but improving the local standing buffer capacity over time.

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