KPP Ems-Dollard

Development of effect chain model
Results 2012

Deltares

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1206237-000

Deltares

Title
KPP Ems-Dollard

Client	Project	Reference	Pages
Waterdienst	1206237-000	1206237-000-ZKS-0002	87

Trefwoorden
effect chain model Ems Dollard hydrodynamics mud nutrients primary production habitat

Samenvatting
This report describes the activities carried out in the framework of the project 'KPP Ems-Dollard' aiming at the development and application of an effect chain model for the Ems-Dollard estuary. Progress on the following parts is discussed:

1 Hydrodynamic and mud transport modeling
2 Ecological modeling
3 Habitat suitability evaluation

The effect chain model is fully operational from a technical point of view. It has been applied for a historical scenario with reduced suspended sediment concentrations. For the historical situation, a significantly higher primary production rate is computed.

Referenties
KPP Eems-Dollard

Versie	Datum	Auteur	Paraaf	Review	Paraaf	Goedkeuring	Paraaf
2.0	Nov. 2012	dr. W. Stolte		dr. D.S. van Maren		ir. T. Schilperoort	
		dr. ir. J.T. Dijkstra		dr. F.J. Los			
		dr. ir. T. van Kessel	TVK	A.J. Nolte			
		dr. M.S.J. Chatelain					
		C. Spiteri PhD					

State
final

Contents

1 Introduction

The present report describes the activities carried out in 2012 in the framework of the project KPP Ems-Dollard. The objective of this project is to develop and apply an effect chain model of the Ems-Dollard, consisting of a hydrodynamic model, a fine sediment transport model, an ecological model on nutrients and primary production and finally a habitat model to assess the habitat suitability or changes herein for several target species.

The project KPP Ems-Dollard is a long-term project started in 2009. Since then, all activities have been reported in yearly progress reports (see list of references). This year (2012) is the last year of the project. The present report is an incremental report only describing the activities undertaken after completion of the 2011 report. For a concise overview of all activities and overall conclusions is referred to the end report.

Developments and applications will not stop after 2012. A new project has been initiated, (‘KRW Ems-Dollard’), in which the effect chain model developed in the present project will be further applied and extended.

The structure of this report is as follows. In Chapter 2 this year’s developments on hydrodynamics and fine sediment transport are discussed. In Chapter 3 improvements made to the ecological model are discussed. Also, results are presented on a comparison between two simulations, one representative for the present situation and another representative for a historic situation with a lower suspended sediment concentration. In Chapter 4 developments on habitat assessment in the Ems-Dollard are discussed. Finally, conclusions are drawn in Chapter 5.

2 Hydrodynamics and Mud Transport

2.1 Hydrodynamics

The following changes were made to the hydrodynamic model in comparison with the 2011 model:

- Adapted freshwater discharge
- Adapted sea boundary conditions
- Adapted bathymetry near Bocht van Watum and in upper Ems

These changes have been carried out in the framework of the project KRW Ems-Dollard. More details will therefore be reported in the KRW project (Van Maren et al., 2011 and 2012). Here only two main results on tidal propagation and salinity distribution are shown.

Figure 2.1 shows a comparison between modelled and observed water levels and tidal components at Huibertgat. Compared to earlier results, the model performance has been significantly improved as a result of the improved sea boundary conditions.

Figure 2.2 shows a comparison between modelled and observed salinity at Groote Gat Noord. The new results are only slightly different from the previous results and remain a few ppt too high compared to observations. Both computations show a similar seasonal trend as the observations, but at a too high salinity level.

Both the new and old results have been used for fine sediment transport computations, see the next section.

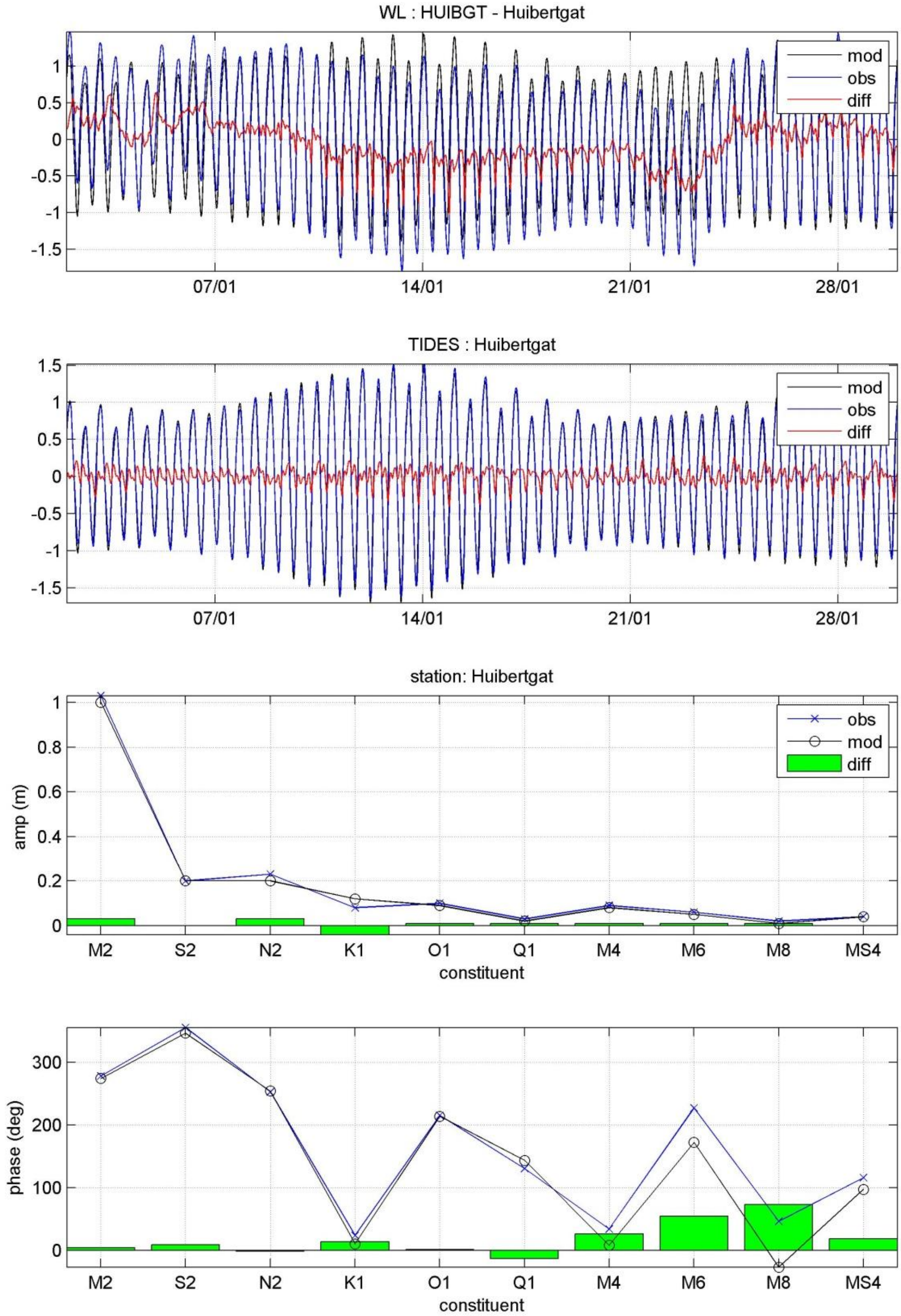


Figure 2.1 Result of calibration hydrodynamic model (runID 047. Comparison between modelled and observed water level (WL), tide, tidal amplitudes and tidal phases of the main tidal constituents.

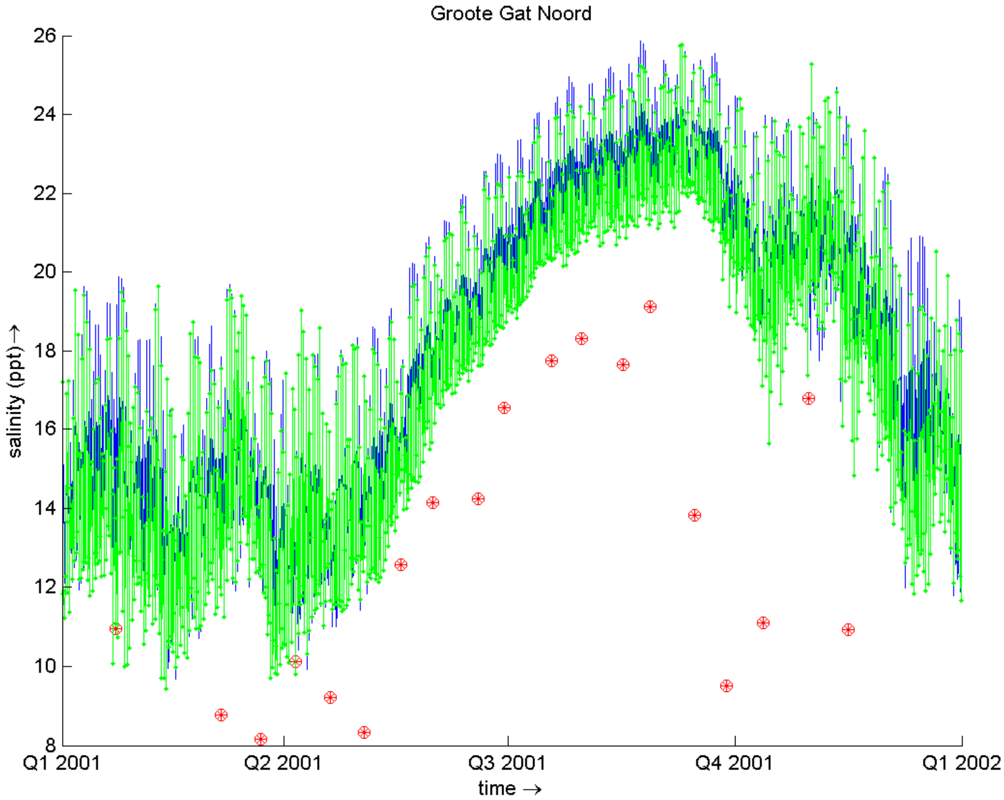


Figure 2.2 Modelled and observed salinity at Groote Gat Noord in 2001. Red dots: observations. Green dotted line: previous results. Blue line: new results. .

2.2 Mud transport

Based on the findings in 2011, the sensitivity of the suspended sediment levels to wave forcing was further investigated. Previous simulations were based on a saturated wave field in equilibrium with local depth for a given (time-varying) wind speed and (constant) fetch length. In 2012, assimilated waves were used. This technique consists of a combination of wave observations and wave computations. At locations where wave observations are available, they are used as direct input for the mud transport model. In this way, only a very limited number of grid cells of the numerical model are covered. The wave properties in all other grid cells are obtained by interpolation between the wave buoys. The interpolation algorithm uses year-average wave fields computed with SWAN projected on the grid of the mud transport model. This technique is described in more detail in the EIA Sand Mining Maasvlakte-2 (Royal Haskoning, 2006). The use of additional wave buoy data would further improve the accuracy of the wave forcing in the model locally, but no great benefit for the overall performance is expected.

Figure 2.3 and Figure 2.4 show the computed daily-average near-surface SPM concentration at Groote Gat Noord and Huibertgat, respectively. The blue lines represent the new simulation with assimilated waves (t18), the red line represents the original simulation with saturated waves (t16). The use of assimilated waves results in a somewhat increased variability, with higher peaks and lower troughs. However, based on the available low-frequency observation it difficult to quantify the change in model skill.

Figure 2.5 shows the spatial distribution of the year-average near-surface SPM concentration computed with assimilated wave fields (t18). Compared to the original simulation (t16), the sediment concentration gradient along the thalweg of the Ems estuary is slightly higher. In the new simulation, the average concentration at the North Sea is lower. This is explained as follows. Peaks in bed shear stress at the North Sea are higher for the assimilated wave field. To avoid too large SPM peaks, the critical shear stress for erosion has been increased at the North Sea in the new simulation. This results in lower SPM levels during calm weather and also in a lower year-average value at the North Sea. It is remarked that it is common that a new bed shear stress forcing may require a recalibration of a mud transport model.

Figure 2.6 and Figure 2.7 show the computed mud fraction in the seabed for simulations t16 and t18, respectively. Whereas the computed spatial distribution of the mud fraction resembles the observed one for simulation t16 (see Fig. 2.9 in Dijkstra et al., 2011), a much too low mud fraction is computed at tidal flats and shallow areas for simulation t18. A better agreement between computed and observed mud fraction could be obtained for simulation t18 after further recalibration (e.g. by changing the critical shear stress for erosion, the bed roughness for waves, or both). Until such recalibration has been carried out, simulation t16 is most suitable for use higher up in the effect chain.

The next step was to apply the new hydrodynamics (see previous section) to the mud transport model (simulation t19). Figure 2.8 and Figure 2.9 show the computed SPM concentrations at Groote Gat Noord and Huibertgat, respectively. Figure 2.10 shows the year-averaged spatial distribution of SPM concentration. From these figures it is clear that the use of the new hydrodynamics results in a deterioration of the model performance. Computed SPM levels decrease significantly and are clearly below observed levels. The most probable reason for this decrease is the use of a different numerical solver that is more stable but also more dissipative (higher numerical diffusion). The numerical scheme had to be changed as simulations with the original scheme became unstable in combination with the new hydrodynamic forcing. Based on these results, it was decided not to use results from the mud transport model based on the new hydrodynamics higher up in the effect chain model. Instead, results from the 2011 hydrodynamics have been used.

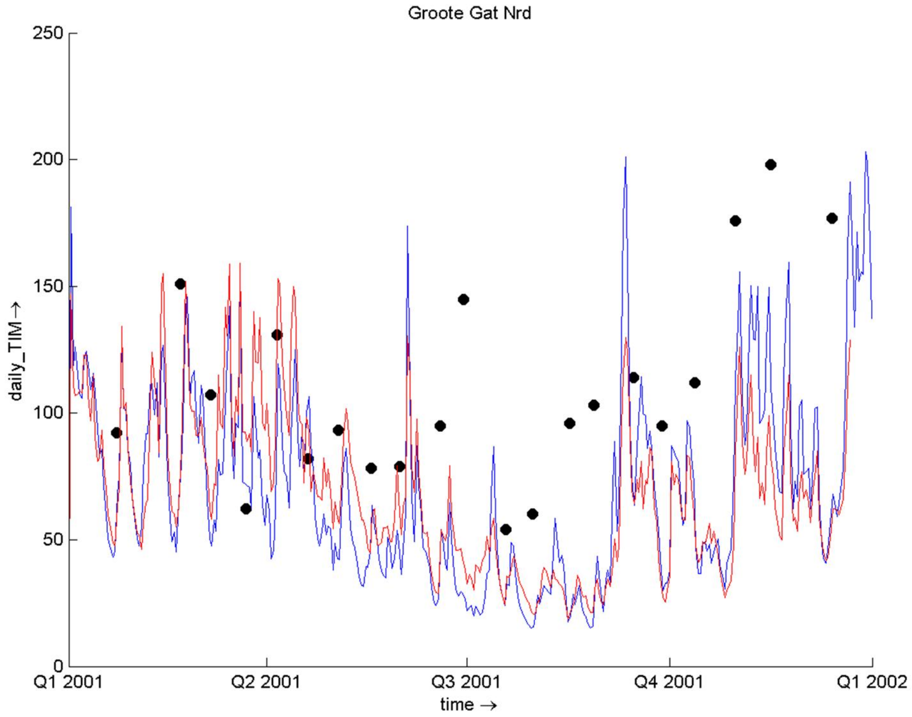


Figure 2.3 Daily-average near-surface SPM concentration (mg/L) at Groote Gat. Red line: old simulation (t16), blue line: with assimilated waves (t18).

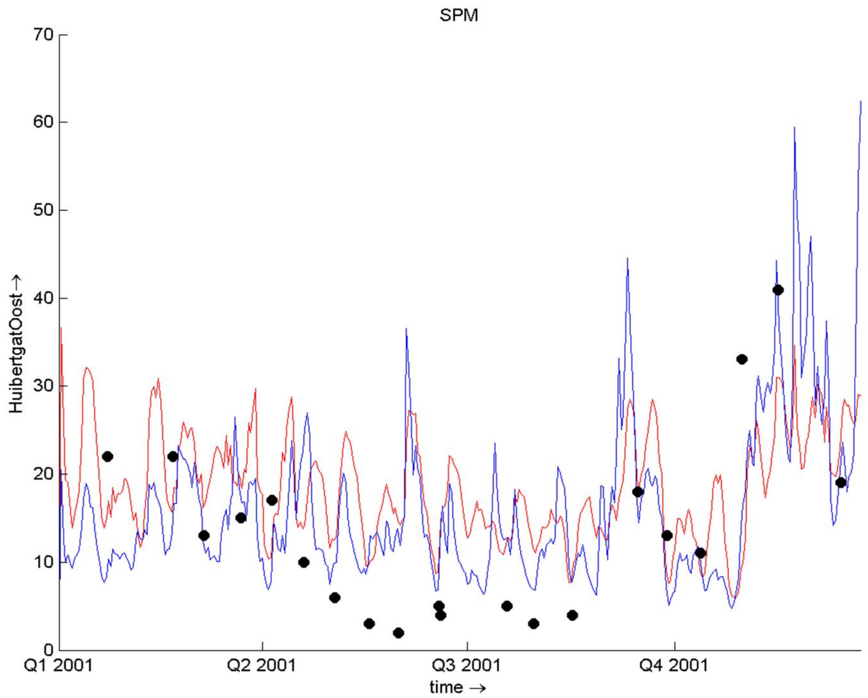


Figure 2.4 Daily-average near-surface SPM concentration (mg/L) at Huibertgat. Red line: old simulation (t16), blue line: with assimilated waves (t18).

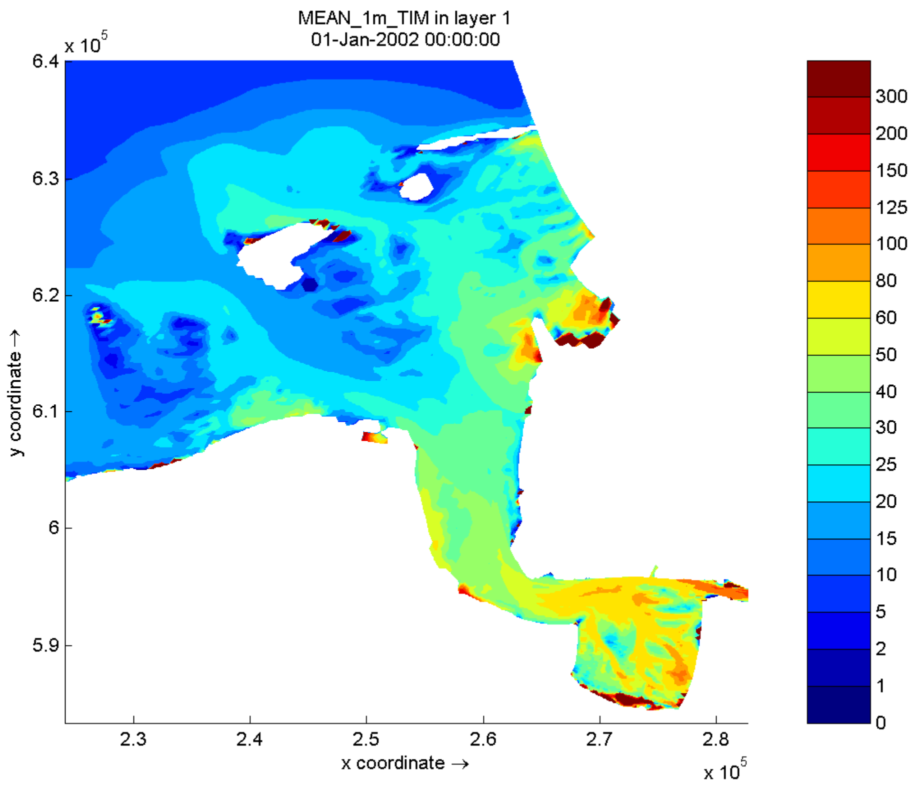


Figure 2.5 Year-average near-surface SPM concentration (mg/L), old hydrodynamics with assimilated waves (t18).

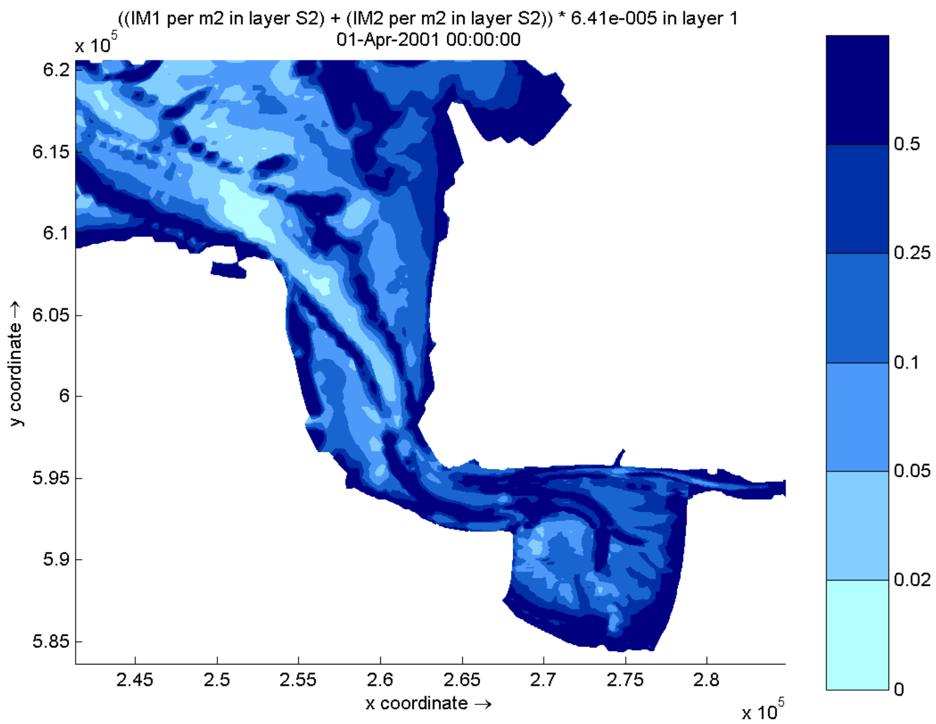


Figure 2.6 Computed mud fraction in the bed for simulation t16.

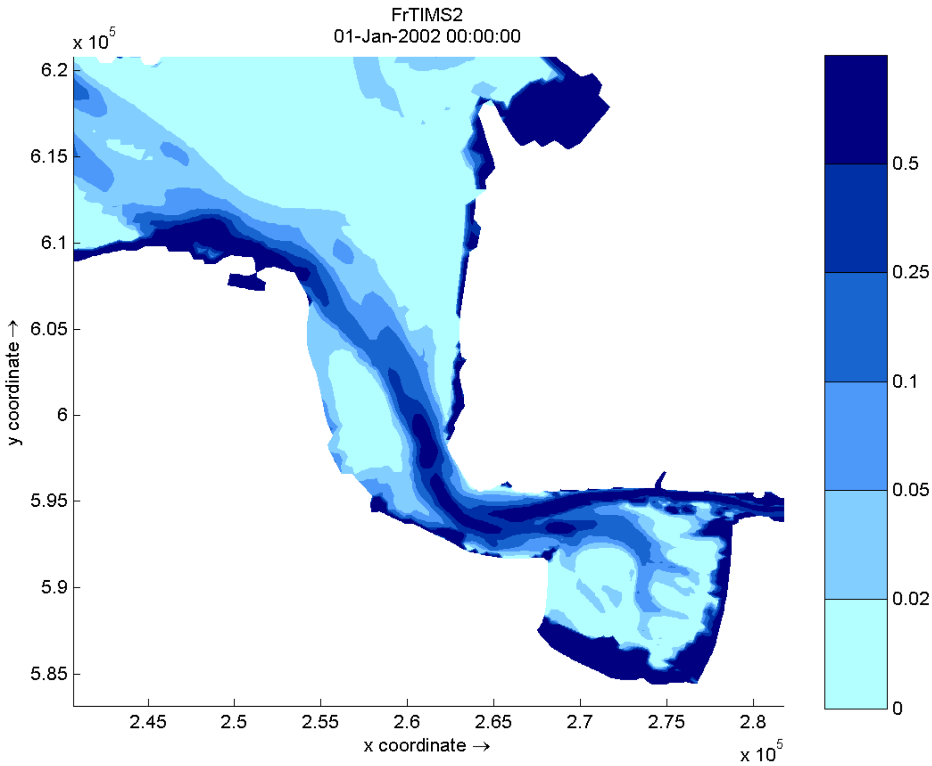


Figure 2.7 Computed mud fraction in the bed for simulation t18

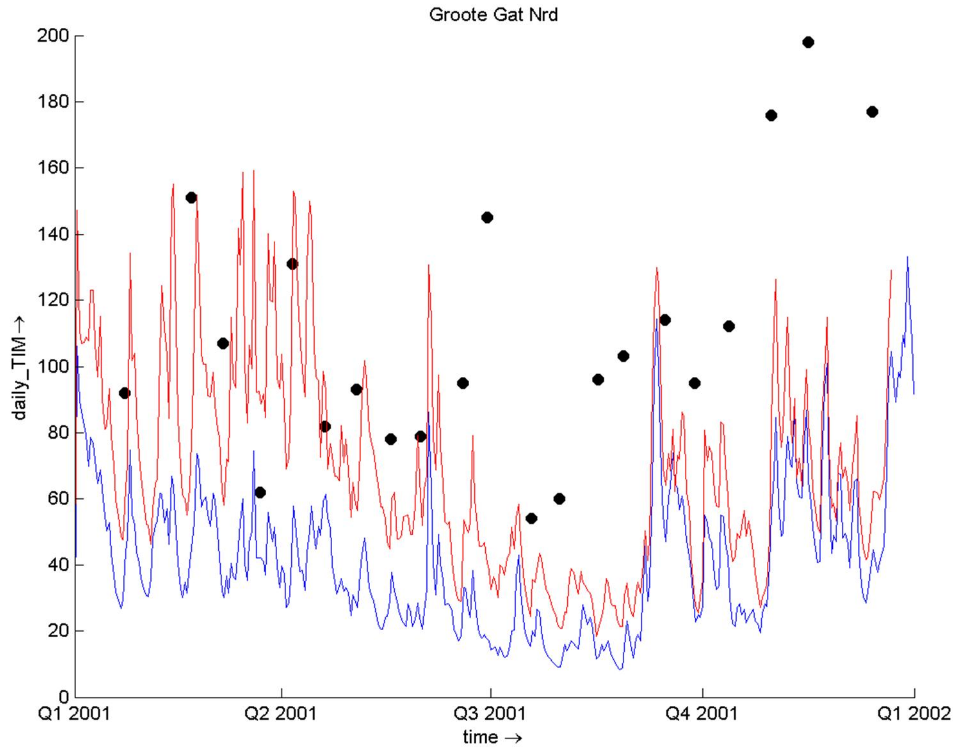


Figure 2.8 Daily-average near-surface SPM concentration (mg/L) at Groote Gat. Red line: old simulation (t16), blue line: new hydrodynamics (t19).

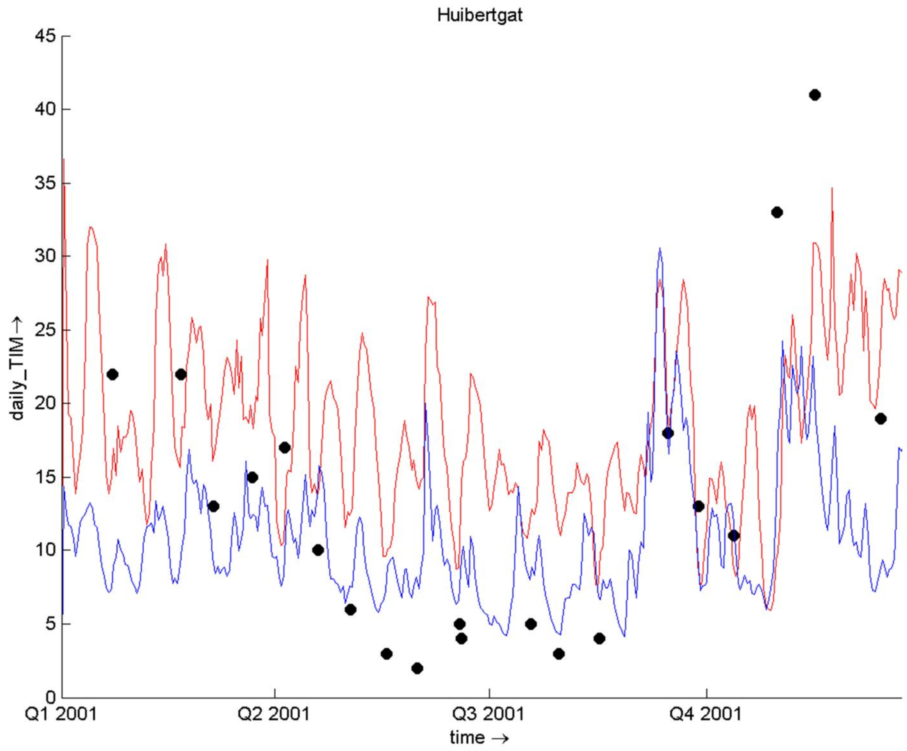


Figure 2.9 Daily-average near-surface SPM concentration (mg/L) at Huibertgat. Red line: old simulation (t16), blue line: new hydrodynamics (t19).

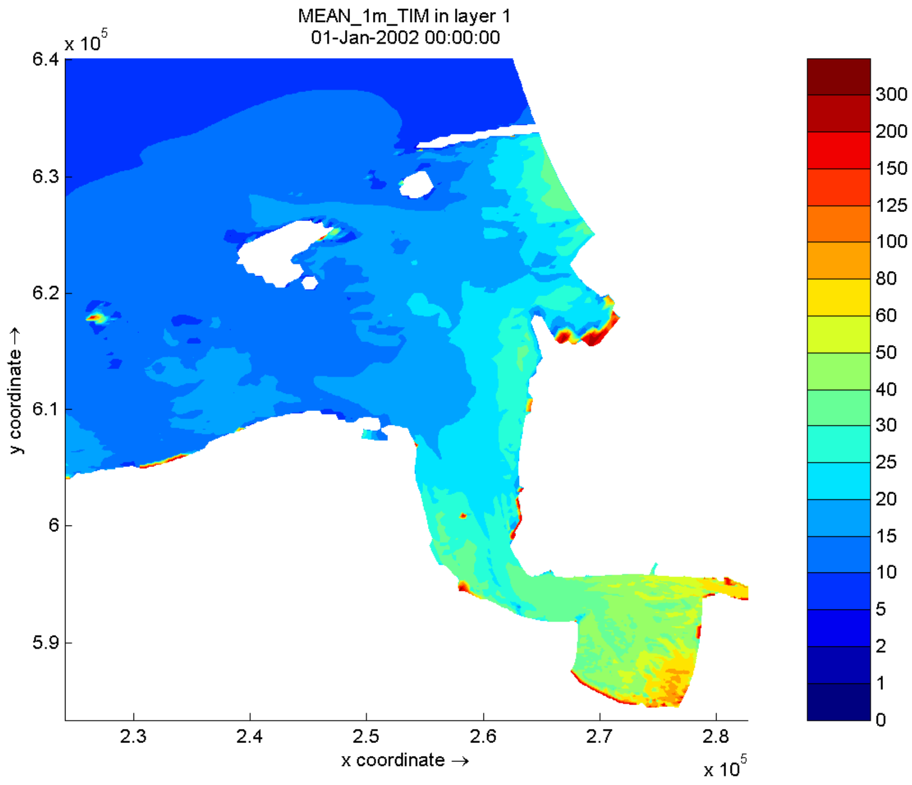


Figure 2.10 Year-average near-surface SPM concentration (mg/L), new hydrodynamics (t19).

3 Water quality and primary production

3.1 Introduction

The water quality and primary production model provides a building block in the “effect chain” modeling approach, linking the hydrodynamic (Delft3D-FLOW) and sediment transport (Delft3D-SED) to the ecological assessment in HABITAT. The primary objective of the water quality/primary production model is to get a better understanding of those factors that determine the biogeochemical and ecological dynamics in the Ems-Dollard estuary. As already described in Dijkstra *et al.*, 2012, the model will be used to provide insight in the factors controlling primary production, namely to determine whether primary production is nutrient-limited or light-limited. Considering the high nutrient concentrations and the reduced light climate due to suspended sediments in an estuary, primary production is expected to be predominantly controlled by light availability. The availability of light to primary producers is determined by: (a) the amount of incident light, (b) the bathymetry, (c) vertical mixing, (d) the presence of inorganic and organic particles and (e) the concentration of pelagic algae themselves. Consequently, the water quality/primary production model should accurately describe these factors.

The second major issue concerns dissolved oxygen levels and the occurrence of oxygen-depleted zones. Oxygen can be considered as a key variable, meaning that its concentration is determined by the complex interaction of physical (advection and diffusion, governed by hydrodynamic circulation), chemical (re-oxidation of reduced species) and biological (primary production vs organic matter degradation) processes. The interaction of these processes forms the basis of the water quality/primary production model described in the following sections. Oxygen concentration is also an important output variable to be used in the habitat suitability study for a number of benthic species.

In this report chapter, details on setting-up, development and the improvements made to the water quality/primary production model until 2012 are presented. Starting from the model set-up available at the end of 2011 (see Dijkstra *et al.*, 2012) and the issues that required further development and/or improvement identified therein, the following objectives had been formulated for 2012:

1. Improve the performance of the model in order to be able to calculate scenarios in a reasonable amount of time (less than 1 day calculation time for 1 year simulation).
2. Include new suspended sediment forcing from sediment model
3. Test the sensitivity of the model to variation in suspended matter concentrations
4. Improve description of phytoplankton biomass by adapting growth/mortality descriptions

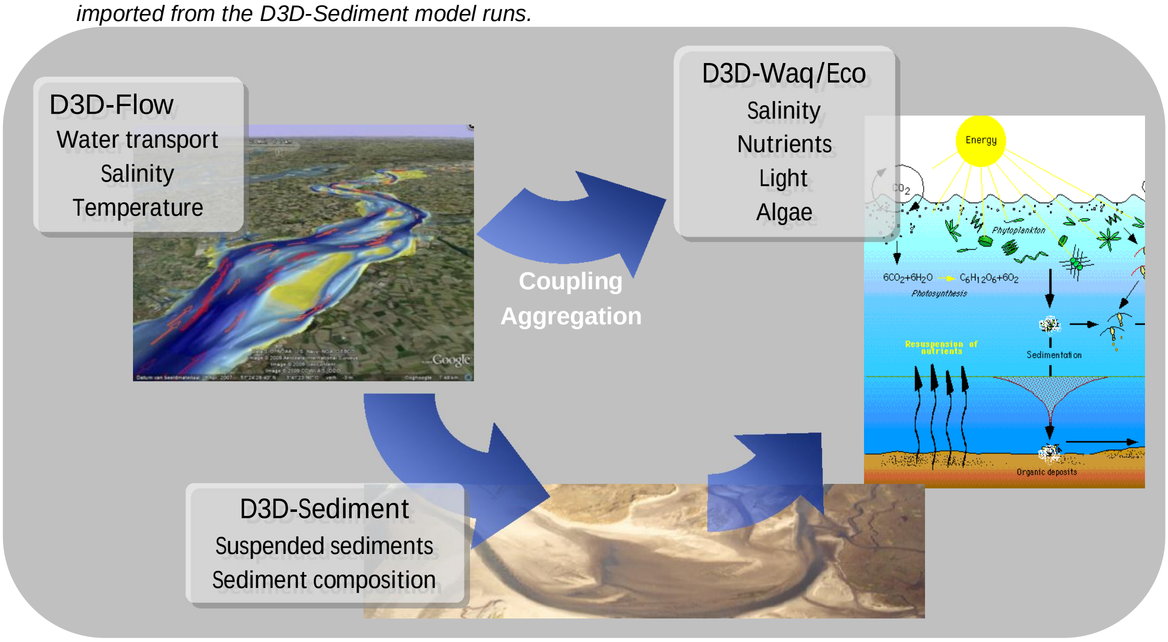
Two more goals that were formulated for this year, to include phosphate return flux from the sediment and to include grazers into the model were not carried out due to time limitations, and can be dealt with later if necessary, in specific studies addressing these issues.

3.2 Model Setup

3.2.1 Introduction

This part of the report deals with the simulation of water quality with regards to nutrients, dissolved oxygen, primary production by phytoplankton and –benthos, and biomass concentrations of phytoplankton and – benthos in the estuary. The model chain set up for this purpose consists of a water quality module (Delft3D-Waq/Eco), where input of the hydrodynamic model (Delft3D-Flow) is used to calculate transport and dispersion of substances, and a sediment model (Delft3D-Sediment) is used to prepare and deliver necessary concentrations of suspended sediment, which in turn is used to calculate light availability for primary production.

Figure 3.1 Schematic visualization of the necessary input into the D3D-Waq/Eco model. Hydrodynamic flow fields, temperature and salinity are imported from the D3D-Flow model, suspended sediment concentrations are imported from the D3D-Sediment model runs.



3.2.2 Model schematization

A relatively fine mazed curvilinear grid was applied to model water quality. The grid is based on the grid used for D3D-Flow calculations. Until 2012, a 2x2 horizontal aggregation was used (Dijkstra et al. 2011). During 2012, a further aggregation was applied, to reduce calculation time of the mode. The grid was altered in two ways compared to the previous years.

1. A horizontal aggregation was applied, so that for shallow areas, 2x2 aggregated grid compared with the hydrodynamic grid was applied. This way, a considerable gain in calculation time was established, while much of the detailed spatial patterns in areas with mud flats was retained. In deeper areas, a 4x4 aggregation was applied.
2. In one version of the model, the hydrodynamic model (8 layers) was vertically aggregated to one layer, so that it in reality becomes a 2D model. The greatest advantage of this aggregation is faster calculation (approximately 10-fold), which makes it possible to run larger numbers of scenarios within the same time. The disadvantage of

vertical integration is that no vertical resolution in variables is retained. However, the high tidal current velocities in the estuary ensure vertical mixing to such an extent that vertical gradients are most of the time not present. The 3-D version of the model with 8 sigma layers was maintained at low additional effort, which means that when needed, 3D calculations can also be done. For this report, results from both version of the water quality results are presented.

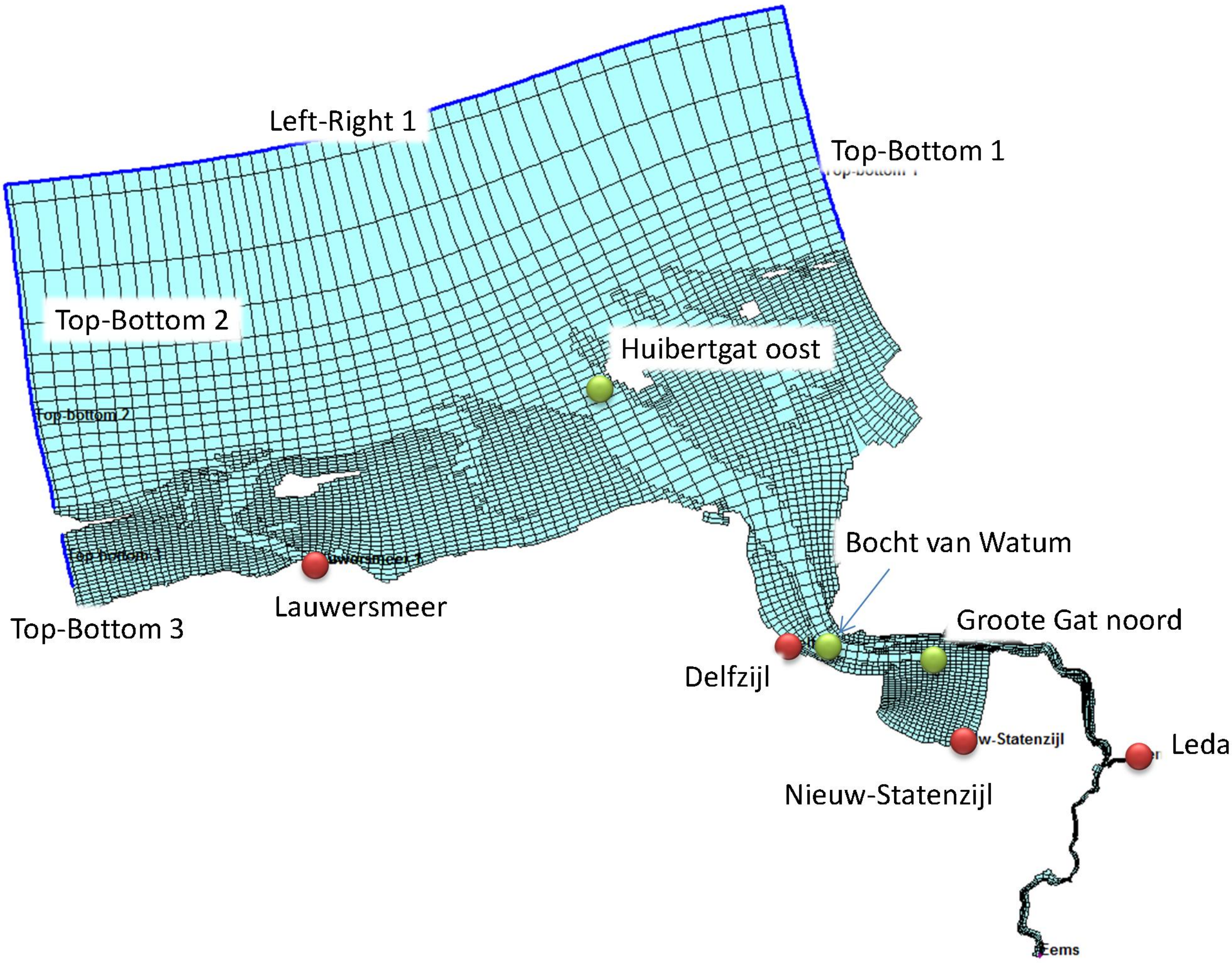


Figure 3.2 Model schematization for Eems-Dollard water quality model (D3D-WAQ). The grid is based on the hydrodynamic grid. For this purpose, the water quality grid is 2x2 cells aggregated for parts shallower than 5 meters, and 4x4 in deeper (>5 meters) parts of the estuary. Open boundaries (blue lines) and discharge points (red dots) are indicated.Green dots represent MWTL monitoring stations, where validation of water quality parameters was performed.

3.2.3 Coupling with Hydrodynamic model

The water quality modelset up in Delft3D-WAQ is coupled to the results of a hydrodynamic model (implemented in Delft3D-FLOW) that simulates hydrodynamic circulation, water velocities, temperature and salinity. The results of the hydrodynamic model are the basis for all further modeling activities related to suspended matter, nutrients and phytoplankton productivity and hence also to the Habitat model. The water quality model is coupled to a full-year hydrodynamic flow-field, simulating the conditions for the year 2001. The full-year hydrodynamic simulation accounts for daily discharge measurements of the Ems river (Figure

3.3). This is the basis of an accurate description of salinity in the estuary. At the end of 2012, a new hydrodynamic run has become available (see Chapter 2.1). The new hydrodynamic model has open boundaries in the Wadden Sea area, which is a conceptual improvement. For following projects using the Ems-Dollard WQ model, this newer hydrodynamic model may be used as a base. For the scope of the current research project, no water quality simulations have been done using this newer hydrodynamic model.

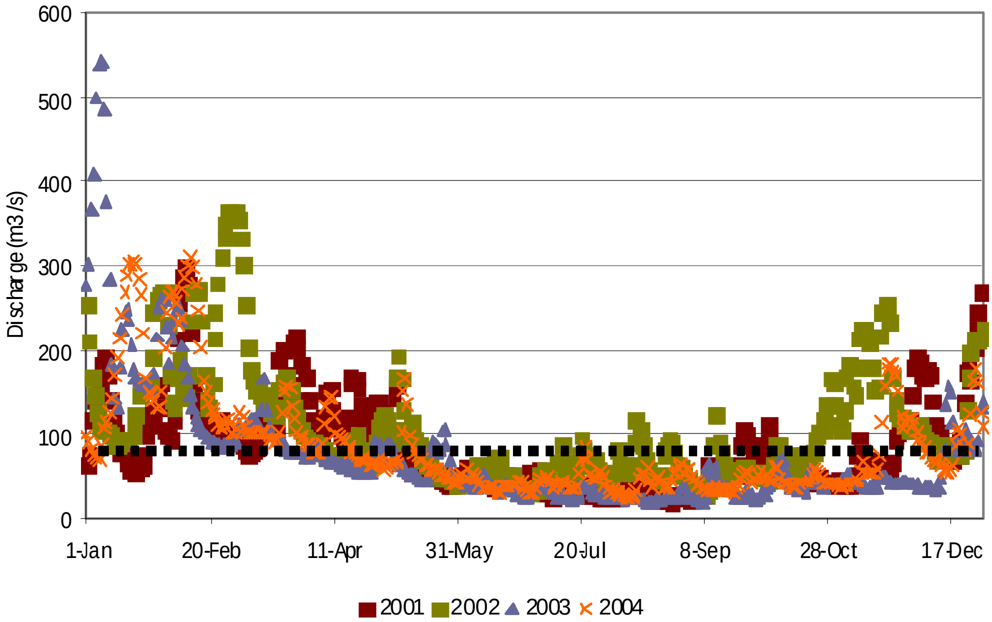


Figure 3.3 Time series for discharge (m^3/s) in the Ems River for different years. The 2001 daily discharge values are used in the hydrodynamic model that was coupled to the water quality model.

3.2.4 Coupling to sediment transport model

The inorganic suspended matter concentration, and hence the underwater light regime, varies considerably both in time and space. For this reason, significant effort has been put in obtaining a realistic, but modeled, spatial and temporal suspended sediment field and described in Chapter 2.2. The suspended sediment needs to be realistic for a satisfactory description of the light climate, and hence a sufficiently good description of primary production. The suspended sediment field needs to be modeled, in order to be able to simulate and review the effects of changes for example morphological changes on the water quality and primary production in the estuary.

A full-year model output of the sediment transport model was used as input to the water quality model in the form of a segment function, i.e. spatially and temporally-variable (Figure 3.4). In this case, the light regime in the water quality model is subject to the daily local variations in two sediment fractions IM1 (settling velocity 1 mm/s) and IM2 (settling velocity 0.125 mm/s). However, the temporal and spatial distribution of suspended sediment is notoriously difficult to model. Especially at the seaward station Huibertgat Oost, suspended sediment concentrations are overestimated during summer. For details, see Chapter 2.2. Despite the shortcomings with respect to a correct description of seasonal variation, this method is the preferred option above for example using measured values, because variations due to tide and wind are captured more realistically, and scenarios with respect to changing sediment dynamics and dredging/dumping strategies could be directly compared to the reference simulation.

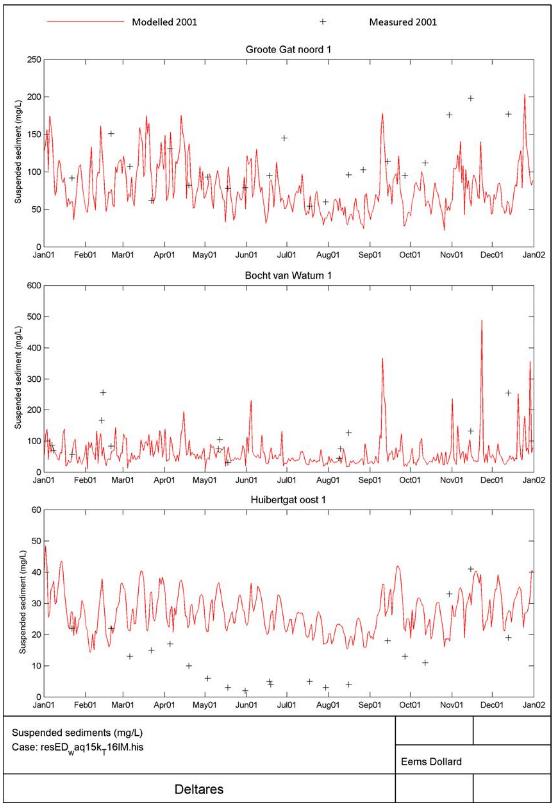
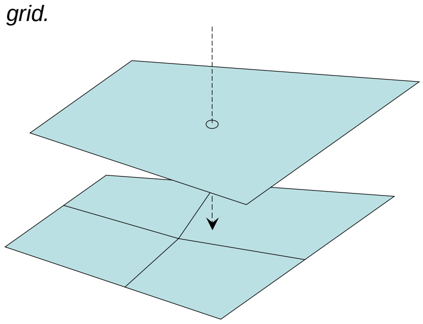


Figure 3.4 Measured suspended sediment concentration and simulated sediment concentrations (sum of IM1 & IM2 fractions) from the sediment model output used as segment function in the water quality model.

Figure 3.5 Method of resampling to calculate suspended sediment concentrations in aggregated grid cells.The centre of aggregated cells is used for resampling the sediment model grid.



The direct coupling to the sediment model output requires modification of the sediment output files in a way that they can be imported as a so called segment function (containing concentrations for all time steps and all model segments) in the water quality model. This includes, for example, spatial aggregation to match the aggregated model domain used in the water quality simulations. Aggregation in the horizontal direction was done by resampling the daily averaged original concentrations of IM1 and IM2 at the centre of the aggregated grid cells.

3.2.5 Boundary conditions

There are 4 open boundaries in the model, all located in the Wadden Sea and North Sea (Figure 3.2). This means that water and substances may travel over these boundaries so that concentrations of substances at the boundary matches the observed conditions as closely as possible. Two alternatives wereconsidered for setting up the downstream boundary conditions: a) including substances loads via nesting from the North Sea model (ZUNO model, Los et al., 2008) using Delft3D-NESTWQ and b) assignment of boundary concentrations based on field concentrations. After several test runs, it was decided to use field measurements (Table 3.1) since the performance of the ZUNO model in the eastern Wadden Seais still questionable, implying that the nested values do not corrrespond to the conditions in the field. The location of the monitoring locations considered for the different boundaries is shown in Figure 3.6. The field measurements from these stations include salinity, dissolved oxygen, TotN, NO3, NH4, TotP and PO4 concentration. Figure 3.7shows the 2001 time series for inorganic nutrients at the three monitoring locations used to contrain the downstream boundaries.As a first approximation, DetN is assumed to be the difference between TotN (not shown) and DIN, whereas DetP is approximated as the difference between TotP (not shown) and PO4.

Table 3.1 Monitoring locations used for defining nutrient concentrations in boundary conditions

Boundary name	Monitoring station used as boundary conditions
Left-right 1	NZR9TS010 (Terschelling 10)
Top-bottom 1	Rottum3
Top-bottom 2	NZR9TS010 (Terschelling 10)
Top-bottom 3	WZ480 (Zoutkamperlaag)

For the model to be able to calculate the transport of substances over these boundaries, a concentration needs to be specified at each open boundary for the whole period of the model run for each *active* substance, that are all substances that are transported by the currents (this excludes sediment substances). Observations from nearest possible monitoring stations were used for this purpose. Actual used nutrient concentrations are presented in Figure 3.7. For Terschelling10 and Zoutkamperlaag, monthly values were available. For Rottum3, only summer values were available.

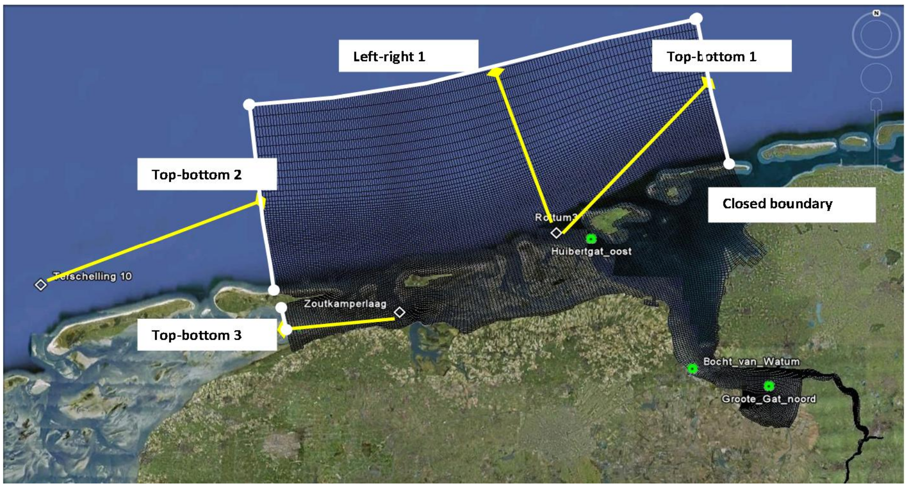


Figure 3.6 Location of boundaries, measurement stations for boundary conditions (squares) and monitoring stations for validation (green circles)

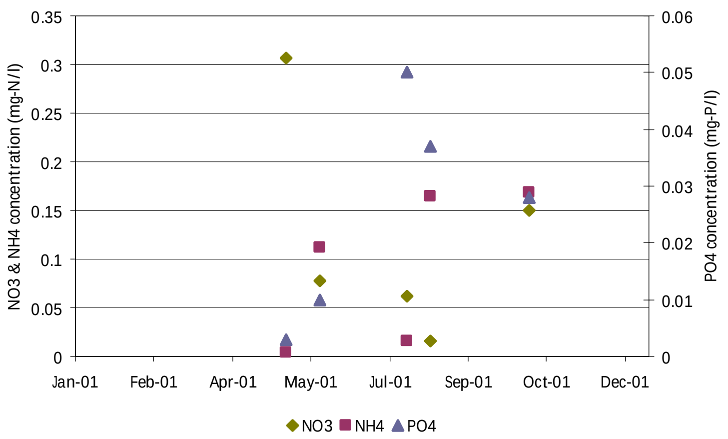
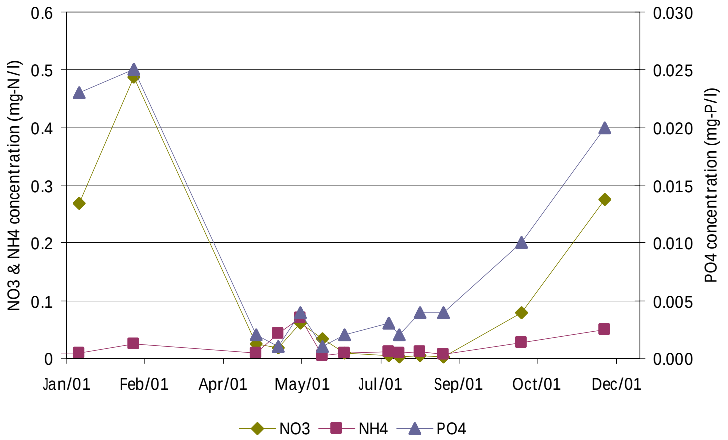
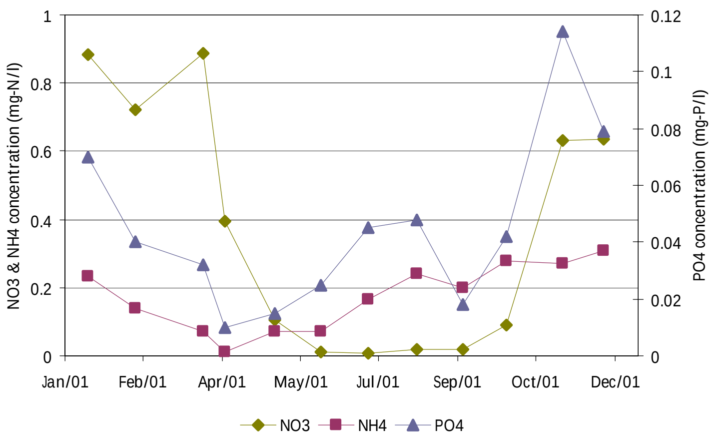
	Rottum3
	NZR9TS010 Terschelling10
	WZ480 Zoutkamperlaag

Figure 3.7 Measured inorganic nutrient concentrations at the stations used for boundary conditions

3.2.6 Water quality/ecological processes

The water quality model includes the main processes of the Generic Ecological Model (GEM) in combination with the phytoplankton module BLOOM, whichmodels the competition between phytoplankton species and the adaptation of species to limiting nutrients or light. (Los et al. 2008, Blauw et al. 2008). The model set-up described in Deltares (2010) contained the main reactions that determine nutrient dynamics, including the effect of light availability and primary production processes (Figure 3.8 and Table 3.2). A list of all state variables is given in Table 3.3.

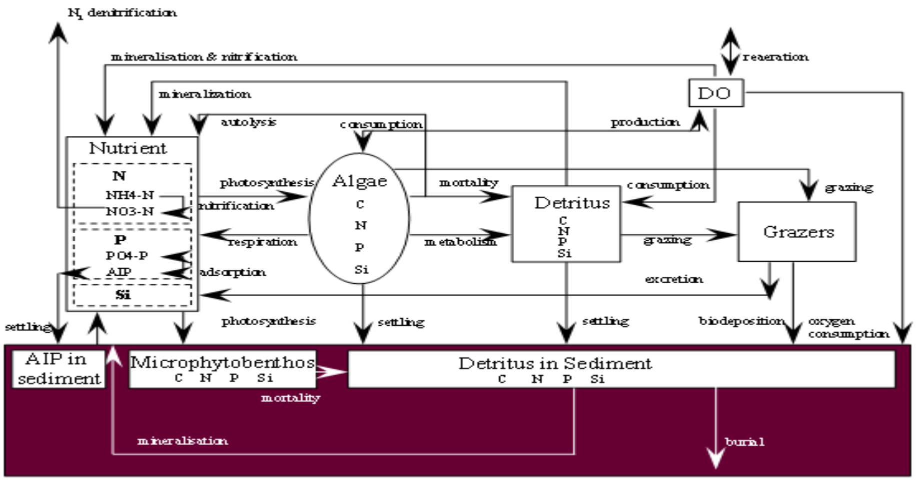


Figure 3.8 Schematic overview of the main DELWAQ processes and variables

Table 3.2 Overview of the main processes included in the water quality model. Note that an additional process, benthic production by microphytobenthos was implemented in 2011.Grey processes are not yet active in the current set-up.

Process	**X on** X **off**
sedimentation and resuspension	X
Re-aeration of oxygen	X
aerobic decomposition of organic substances	X
Denitrification	X
Nitrification	X
phosphorus sorption/desorption	X
light extinction	X
Phytoplanktongrowth/respiration/mortality	X
atmospheric deposition	X
microphytobenthos	X
Grazing	X
sediment diagenesis	X

Benthic primary production - The inclusion of benthic diatoms/microphytobenthos is formulated as a competing algal species and is modelled within the BLOOM module. The microphytobenthos type of algae can occur in two forms, fixed (attached to the sediment and therefore not transported over the model grid) or suspended (in the water column, following mixing and transport of the hydrodynamic model). The fixed algae received light from the deepest layer when using the 3D model, and from the deepest 10 % of the total depth when using the 2D model. Under certain conditions determined primarily by the computed critical shear stress, the sediment is eroded and the microphytobenthos are released into the water column. In the water column, these microphytobenthos are regarded as “regular” phytoplankton types and therefore integrated in the BLOOM computation. The settling of the suspended microphytobenthos into deep areas will lead to their death. However, if they settle in shallow areas with sufficient light, they are converted back to the fixed microphytobenthos type. For the current model set-up, the threshold shear stress for the erosion of microphytobenthos was set to a very high level, effectively blocking the process of resuspension. Therefore, no pelagic microphytobenthos have been tested in the model. In a later step, when also validation data come available, threshold values for shear stress could be set to a more realistic level, which will allow microphytobenthos to spend part of their time in the water column.

It was assumed that microphytobenthos in the Ems/Dollard consist of diatoms. Therefore, growth and mortality parameters that are used for marine diatoms in the current model were also implemented for the microphytobenthos type of algae. However, since microphytobenthos is fixed to the sediment and is normally not transported in the surrounding water, it is usually facing larger variations in salinity than pelagic algae and has adapted a higher tolerance towards this factor. Therefore, the tolerance to low salinity water of the microphytobenthos type in the model was set higher as compared to the Marine Diatoms type.

Table 3.3 An overview of the state variables in the model.

State variable	**Delwaq name**
Water column	
Salinity	Salinity
Pelagic phytoplankton community	4 species/groups,12 types [1]:
Dinoflagellates	(DIN_E,N,P)
Marine Diatoms	(MDI_E,N,P)
Marine flagellates	(MFL_E,N,P)
Phaeocystis sp.	(PHA_E,N,P)
detritus fraction of organic carbon, nitrogen, silica and phosphorus	DetC, DetN, DetSi, DetP
inorganic nitrogen (ammonia, nitrate) inorganic dissolved silica inorganic phosphorus (dissolved ortho-phosphate)	NH4, NO3 Si PO4
dissolved oxygen	OXY
Sediment	
organic fraction of organic carbon, nitrogen, silica and phosphorus	DetCS1, DetNS1, DetSiS1, DetPS1
Phytobenthos (3 types)	Benthic diatoms (ULF_E,N,P)

[1] Each phytoplankton species is composed of 3 types: N-type representing the ecophysiological condition of a species under nitrogen limitation, a P-type for phosphorus limitation and an E-type representing the state of a species under light limitation

Algal mortality is caused by temperature dependent natural mortality, grazing by consumers, as well as salinity stress mortality. Salinity driven mortality is described with a sigmoidal function of chlorinity, governed by two parameters. For marine algae, mortality rate increases with decreasing chloride concentration or salinity. The fresh water algae mortality rate increases with increasing chloride concentration. At this moment, only marine algae are included since only a very small part of the estuary consists of fresh water. Also, the fresh water part of the model domain contains extremely high concentrations of suspended sediment, and therefore low light availability. Hence, no significant primary production can be expected from freshwater species in this area. The list of algae may be reconsidered and extended with freshwater species when it is clear that freshwater primary production is underestimated.

3.2.7 Water-borne nutrient loads to the estuary

The Ems River is considered as a point source for nutrients. The time series concentration for NO3, NH4 and PO4 and other dissolved substances was interpolated from the OSPAR compilation of field measurements and resulted in the seasonal variation as displayed in Figure 3.9. These concentrations, combined with the daily Ems River discharge rates (same as used in the D3D-Flow model) resulted in the nutrient loading shown inFigure 3.10. Detritus Nitrogen and Detritus phosphorus was caculated as the difference between total nitrogen/phosphorus and DIN/DIP. The discharge rates and nutrient concentrations of the other four point sources, Eems/Leda, NW-Stratenzijl, Delfzijl and Lauwersmeer are assumed to be the same as those reported in Deltares (2010).

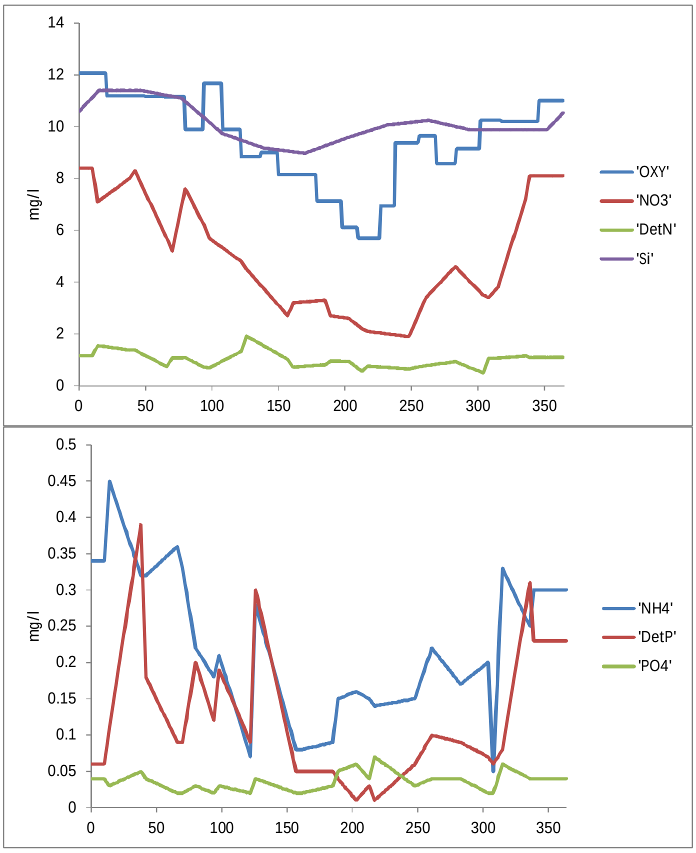


Figure 3.9 Concentrations of dissolved substances applied to calculate the Ems load.

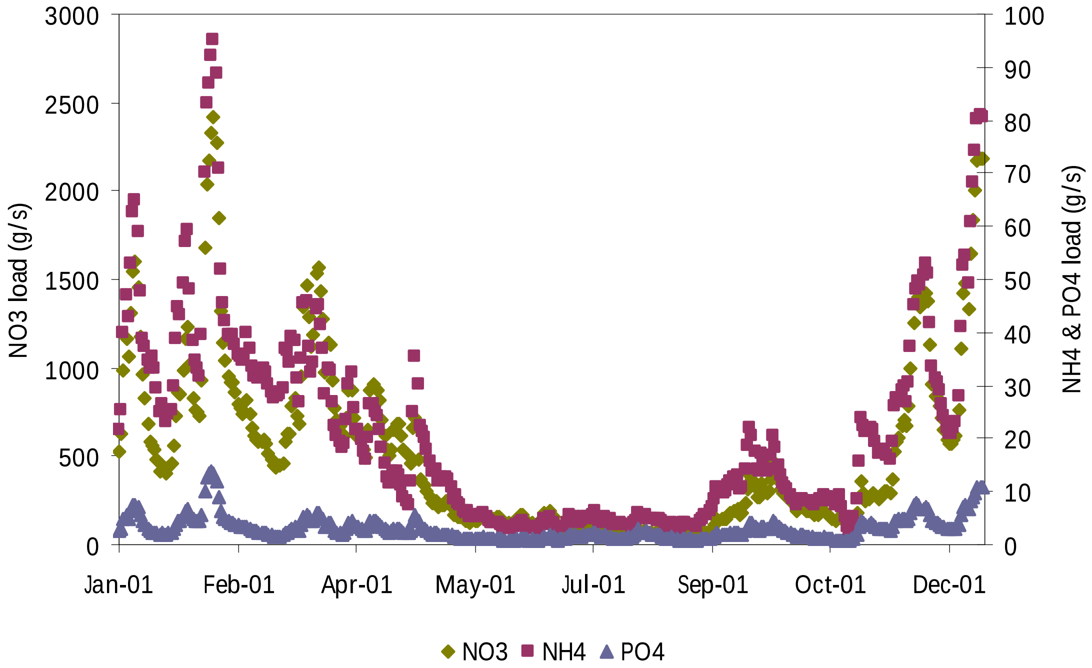


Figure 3.10 Time series of nutrient loads from the Ems river

3.2.8 Atmospheric deposition of nitrogen

A constant deposition of oxidized and reduced nitrogen in time and space was assumed. Modelled data for atmospheric deposition in the Eems-Dollard area were obtained from EMEP through the Baltic Nest interface (nest.su.se). Spatial variation in the area is visualized by interpolated EMEP modelling results (Figure 3.11, origin: Baltic Nest). There is a gradient where lowest deposition is at the North Sea side of the estuary, and higher values towards the Ems outlet. As an approximation, values were chosen so that they correspond with the values close to the North Sea boundary (light blue part of the figures), which were 600 mg/m^2/yr and 900 mg/m^2/yr for oxidized and reduced nitrogen respectively.

The monthly variation within the year 2001 was not available from this source, but can be extracted from measured data of wet deposition at a monitoring station, Kollumerwaard, the closest station compared to the area of interest (Figure 3.12). As expected, wet nitrogen deposition varied with precipitation. Variation of wet nitrogen deposition within a year was mostly noticed in the first two months, where values were lower than in the rest of the year. Therefore, the choice for a constant atmospheric deposition might give an underestimation of nitrogen load only in January and February. This underestimation is clearer in 2001 as compared to an average year (Figure 3.13). Note that the variation in dry deposition is not included in this analysis. The values for average deposition that were finally used in the model set-up are shown in Table 3.4.

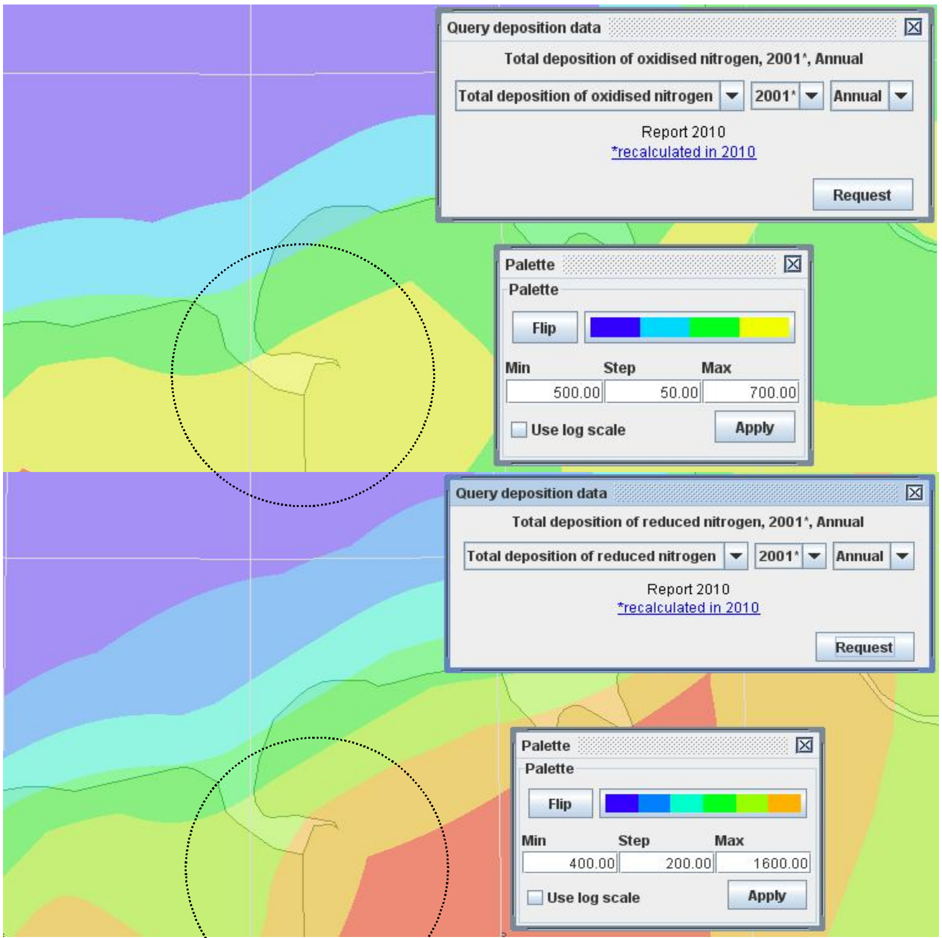


Figure 3.11 Maps of the area around the estuary showing the spatial distribution of total deposition of oxidized (top) and reduced (bottom) nitrogen. Data shown by the screenshots are modelled values from EMEP through the Baltic Nest interface (nest.su.se)

Figure 3.12 Location of the monitoring station "NL09 Kollumerwaard" for wet deposition closest to the estuary

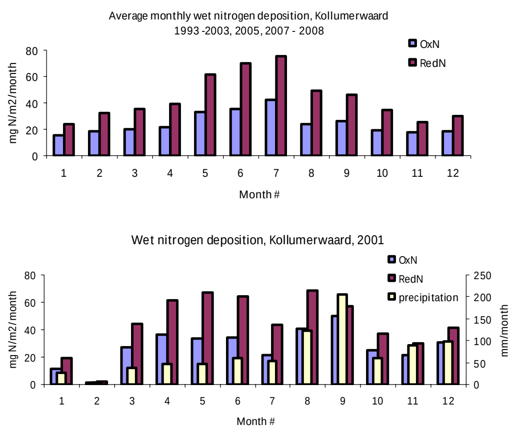


Figure 3.13 Monthly wet nitrogen deposition for 2001 (left panel) and averaged over the years 1993-2003, 2005, 2007-2008). Data from EMEP (www.emep.int).

Table 3.4 Area-specific atmospheric deposition values used for the current model set-up.

	DELWAQ variable	mg N/m^2/year	g N/m^2/day
Oxidized N	NO_3	600	1.64 . 10^{-3}
Reduced N	NH_4	900	2.46. 10^{-3}

3.2.9 External forcings

Surface irradiance - Surface radiation values (J/m^2/d) obtained from the KNMI station "Nieuw Beerta" were converted to daily averaged radiation in W/m^2 (Figure 3.14). It is assumed that the surface radiation varies in time but is spatially constant.

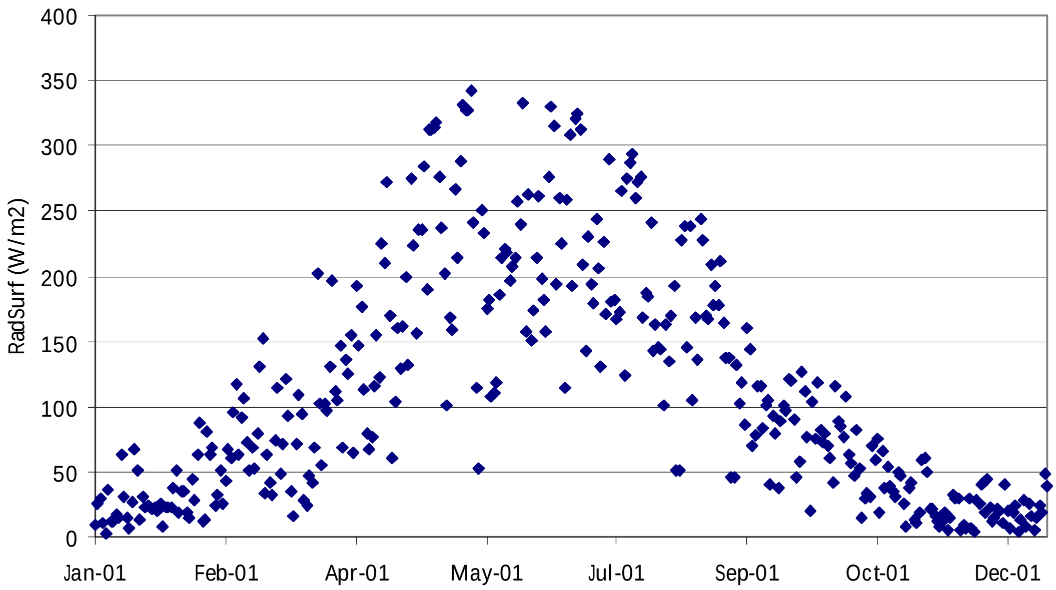


Figure 3.14 Measured daily averaged irradiance, calculated from surface irradiance at station "Nieuw Beerta".

Wind velocity - Daily measurements of wind speeds for year 2001 at "Nieuw Beerta" were obtained from the KNMI database. As for surface irradiance, wind velocity is assumed to be spatially constant but temporally variable specified as time series measurements. Wind velocity is included in the water quality model since it is used to determine oxygen re-aeration between the water surface and the overlying atmosphere and vice-versa.

Water temperature –The simulated water temperature from the hydrodynamic model is used as a forcing function in the water quality model, resulting in spatially and temporally varying temperature fields (Figure 3.15).

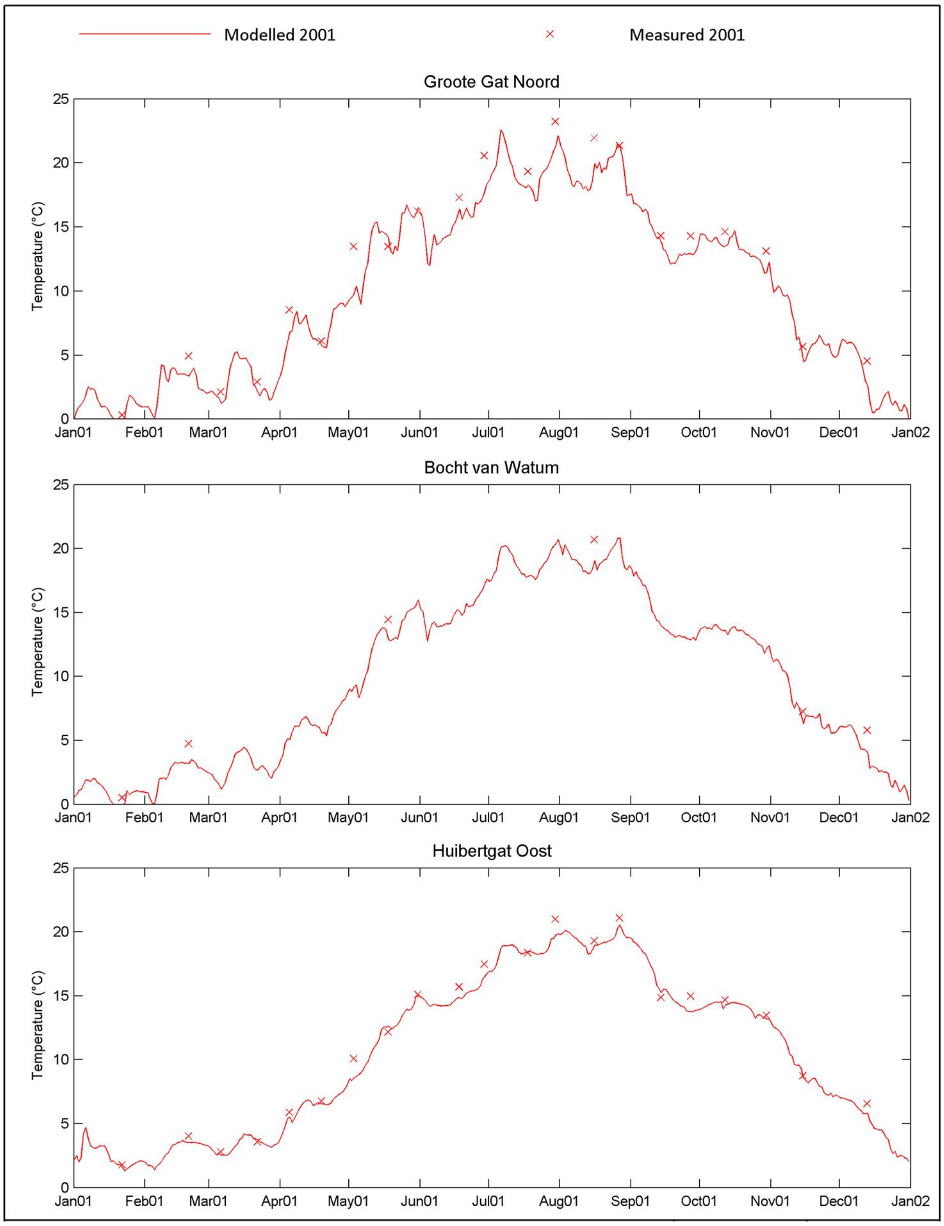


Figure 3.15 Measured water temperature and modelled results from the hydrodynamic model, used as a forcing for the water quality model at three reference monitoring stations.

3.2.10 Initialization of the model

The initialization of the model was done by running it for two full years repeating the conditions for 2001.

3.2.11 Software and numerical aspects of the model

Delwaq version 24278 was used for the runs in this report. A known incompatibility of the current model set-up with earlier versions of the software is that earlier versions can not be used when modelling benthic algae in the 2D schematization.

Based on earlier test runs (Dijkstra et al., 2011), it was decided to use numerical integration Scheme 21 and a time step of 10 minutes in the reference water quality simulations. This combination of settings presented the most reasonable trade-off between accuracy, maintainance of salinity gradients, limiting the computational time to around 3 days for one year simulation.

3.3 Scenarios

3.3.1 Historical suspended sediment concentrations

Suspended matter concentrations have increased up to about a 10-fold in the inner parts of the estuary and the Ems river since the 1950's (Figure 3.16).

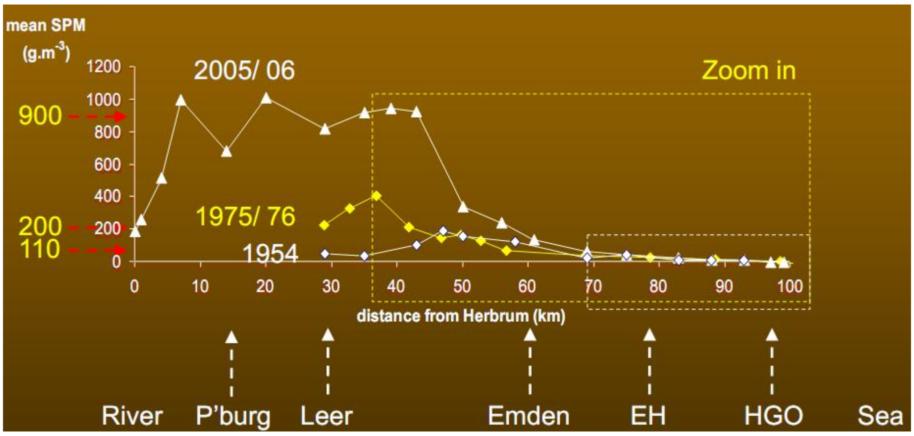


Figure 3.16 Suspended sediment concentrations in different years. The 1954 scenario was used as inspiration to investigate the effect of restored transparency on primary production (Raad voor de Wadden, 2010).

The effect of lower suspended sediment concentrations, simulating conditions close to the 1954 situation, was tested by applying a multiplication factor to the concentrations that were used in the 2001 reference run (Figure 3.4). The multiplication factor that was used in this comparison varied from 1 in the North Sea, to 0.2 in the River Ems, with a continuous gradient in between (see Figure 3.17 for the spatial variation of the reduction factor used for suspended sediment forcing).

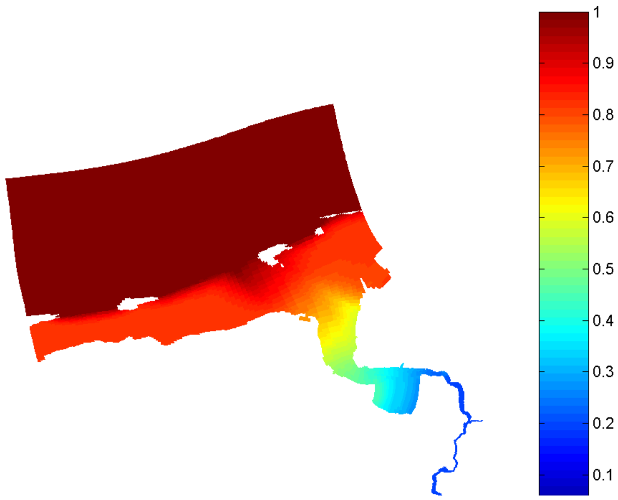


Figure 3.17 Reduction factor of forced inorganic matter concentration for the scenario with historical suspended matter concentrations. The reduction varies from 0 at sea till 10 in the Ems river.

3.4 Results

3.4.1 Consistency checks

The model has been run for the year 2001. A conservative state variable called "continuity" has been simulated. This variable is initialised at a value of 1.0 and is given boundary conditions equal to 1.0. If the hydrodynamic forcing (water volumes and water fluxes) is consistent, the resulting concentration consistently should be 1.0 in space and time. Deviations from this result could indicate errors due to differences in numerical schemes for the hydrodynamic model and the water quality model, or inconsistent boundary conditions. This demonstrates the conservation of water in the hydrodynamic forcing, which ensures conservation of mass in the water quality model. Figure 3.18 illustrates that the model is consistent in this respect.

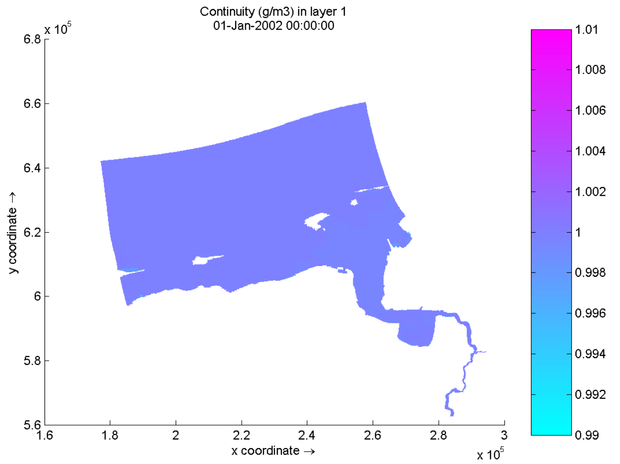


Figure 3.18 Distribution of continuity at the end of a one year simulation. Only at some intertidal areas, the value deviates from 1.

3.4.2 Model validation and comparison with historical scenario

The following main water quality parameters are included in the model validation:

1. Salinity
2. Oxygen
3. Chlorophyll-*a*
4. Nutrients:
    a. Total nitrogen, NH4, NO3
    b. Total Phosphorus, PO4
5. Light extinction
    a. – this parameter could only be compared to measurements for other years, since no extinction measurements for 2001 were available.

For each substance, time series of model results and field measurements (when available) are shown for the three reference locations, Groote Gat Noord, Bocht van Watum and Huibertgat Oost.

3.4.3 Validation of 2D and 3D model setup

Both the 2D and 3D model results are presented in this part. Chronologically, validation was first done with the faster 2-dimensional model, after which 3D calculations were performed for comparison. For further studies, both models can be used, depending on the level of detail that is required and the calculation speed restraints.

3.4.4 Salinity

WAQ salinity follows in general the salinity as it is calculated in the hydrodynamic (FLOW) model. It even performs better especially in the beginning of the year, as compared to the observations. Since the salinity gradient throught the estuary is remained, and the variation in time is similar between the WAQ and FLOW results, there does not seem to be any negative effect of the aggregation of the WAQ model as compared to the FLOW model. Also, the 2D WAQ model does not perform much different from the 3D model based on salinity.

WAQ salinity agrees in general well with observations at all three stations, but in the Dollard (Grootegat Noord) there is a measured drop in salinity that could not be reproduced by the model (Figure 3.19). Autumn 2001 is known to have been a very wet period. One possibility for this difference is that there was a significant input of rain directly on the water surface in that period, which is a process that is not modelled. Alternatively, the fresh water transport from the polder exits is underestimated in the current model grid.

At the most seaward station Huibertgat Oost, modelled salinity compares on average well to the observed values. During summer, observed values are slightly higher than the modelled ones, while towards the end of the year, observed values drop, which is not seen in the modelled salinity. Most likely, salinity at this station is very much influenced by the boundaries in the North Sea and Wadden Sea, which have been approximated in the model by monitoring stations nearby (section 3.2.5). However, it is likely that the salinity at the monitoring stations deviates from the salinity at the location of the model boundaries, which may explain the differences between modelled and observed results.

3.4.5 Extinction of light and the effect of reduced sediment concentrations.

For the year 2001, no measurements of light extinction were available from Waterbase. The total extinction of light is one of the outputs of the model, and here, we compare this extinction between the different model runs (2001 and historical) and with measurements from other years. Light extinction is highest in the upstream parts of the estuary and compares reasonably well with measurements from other years (Figure 3.21). The effect of lower suspended sediment concentrations was, due to the nature of this reduction (section) also highest in this upstream part. However, the effect on phytoplankton biomass was strongest at Bocht van Watum, which indicates the strong horizontal gradients in the estuary, and the role of horizontal advection in the distribution patterns of substances.

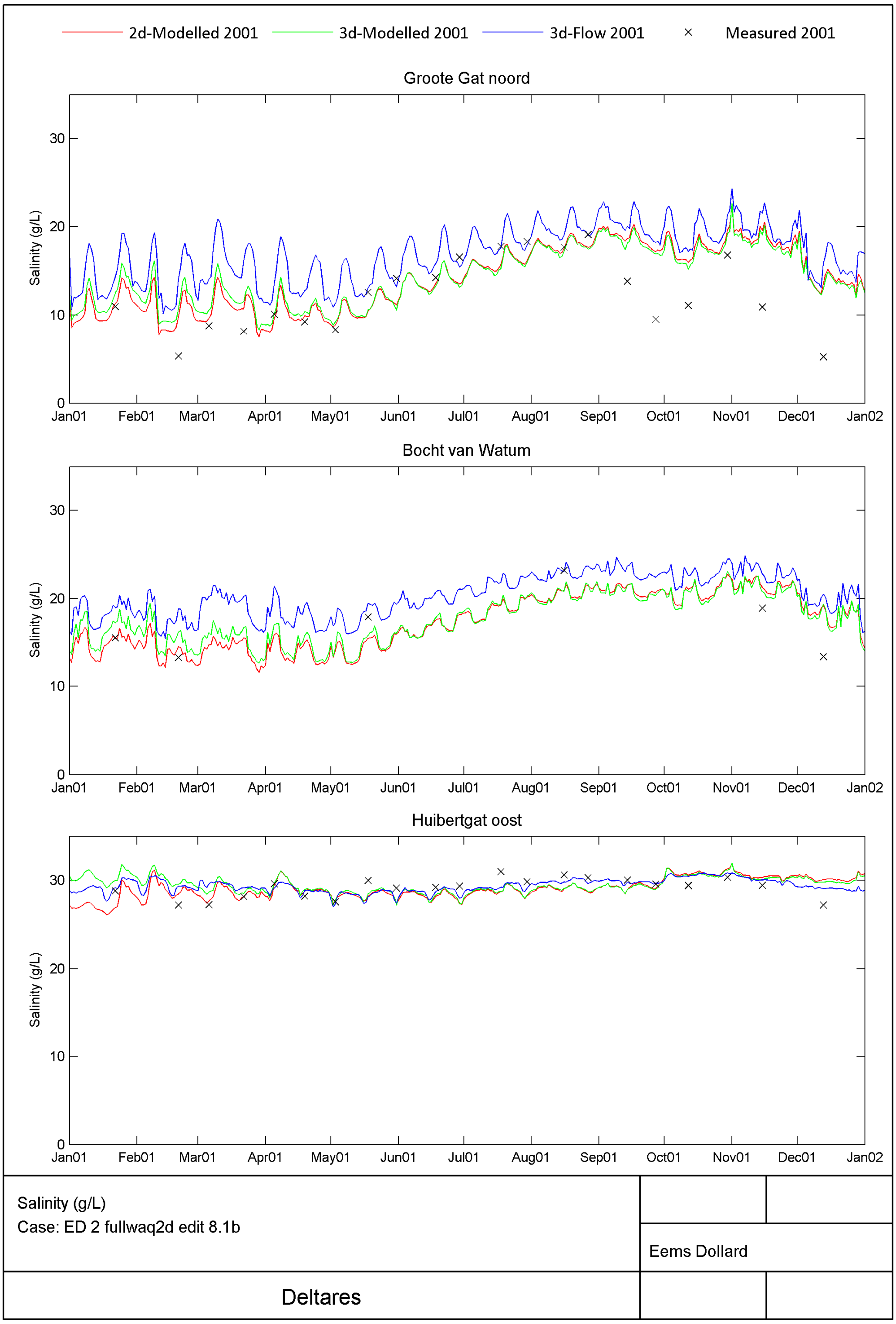
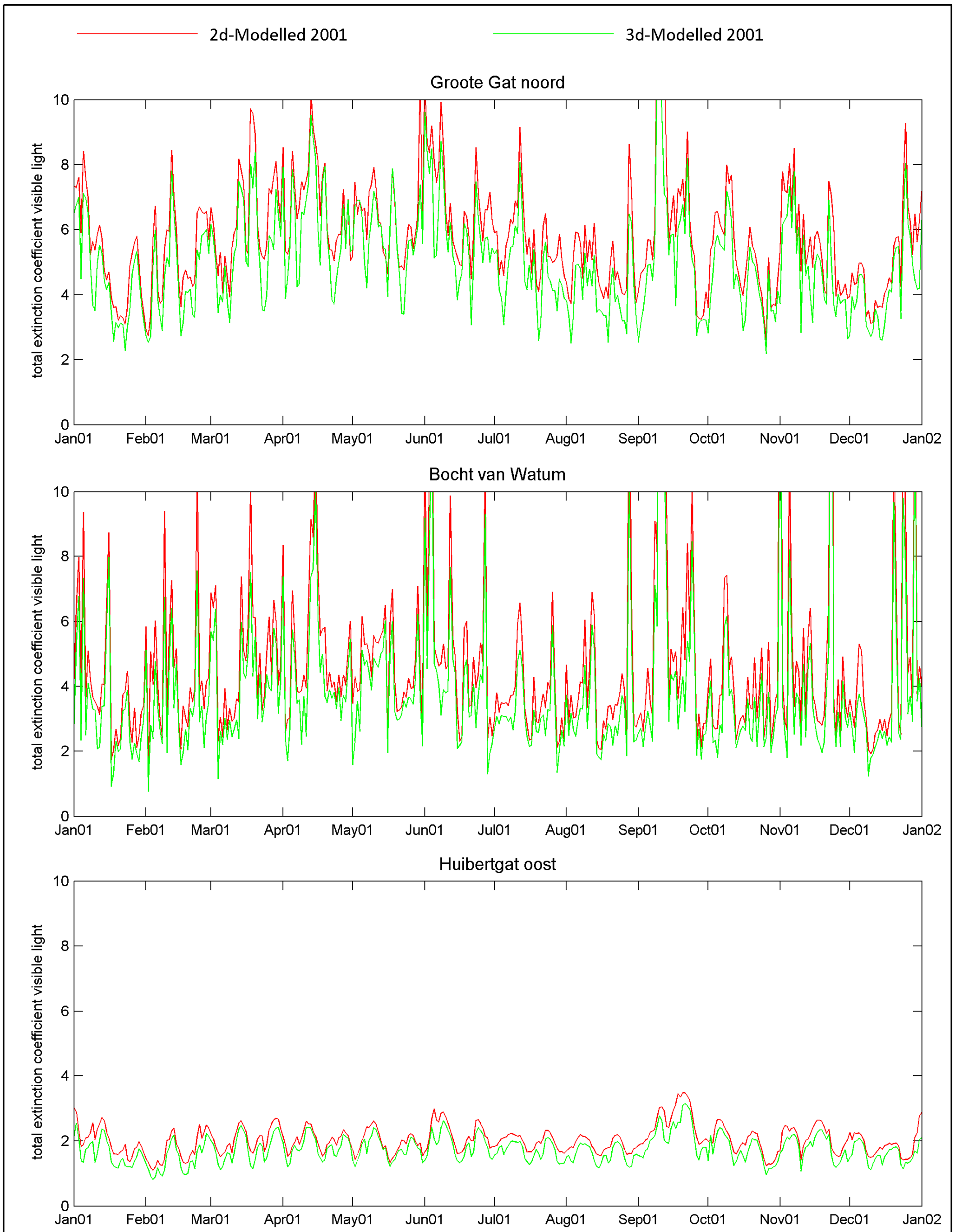


Figure 3.19 Modelled and measured salinity in the surface water at three MWTL stations



total extinction coefficient visible light
Case: ED 2 fullwaq2d edit 8.1b

Eems Dollard

Deltares

Figure 3.20 Modelled total extinction in the surface water at three MWTL stations

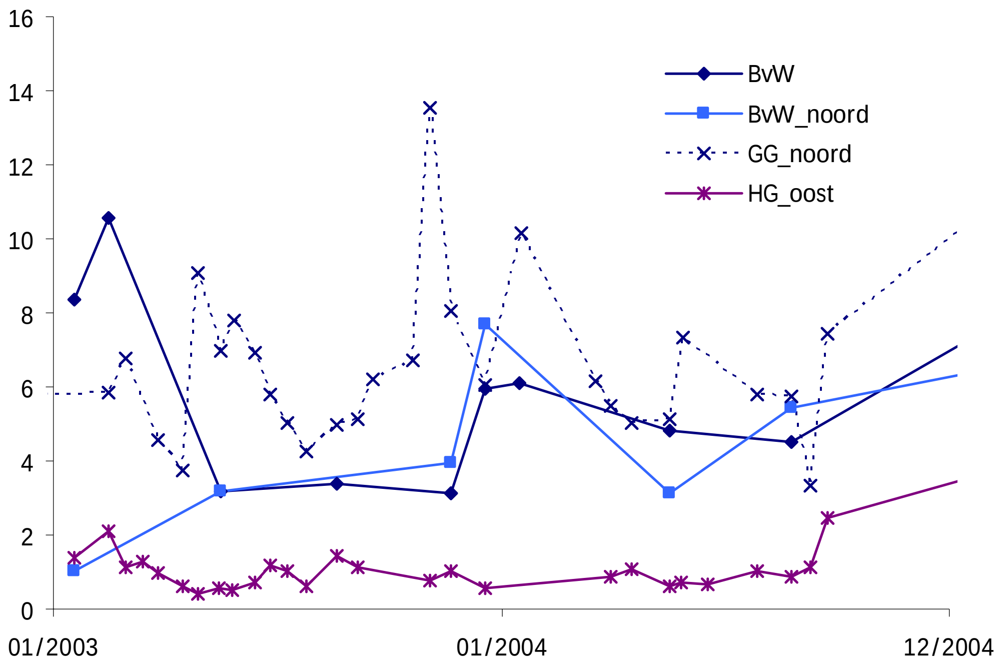
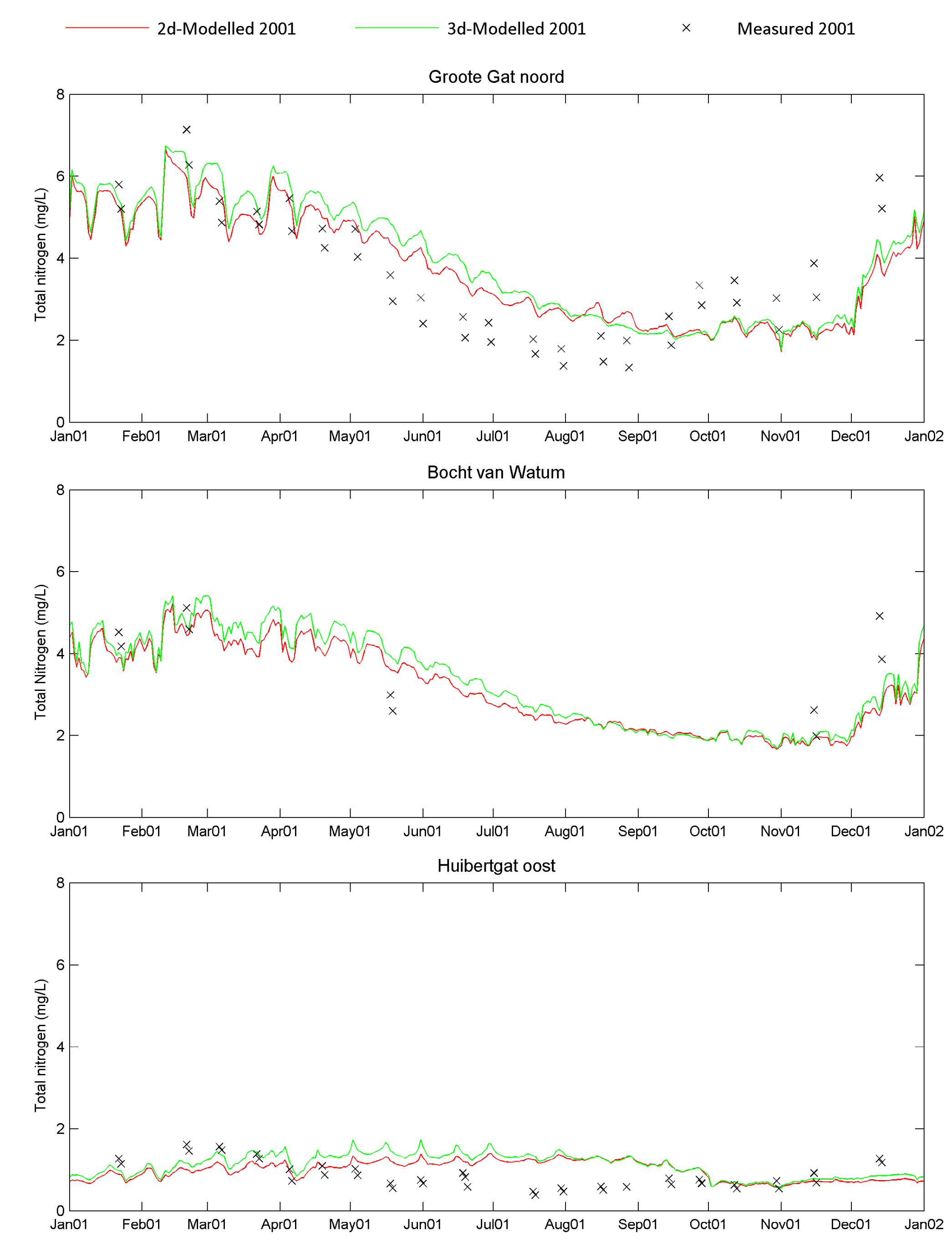


Figure 3.21 Total extinction coefficient for the reference stations for the years 2003 and 2004. Data for 2001 were not available.

3.4.6 Nutrients

The concentration of dissolved nitrogen nutrients in the Dollard are dominated by the inflow of nutrients at the Ems mouth. Concentrations levels are in general in agreement with the observed values, but the seasonal variation is not always reproduced well. For ammonium, summer concentrations are overestimated, but concentrations are in general very low. As in Grootegat Noord, ammonium concentrations at station Bocht van Watum seem to be overestimated. The overestimation is at this station during the whole year, indicating that remineralization may be slightly too high in the model.
Ammonium concentrations at the station Huibertgat Oost are slightly overestimated by the model. Nitrate concentrations match well for all stations, but the observed decrease during summer is not reproduced in the modelled values. Possibly, nitrate uptake by phytoplankton is slightly underestimated in the model.
Modelled total phosphorus and inorganic phosphate concentrations are in general in agreement with the observations. However, the observed increase in summer at Groote Gat Noord, probably due to a return flux from the sediment is not seen in the model results. Since this return flux process is not included in the model, it can not be expected that the model will reproduce this pattern.
Reactive phosphate concentrations at Huibertgat Oost compare well with the observations. Like salinity, nutrient concentrations at Huibertgat are rather influenced by the model boundary conditions. Processes in the estuary and loads from the river Ems are likely to have least effect at this station as compared to the other monitoring stations.
Silicate concentrations are not well reproduced at the beginning of the year, and also the observed low concentrations from approximately May are not seen in the model results. Towards the end of the year, modelled concentrations agree again with the obervations. Apparently, silicate loads from the Ems that are used in the model are an overestimation as compared to the real situation.



Total nitrogen (mg/L) Case: ED 2 fullwaq2d edit 8.1b		
	Eems Dollard	
Deltares		

Figure 3.22 Modelled and measured total nitrogen concentrations in the surface water at three MWTL stations

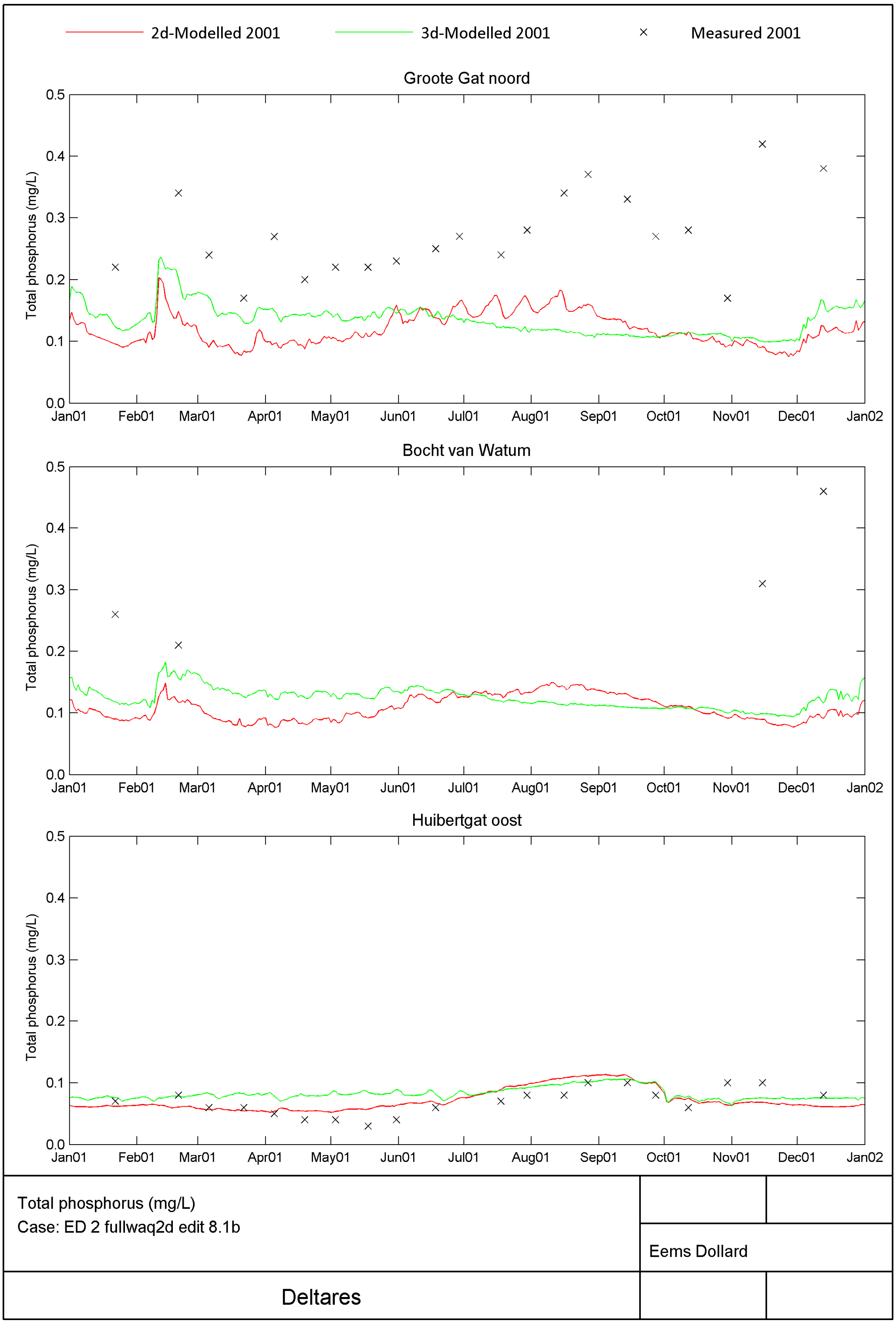
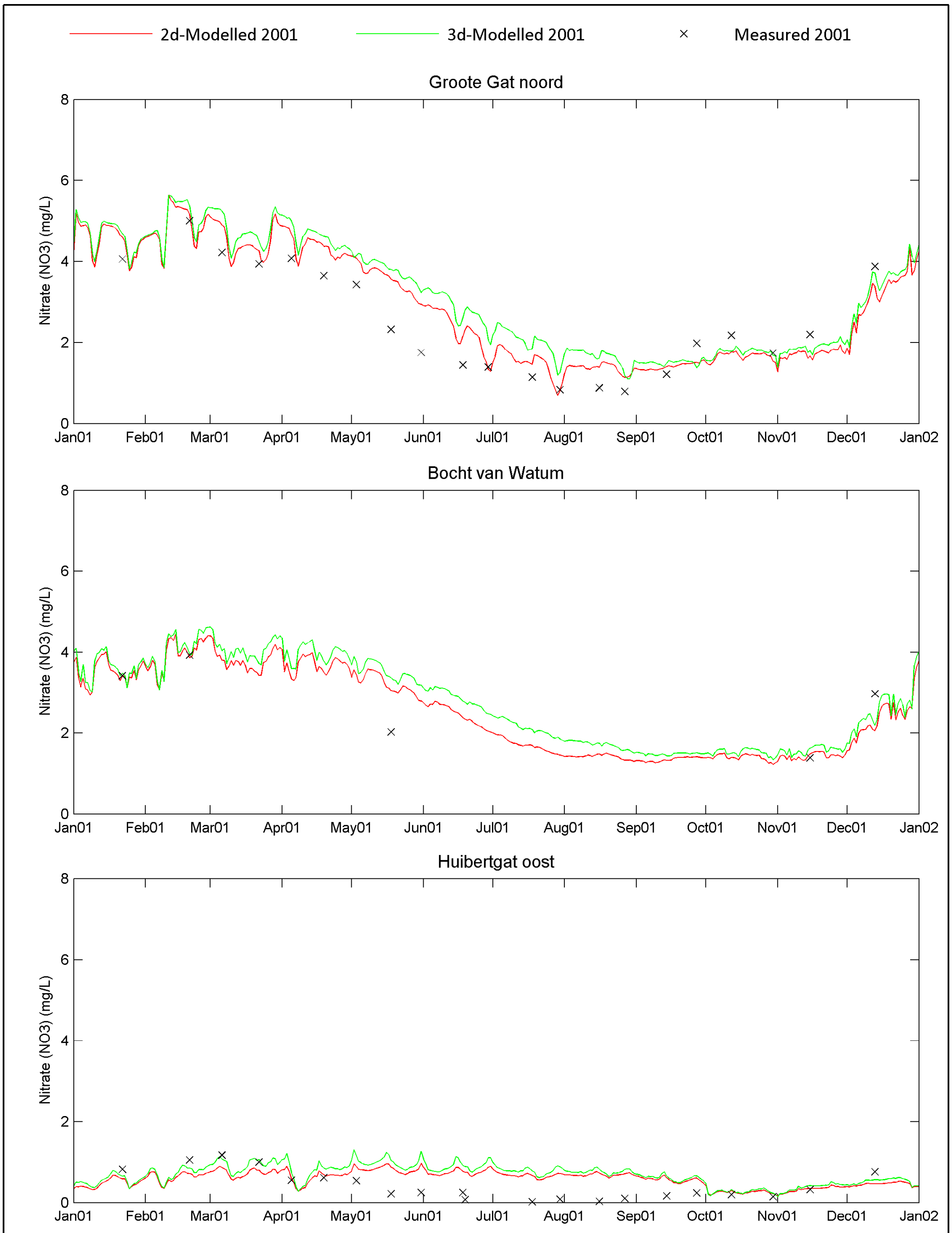
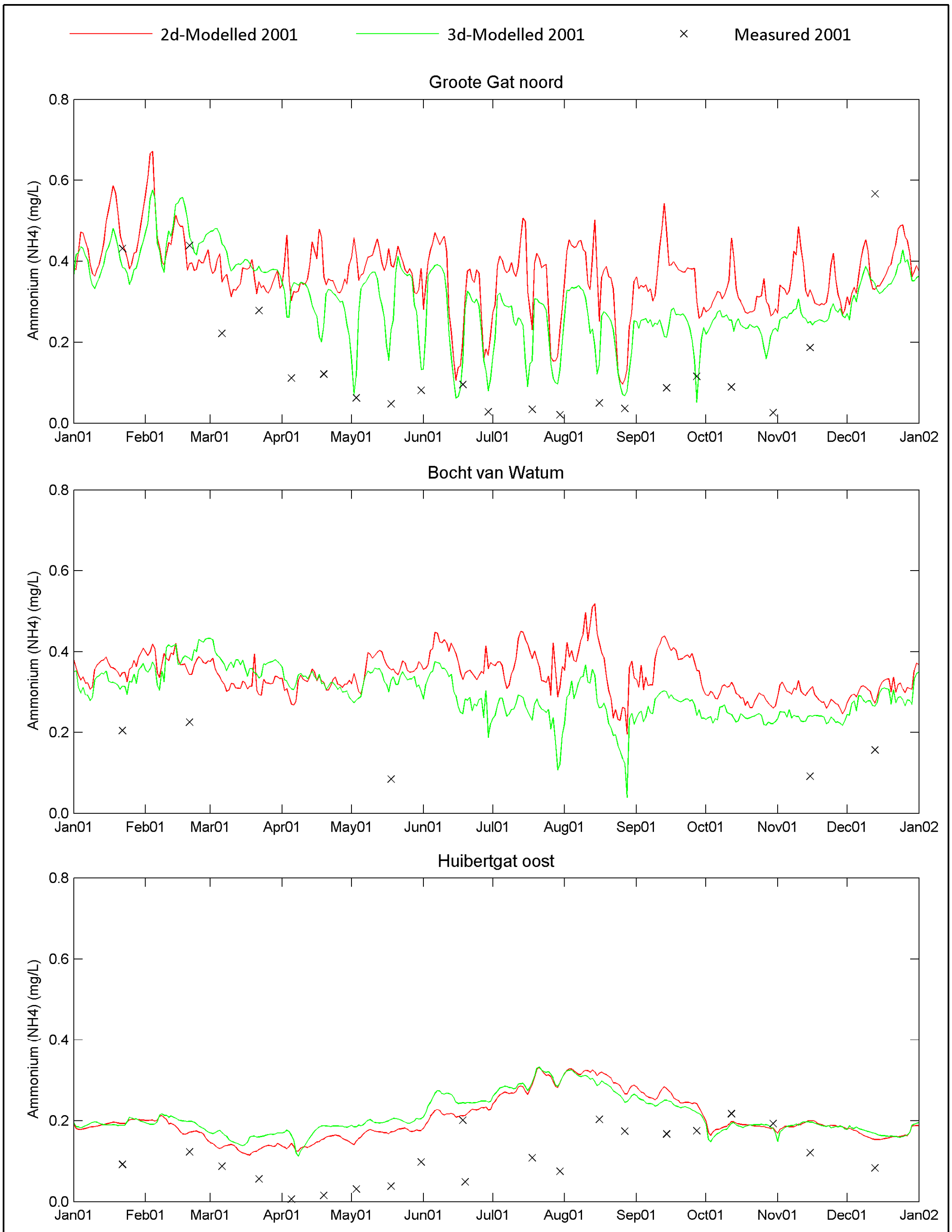


Figure 3.23 Modelled and measured total phosphorus concentrations in the surface water at three MWTL stations



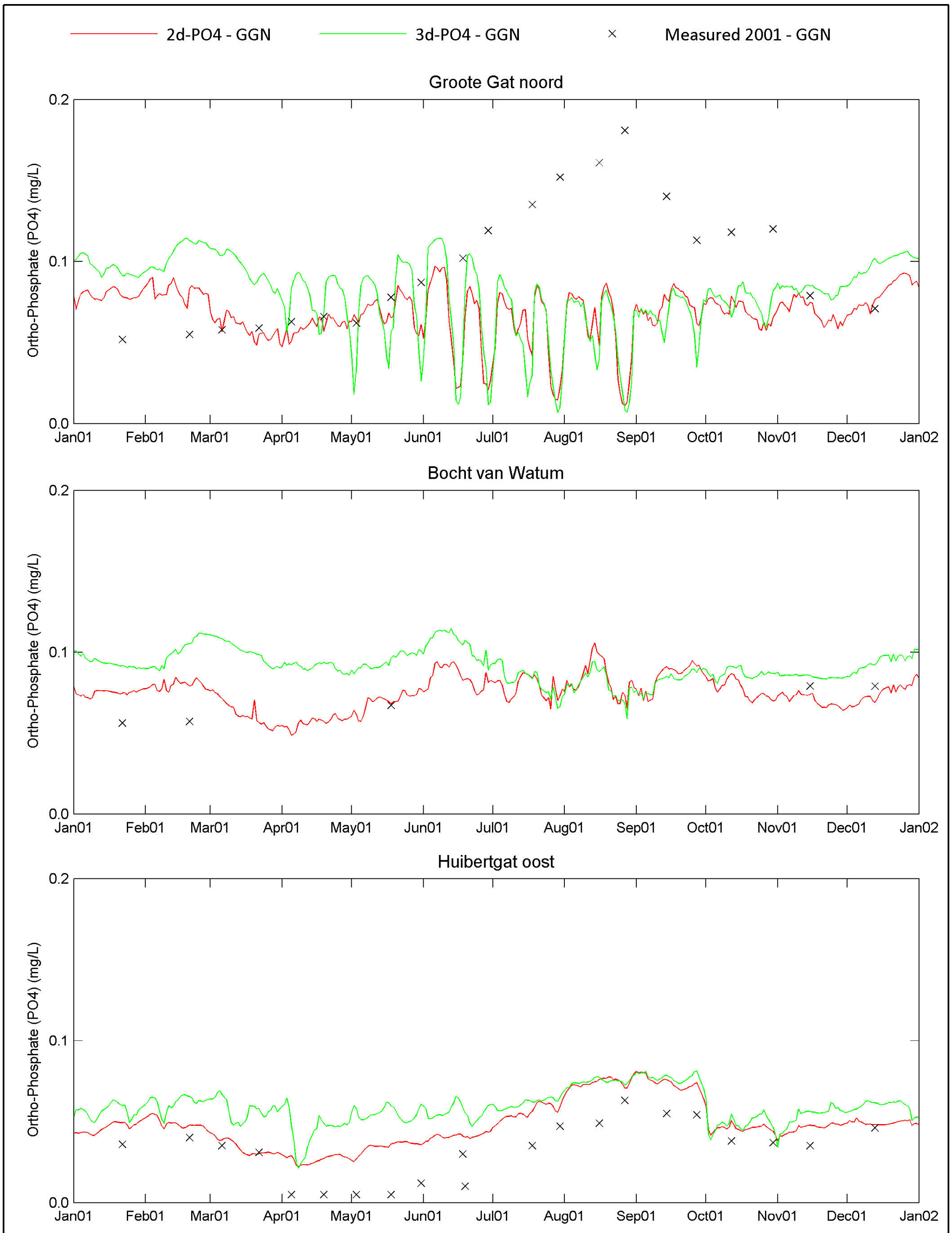
Nitrate (NO3) (mg/L) Case: ED 2 fullwaq2d edit 8.1b		
	Eems Dollard	
Deltares		

Figure 3.24 Modelled and measured nitrate concentrations in the surface water at three MWTL stations



Ammonium (NH4) (mg/L) Case: ED 2 fullwaq2d edit 8.1b		
	Eems Dollard	
Deltares		

Figure 3.25 Modelled and measured ammonium concentrations in the surface water at three MWTL stations



Ortho-Phosphate (PO4) (mg/L) Case: ED 2 fullwaq2d edit 8.1b		
	Eems Dollard	
Deltares		

Figure 3.26 Modelled and measured phosphate concentrations in the surface water at three MWTL stations

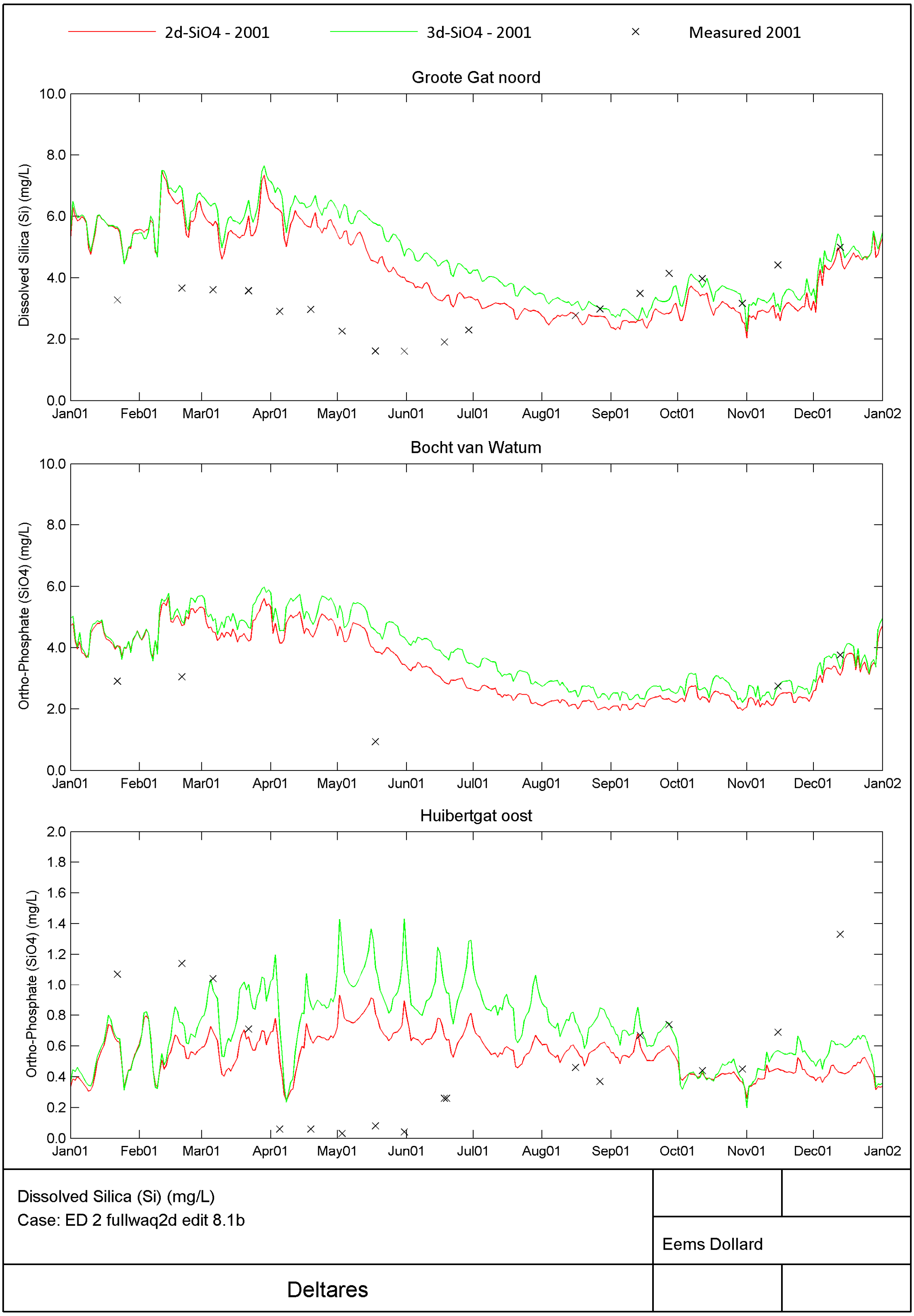


Figure 3.27 Modelled and measured dissolved silica concentrations in the surface water at three MWTL stations

3.4.7 Chlorophyll-*a*

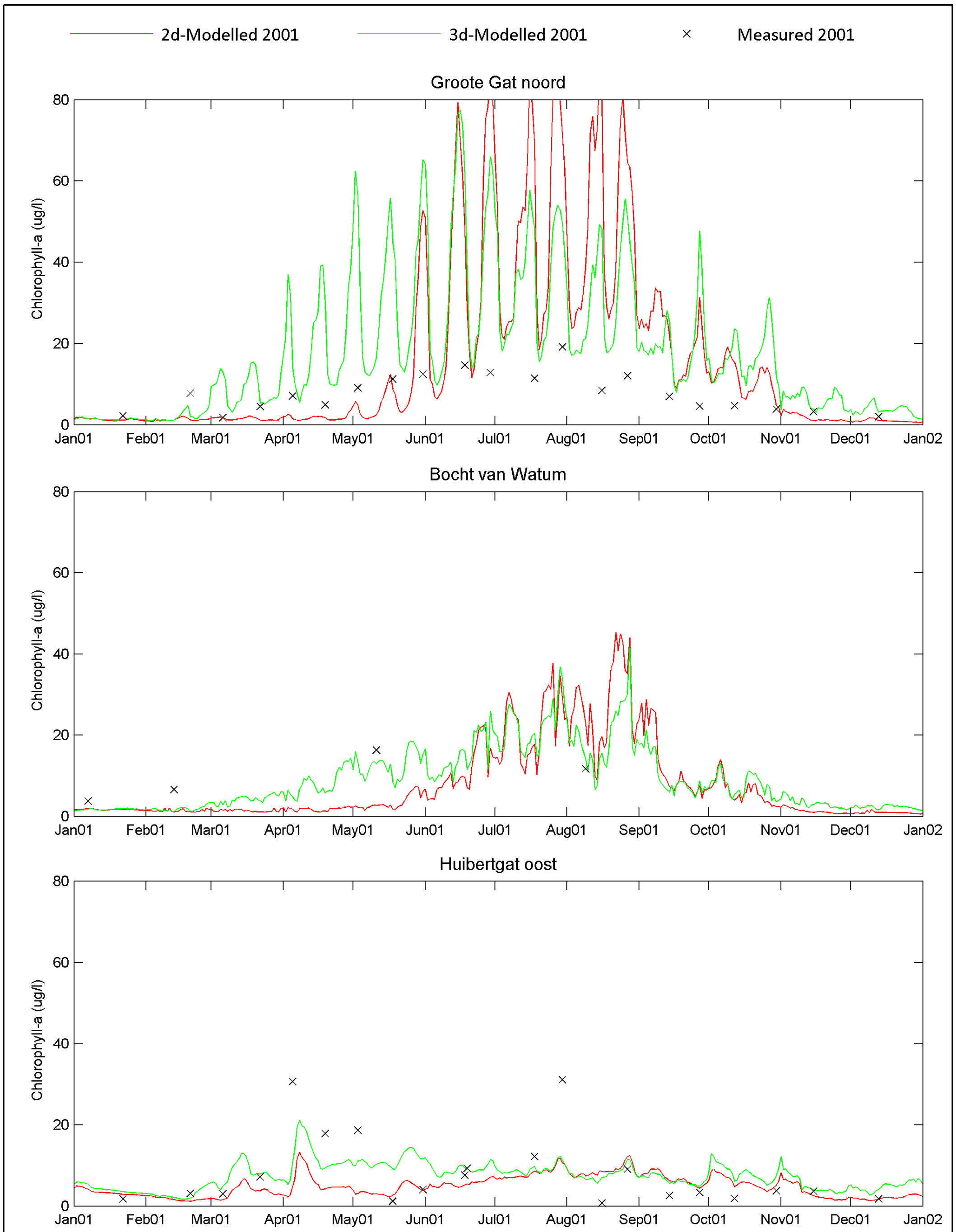
Modelled chlorophyll-*a* patterns, especially at Groote Gat Noord, are highly influenced by the tidal cycle. These oscillations are not seen in the measurements which can be explained by the relatively low monitoring frequency. To our knowledge, no high frequency chlorophyll measurements are available to actually validate the variation shown by the model. Modelled minimum concentrations agree reasonably well with observations, but there is a relative lag in growth in spring as compared to the observations in the 2D version of the model. In the 3D model version, timing of the spring bloom is well reproduced. In late summer, chlorophyll concentrations are apparently overestimated at Groote Gat Noord and Bocht van Watum. At Huibertgat Oost, there are some chlorophyll observations indicating an autumn bloom, which is not well reproduced by the model. Since chlorofyll-a is an output parameter, the overall overestimation of chlorophyll concentrations could also be explained by a difference in chlorophyll-*a* to carbon ratios used in the model, as compared to the actual values. At Bocht van Watum there are not enough measurements to really discuss the seasonal variation of chlorophyll-*a*.

3.4.8 Dissolved oxygen

At all stations, modelled observed oxygen concentrations are generally well reproduced by the model.

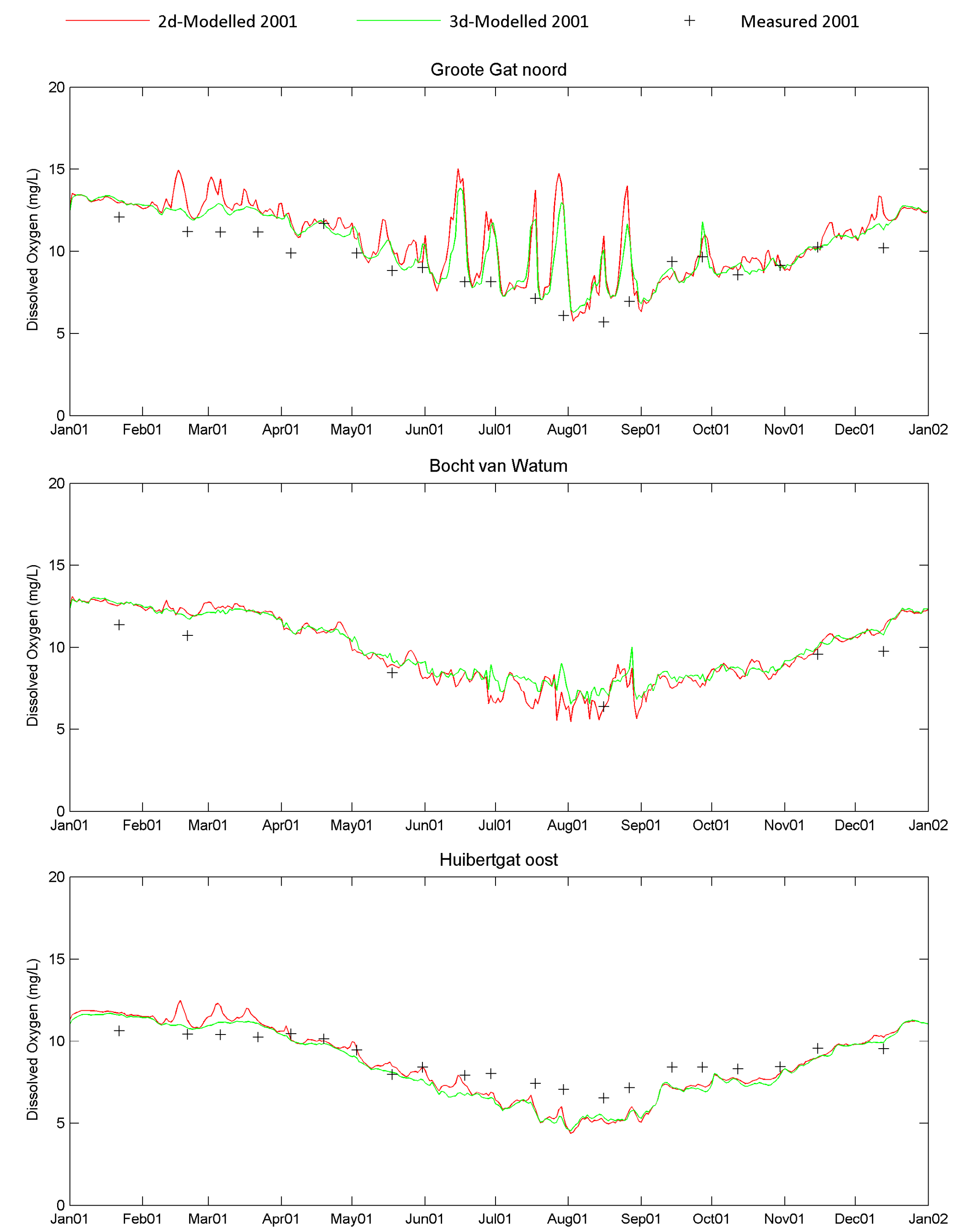
At Groote Gat Noord and Bocht van Watum, concentrations are slightly overestimated by the model during the phytoplankton growth season, which is coincident with the overestimated chlorophyll-*a* concentrations. At Huibertgat Oost, oxygen concentration is underestimated by the model during the autumn. This is coincident with a possible underestimation of autumn bloom (see chlorophyll-*a* results)

Modelled oxygen concentrations follow in general the measured values, but variations due to tidal forcing, which is not visible in the measurements, are dominating especially the summer period. To our knowledge, there are no high resolution measurements to validate this short term variations.



Chlorophyll-a (ug/L) Case: ED 2 fullwaq2d edit 8.1b		
	Eems Dollard	
Deltares		

Figure 3.28 Modelled and measured chlorophyll-a concentrations at three MWTL stations



Dissolved Oxygen (mg/L) Case: ED 2 fullwaq2d edit 8.1b		
	Eems Dollard	
Deltares		

Figure 3.29 Modelled and measured chlorofyll-a concentrations at three MWTL stations

3.4.9 Effects of lower suspended sediment concentrations

The scenario with reduced sediment concentrations was so far only applied on the 2D version of the model. The effects of lower (up to 5 times, seeFigure 3.17) suspended sediments resembling a historical scenario are most obvious in the chlorophyll-*a* concentrations. Due to an improved light climate, algal growth starts earlier during spring. Due to the fact that nutrient concentrations are unaltered as compared to the 2001 reference run, no large changes are seen in the nutrient concentrations. Due to earlier algal growth, the decline of nutrients in spring is more pronounced than in the reference run. Additionally, lower suspended sediments lead to slightly increased oxygen concentrations

Like at Grootegat Noord, lowered suspended sediment concentrations at Bocht van Watum led to lower extinction coefficients of light, and corresponding higher chlorofyll-a concentrations during spring and summer. The effects especially on chlorofyll-a are even more pronounced than in the more inland station Grootegat Noord. The strong effect of the lowered suspended sediment concentration on chlorophyll at the station Bocht van Watum can not easily be explained by primary production at the locations themselves. It is likely due to differences in primary produciton at the shallower areas around the stations, and transport by tidal currents to the station locations.

At Huibertgat Oost station, the effects of reduced suspended sediments are hardly noticable. The reason for this is the relatively low reduction of suspended sediment at this station, and apparently very little transport of phytoplankton out of the estuary that reaches Huibertgat Oost station.

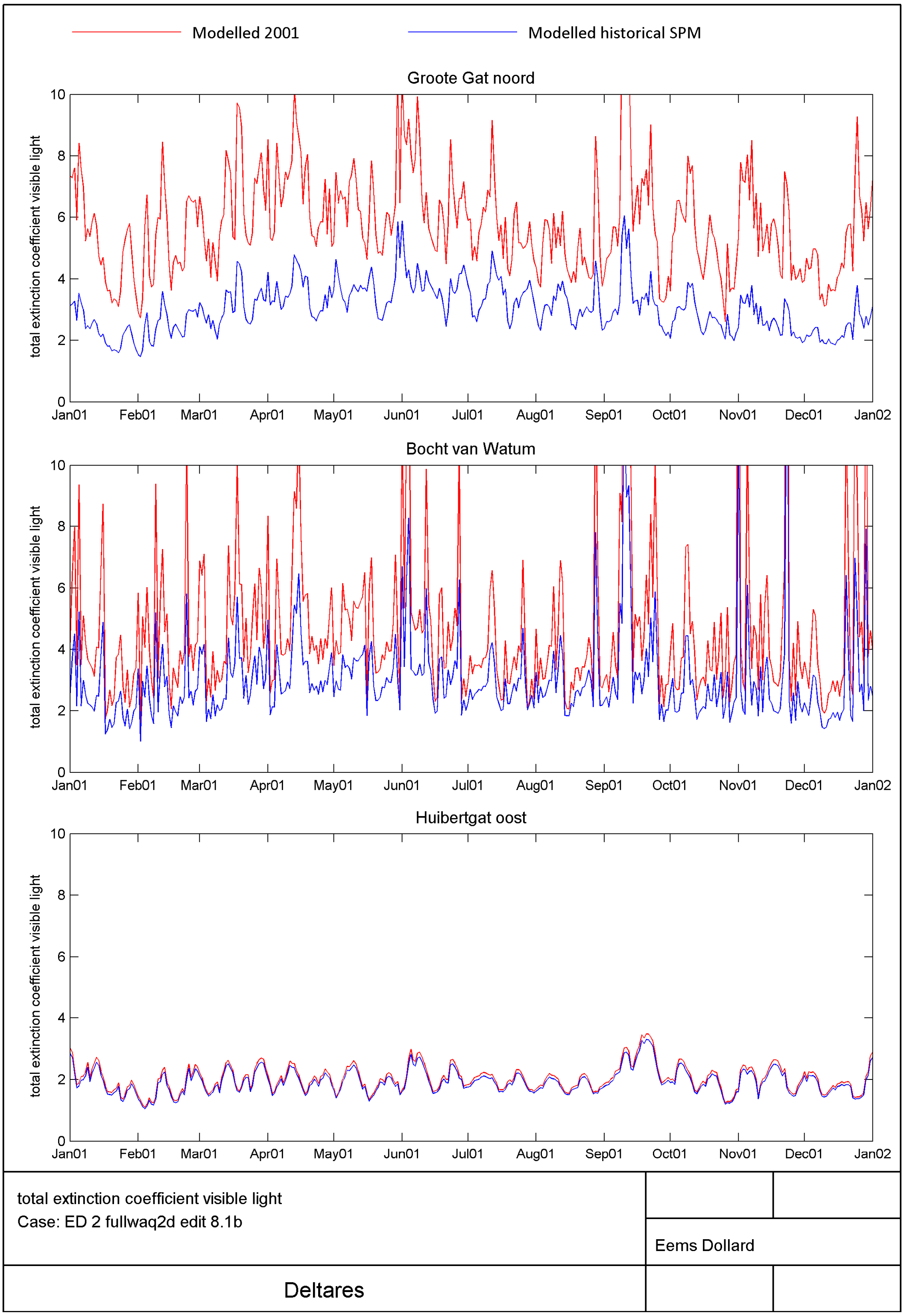


Figure 3.30 Figure 3.31 Modelled total extinction coefficient in the surface water at three MWTL stations using the 2D model version. Reference year 2001 (red) and historical scenario with reduced sediment concentrations (blue).

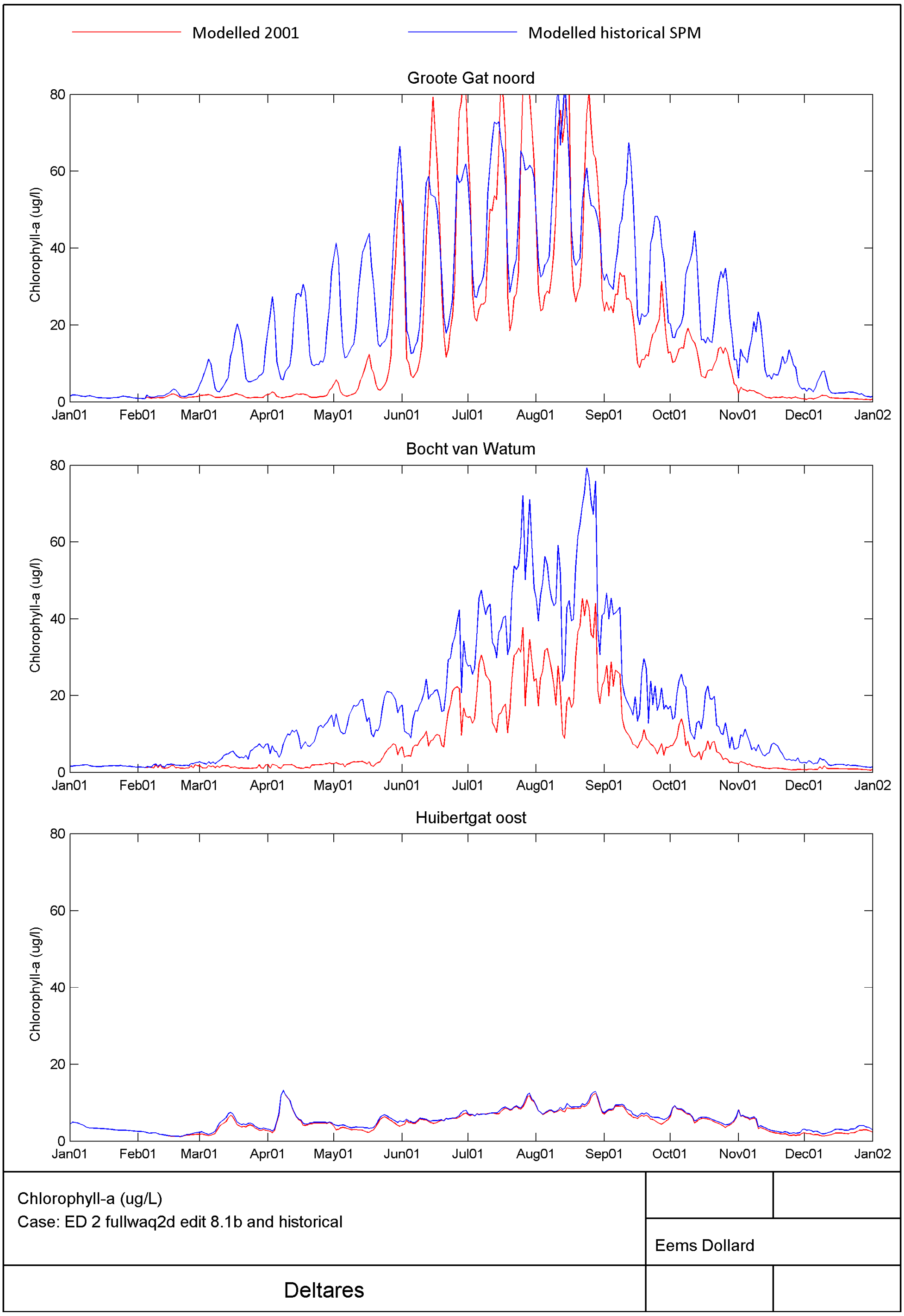


Figure 3.32 Modelled chlorophyll-a concentrations in the surface water at three MWTL stations using the 2D model version. Reference year 2001 (red) and historical scenario with reduced sediment concentrations (blue).

3.4.10 Nutrient budgets through the different parts of the estuary

For calculating the transport of nutrients through the different parts of the estuary, the budgets of 6 different areas are considered (Figure 3.33).

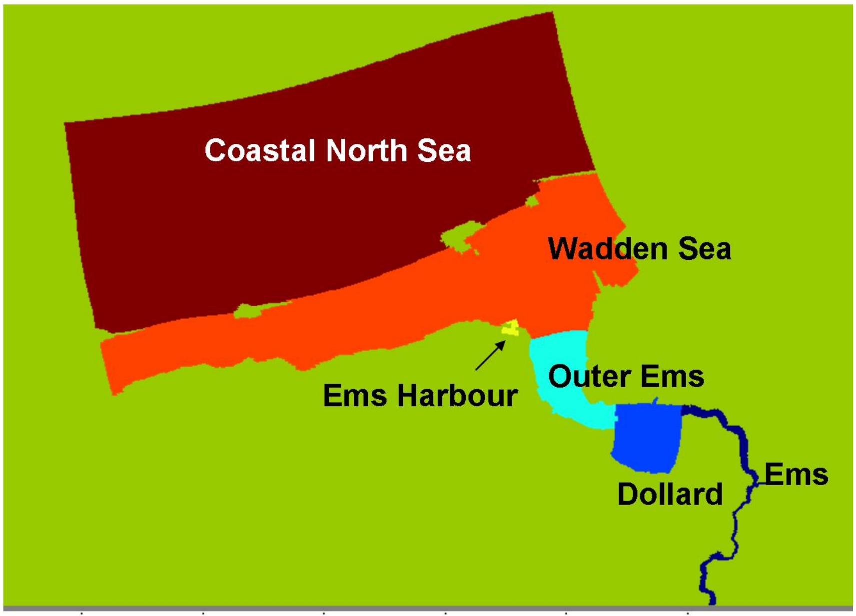


Figure 3.33 Visualization of the 6 different areas for budget calculation of transport, sinks and sources in the estuary.

For the current model set-up, only budgets of total nitrogen were calculated. Phosphorus budgets are likely to be biased since the process of redelivery from the sediments is not included. Nitrogen budgets are also relevant for quantifying the estuary as a net source of nitrogen to the coastal North Sea, where it may contribute to coastal eutrophication effects.
During the reference year, most of the nitrogen coming into the estuary is transported to the Wadden Sea and North Sea (roughly 25 000 tonnes yr^{-1}). Only a small portion is retained in the estuary, in the order of 2 000 tonnes yr^{-1}.

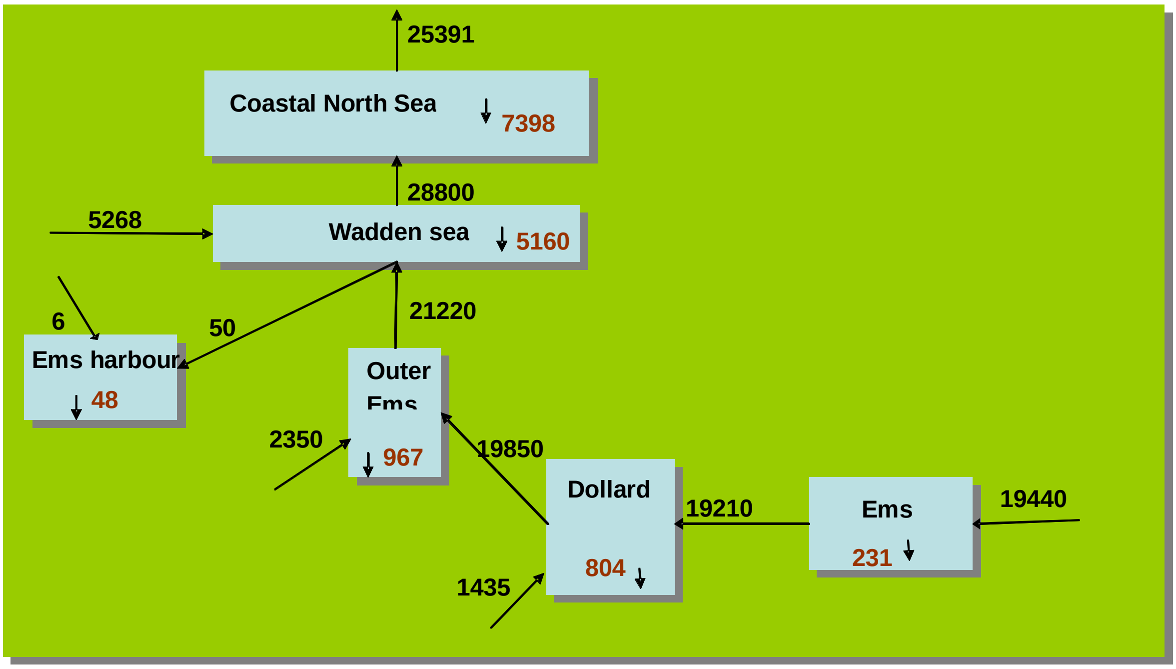


Figure 3.34 Nitrogen (tonnes/yr) budgets through the different parts of the estuary for the reference year 2001

These figures compare well with estimated budgets during 1975-1976, estimated by : van Beusekom & de Jonge (1998). They found a seaward transport of 25,000 tonnes nitrogen yr^{-1}, and a retention in the sediment of less than 1 000 tonnes nitrogen yr^{-1}(Table 3.5).

Table 3.5 Nutrient budgets during 1992 in the estuary in Tonnes N/yr (recalculated from: van Beusekom&de Jonge (1998).

	Phosphorus		Nitrogen	
	Input	Output	Input	Output
River	1116.5	0	30520	0
Atmosphere	21	7	504	6440
Sea (dissolved)	0	927.5	0	24584
Sea (particulate)	427	0	2044	0
Sediment	0	630	0	868
Total	1564.5	1564.5	33068	31892

The main reason for the small differences in overall nitrogen budgets of the estuary is that nitrogen concentrations in 2001 are more or less the same as they were in 1992 (Figure 3.35).

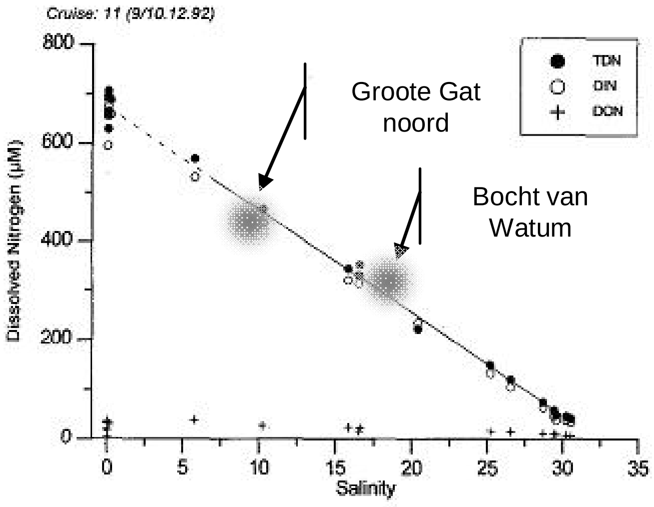


Figure 3.35: Total nitrogen concentrations as a function of salinity in 1992 (van Beusekom & de Jonge 1998). The measured values in the modelled year from the current study (2001) are indicated as grey clouds.

3.4.11 Nitrogen budgets at reduced suspended sediment scenario

Nitrogen transport at lowered suspended sediment concentrations were slightly different as compared to the reference year. The low retention was elevated in such a way that approximately 2 800 ktonnes N yr^{-1} was retained in the estuary, as compared to 2000 in the reference year.

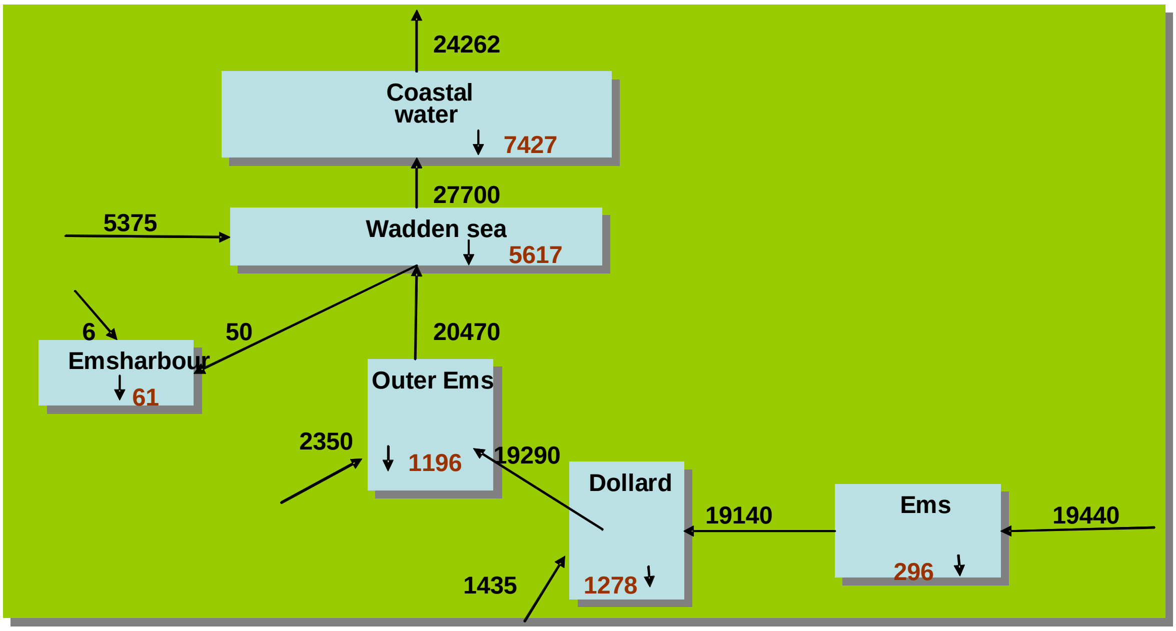


Figure 3.36. Nitrogen (tonnes/yr) budgets through the different parts of the estuary for the reference year 2001, however with suspended sediment concentrations from 1954.

3.4.12 Total primary production in the different sections of the estuary

Total primary production in the Dollard, Outer Ems and to a lesser extent in the Wadden Sea is to a large extent (>50 %) caused by benthic producers. Total primary production in the outer parts is larger, because of the larger area. In the river part (Ems) primary production is relatively strongly enhanced when historic suspended sediment concentrations are applied. This is due to the very high concentrations at present, and the relatively strongest reduction in that part of the model grid.

Nevertheless, also in the estuary, the total production in terms of nitrogen is enhanced by the historic, lower suspended sediment concentration. Lower suspended sediment results in higher production, illustrating the limiting factor of light due to high suspended sediment concentrations in the estuary.

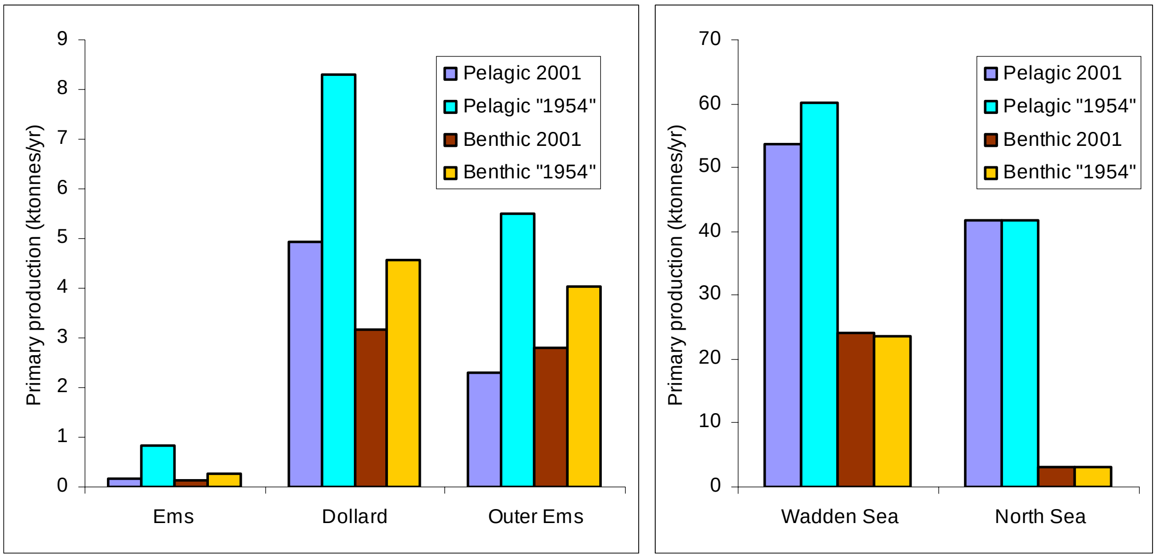


Figure 3.37 Total primary production in ktonnes nitrogen/yr. Note different scales.

3.4.13 Spatial distribution of benthic primary production

Benthic primary producers in the model are restricted to the deepest water layer and are not transported. Therefore, they only occur in very shallow areas and tidal flats, where enough light is available averaged over the tidal cycle (see Figure 3.38). Area-specific biomass of benthic diatoms was in the order of magnitude of 0.5 – 2 g m^2.

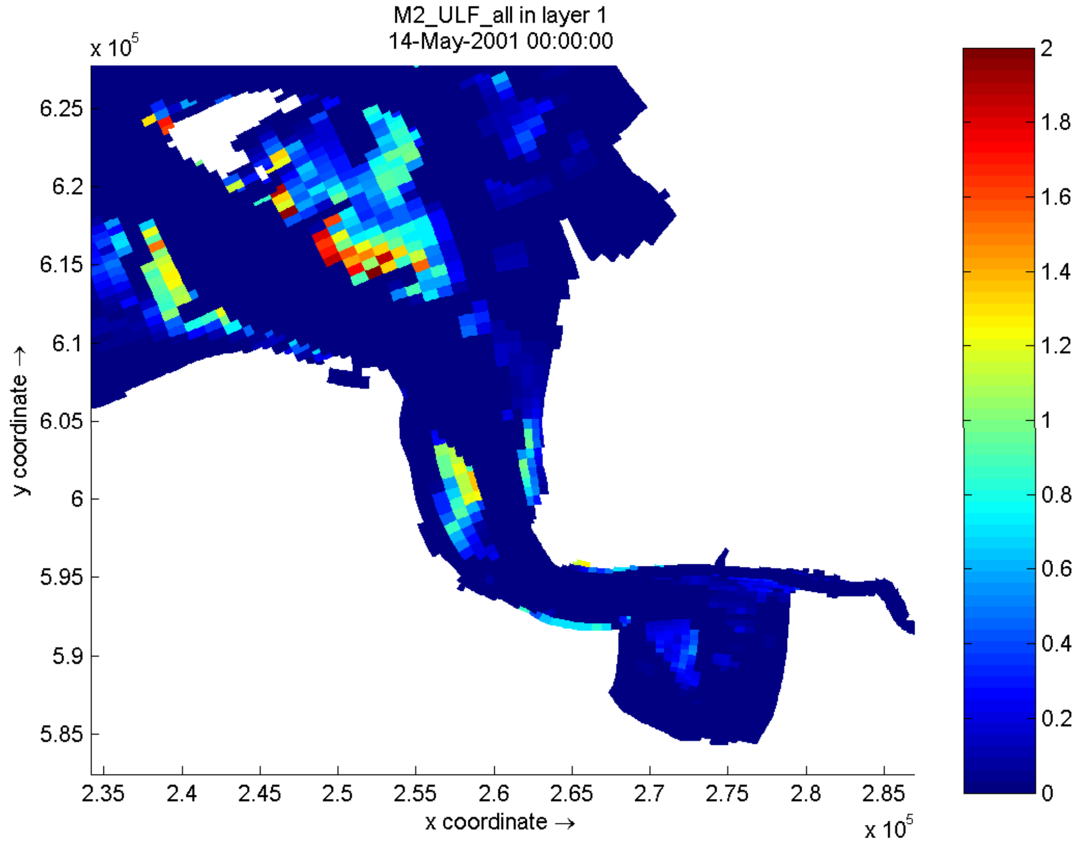


Figure 3.38 Snapshot of benthic diatoms biomass (in mgC/m2) in the Dollard area modeled with Delwaq BLOOM. Growth is restricted to shallow areas due to light limitation.

The modelled biomass of benthic diatoms can at the moment not directly be compared to measured values. For 1977, average sediment chlorophyll-*a* concentrations varied from 20 – 200 mg.m-2 at 6 stations in the Ems estuary (Figure 3.39, from de Jonge & Colijn 1994). This compares well with the modelled biomass of 0-4 mgC.m-2, assuming a chla/C ratio of 0.01 as used in the model. The spatial distribution of benthic diatom biomass coincides with shallow areas, and is highest at mud flats that run dry part of the day. Concluding, the modelled biomass of benthic diatoms is in the right order of magnitude as expected from historic measurements. The spatial distribution and the temporal trends need to be analysed further. Also, new measurements as planned in a coming project can be used for validation (van Maren et al., 2011).

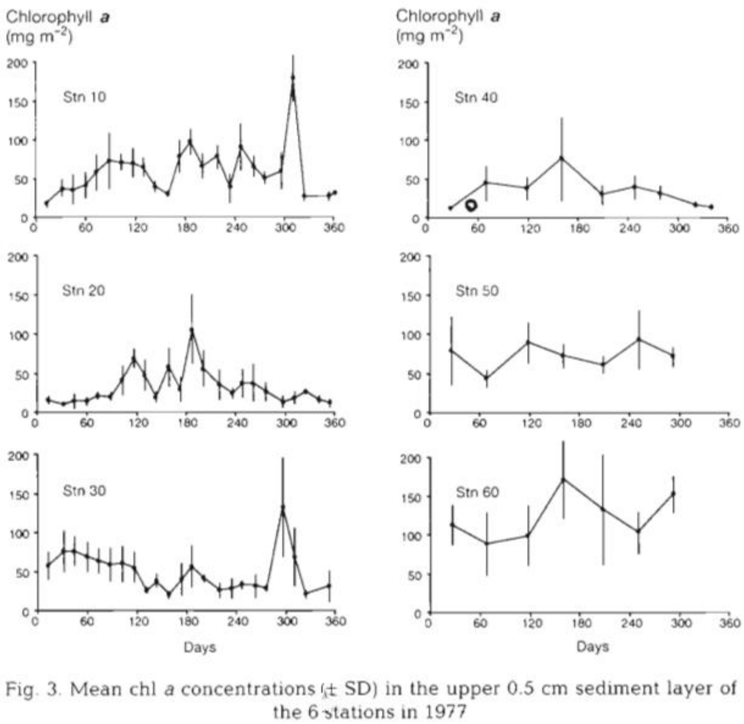


Figure 3.39 Mean chlorofyll-a concentrations in the sediment at 6 intertidal locations in the Ems-Dollard estuary (de Jonge & Colijn 1994).

3.4.14 Limiting factors for primary production

Some information of which factors limit primary production is also obtained by the model. For the three validation stations, the local situation was plotted in Figure 3.40. In this figure, all occurring limitations (different phytoplankton groups can be limited by different factors) are summed up for that particular station. The figure can be interpreted as follows. A value of 1 for a limitation means that that particular factor is limiting for all phytoplankton groups for that particular time interval and location. A value of 0 means that that particular factor is not important for determining the algal growth rate. Any value in between is possible and indicates the proportion of algal groups and part of the time that a certain factor is limiting. At the monitoring stations, light limitation was most common. During shorter periods of time, growth limitation occurred, which means that growth of the algae was limited by their maximum specific production rate.

At Groote Gat noord and Huibertgat Oost, light was continuously limiting phytoplankton growth (limit_e in Figure 3.40). Regularly, growth for some species is limited by the maximum specific growth rate only, meaning that growth has reached a maximum given the ambient temperature and light availability. At Bocht van Watum, light limitation is often not occurring, probably due to the relatively shallowness of the station. Nutrients are never limiting at any of the stations.

On the contrary, benthic primary production was dominantly limited by phosphorus, light, and sometimes by available light (data not shown). This can be explained by the fact that microphytobenthos only grows when light is available, thus to places and periods that sediments fall dry, or remain covered with only very shallow water. At those occasion, light is ample available. So, the distribution of microphytobenthos is limited by light, but once they grow at a certain place, phosphorus becomes the limiting factor for growth in the model.

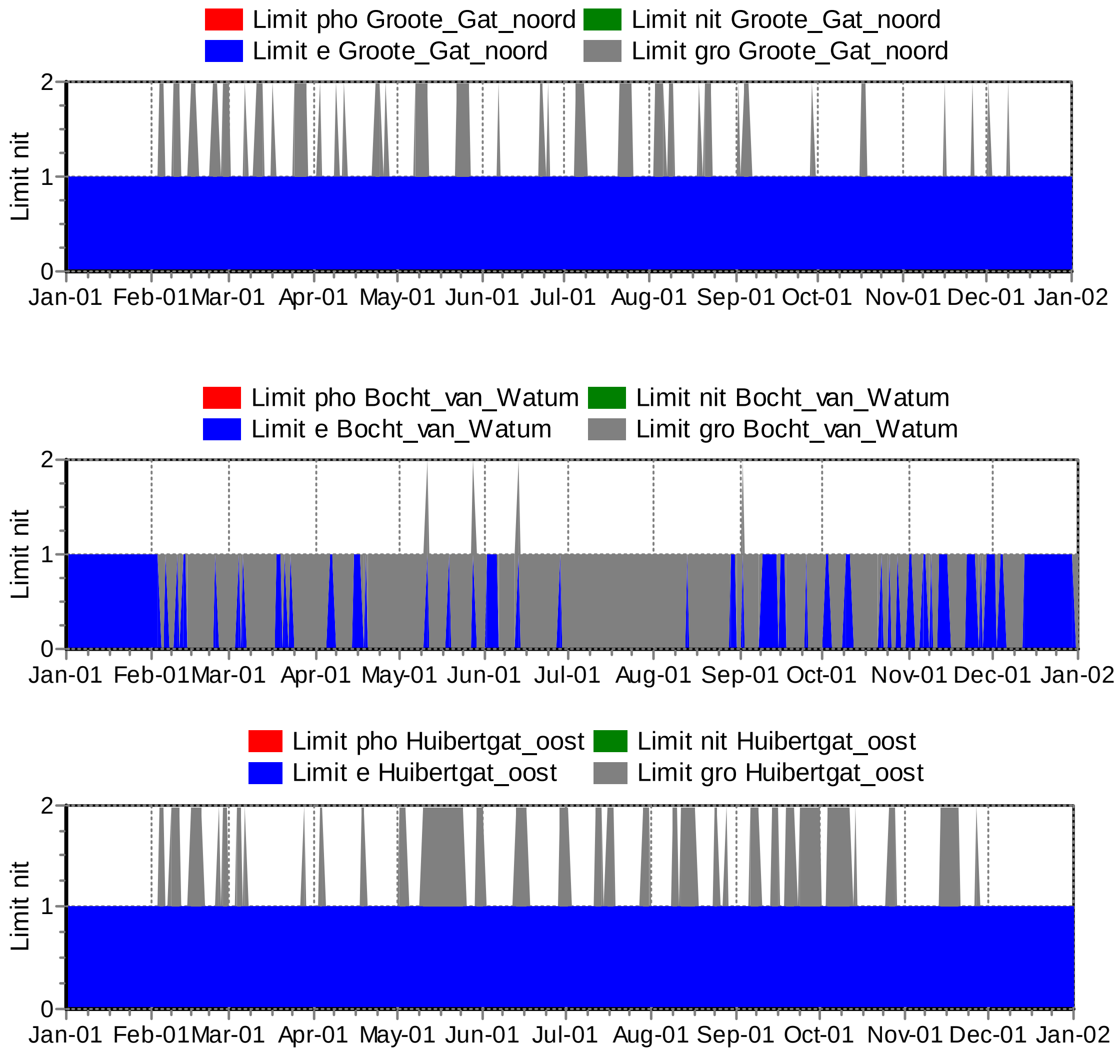


Figure 3.40 Limiting factors for the growth of phytoplankton, where limit e = light limitation, limit nit = nitrogen limitation, limit pho = phosphorus limitation, limit sil = silicate limitation and limit gro = limited only by the maximum growth rate (no other limitation). These results are made using the 3D schematization. 2D simulations gave almost identical results.

3.5 Discussion and conclusions

The current model for water quality and phytoplankton growth is performing technically well and includes both phytoplankton and phytobenthos. It uses a full-year hydrodynamics and modelled suspended sediment concentrations. There is a 2D-version available for fast exploration of a large number of scenarios, and a 3D-version which is performing better in terms of accurately describing phytoplankton biomass. Also, the 3D model predicts the timing of the spring bloom better than the 2D model. During the vertical aggregation, also the suspended sediment fields from the sediment model need to be aggregated vertically. In the current model, this is done by averaging concentrations over all layers. Possibly, a better result with the the 2D model is obtained when only suspended sediment concentrations from the top layer are used, which should be lower than the average and result in earlier spring bloom.

The current model is going to be used in a follow-up project, which aims at calculating the effect of measures taken in the Water Framework Directive on primary production and nutrient transports from the estuary. We conclude that the model is fully operational to test the effects of different suspended sediment concentrations, which has been shown by the historical scenario. However, some fine-tuning regarding the chlorophyll-*a* model results needs still to be done. These adjustments will be done based on a detailed monitoring programme that has started in 2012 (Van Maren *et al.*, 2011) during 2013. Moreover, phytoplankton groups as determined during that monitoring programme will be redefined in the model, based on these measurements and literature values for physiological parameters. The model is also suitable to study the interplay between light and nutrient limitation which occurs along the gradient of the estuary. Although at the monitoring stations, no nutrient limitation occurred, both phosphorus and nitrogen is a limiting factor for primary production in the shallower areas. Moreover, benthic primary production is not predominantly limited by light. Although large parts of the Ems-Dollard are shown to be light-limited with suspended sediment as the main factor determining light attenuation, some effects of nutrient reductions may be expected, especially at the more shallow areas.

4 Habitat suitability

4.1 Introduction

Previous reports have discussed the key components of the Ems-Dollard ecosystem, the principle of habitat suitability modelling, the species of interest and the rationale for focussing on these species, the (derivation of) habitat suitability relationships, the choices and assumptions made for this modelling and the available validation data. This final report focuses first on the spatial and temporal characteristics of the physical parameters that result from the ultimate sediment transport and water quality model runs, and that are used as the input of the Habitat model. Second, habitat suitability maps for several species are made, analysed and compared to available data. Due to the amount of effort involved in this iterative process, this could not be performed for all possibly interesting species. The current study should be seen as an example of what is currently feasible, and what needs to be done to advance the possibilities of this type of studies further. To that end, the effect of the lower historical sediment concentration is analysed for one species, and possible improvements are discussed.

4.2 Representative conditions throughout the year

Some parameters, such as water temperature, vary little spatially. Others, such as depth, will vary little throughout the year. Here, the changes are discussed, in order to display the natural dynamics and to select a set of representative conditions that will be used in the subsequent chapters.
The maps of the parameters that are independent from the water quality calculations, e.g. depth, velocity, wave height and fine sediment fraction, are based on the unaggregated grid of the sediment transport calculations to preserve resolution. The maps of parameters that result from the water quality calculations, such as oxygen, light extinction and Secchi depth, are based on the coarser water quality grid. Spatial differences over the vertical, i.e. due to stratification, are not discussed because the water quality model used a two-dimensional grid and stratification was considered to play a limited role most of the time.
To represent the variation of environmental parameters throughout the year, four ‘seasons’ are used. Each of these seasons summarizes the conditions during two spring-neap cycles – i.e. one month: January corresponds to ‘Winter’, April to ‘Spring’, July to ‘Summer’ and October to ‘Fall’. For all parameters, these time-averaged (i.e. yearly or monthly averages) are used by default, as these characterize a habitat best in most cases. In some cases however, extreme conditions (e.g. minimum oxygen concentration in case of anoxia) are crucial. Where this applies, the use of an extreme condition is mentioned explicitly. As a consequence of the two-dimensional modelling of water quality, all parameters are depth-averaged.

The figures corresponding to the discussed parameters are the size of one page. Therefore they are not all displayed here but in Appendix B, in the order of discussion. The mentioned locations correspond to the map in Figure 4.1.

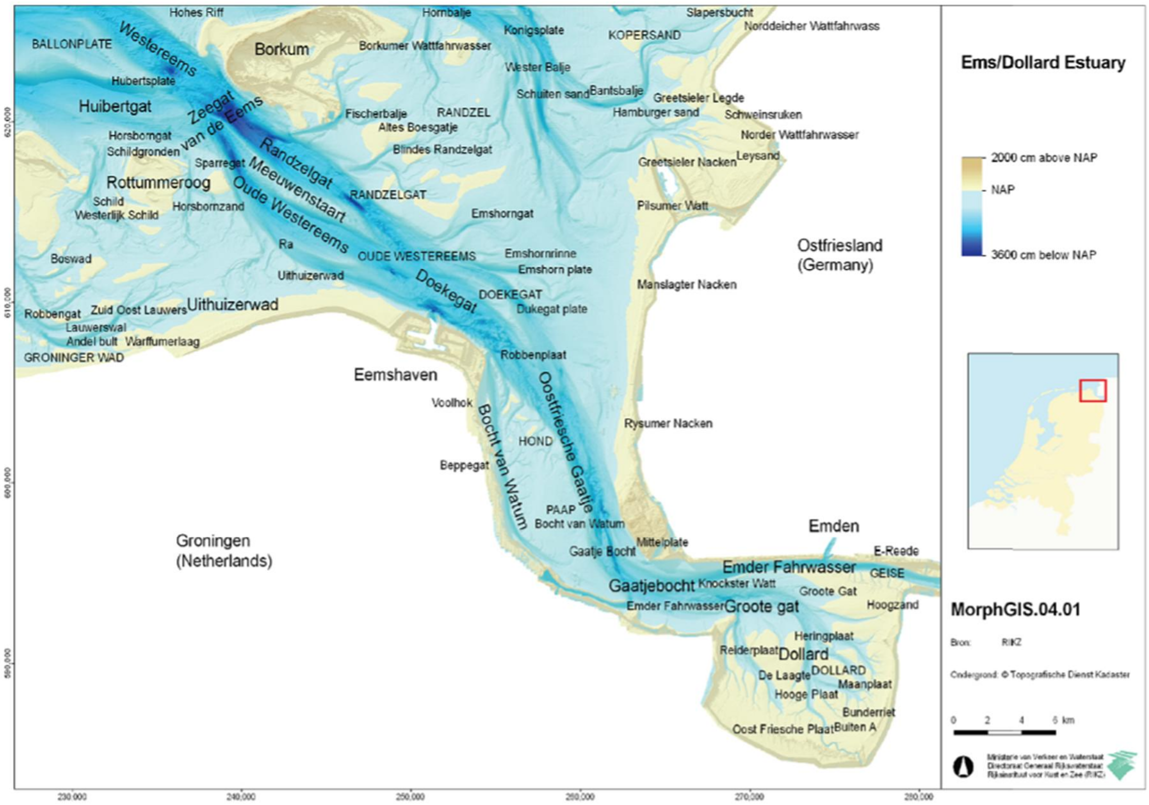


Figure 4.1 Channels, flats and other places in the Ems Dollard estuary.

4.2.1 Depth and duration of dry period

The outer area is characterised by a large deep channel (the Randzelgat, over 30 m deep; Fig 4.1) and large shallow areas behind the barrier islands as well as salt marshes near the coasts of the Netherlands and Germany. The Ems estuary has one present main channel (Oostfriese Gaatje 12-17 m deep) and a former main channel (the Bocht van Watum), separated by the Hond-Paap bank. The banks are rather steep and the estuary is narrow without extensive intertidal areas due to the dikes on both sides. A deep channel (Groote Gat) forms the entrance to the Dollard, which is shallow to intertidal for the majority of its area. The saltmarshes at the back are above mean sea level.

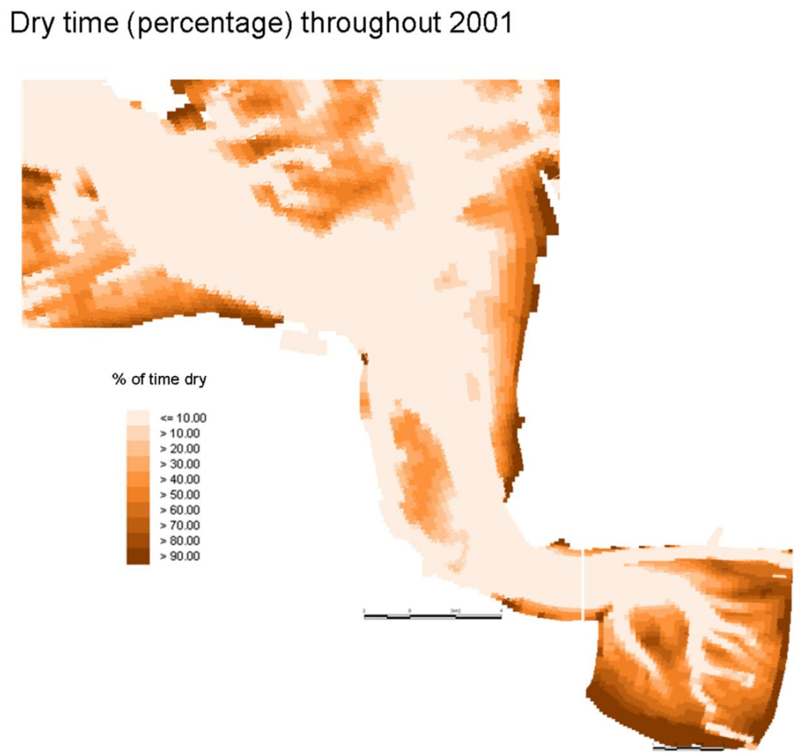


Figure 4.2 Average emersion time throughout 2001.

The depth is considered to remain practically constant throughout the year, apart from the obvious tidal range. The emersion time varied somewhat between the seasons (Fig. B3-5), due to differences in wind setup or river runoff: In all areas, the emersion time in summer was a few percent higher than average, whereas the dry time in fall was shorter.

Remark: The threshold depth applied to determine whether a grid cell is wet or dry was set at 0.4 m. Lower values resulted in areas, e.g. the salt marshes in the Dollard, being classified as wet for the majority of the tidal period, whereas in reality they are dry most of the time. The threshold of 0.4 m might give a small overestimation of the time that a specific area is dry, but gives a more realistic result for the whole area.

4.2.2 Wave height and orbital velocity

In the context of this habitat suitability study, the maximum occurring wave height (Fig. 4.3) is more important than the average wave height: This maximum wave height determines whether epibenthic organisms (such as mussels) are dislodged or not, and whether vegetation is uprooted. Indirectly, the maximum wave height can be regarded as an indicator for the wave dynamics under normal conditions, which are important for mixing and re-aeration.
This maximum wave height, or the bed shear stress or orbital velocity derived from it, is determined by the wind speed, duration, fetch length and the local depth that causes waves to break. The relatively simple approach to calculate these wave heights means that the values should not be used for safety assessments, but they do provide a good measure for dynamics. The values given here are specific for the year 2001; in other years, e.g. with stronger storms, they may differ substantially.

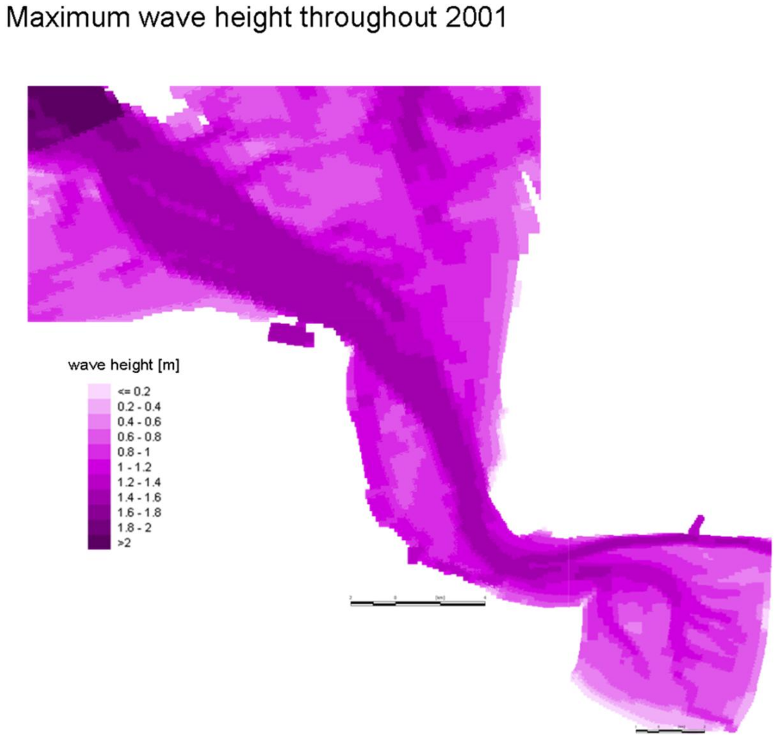


Figure 4.3 The maximum wave height in 2001.

In the outer area, the significant wave height just exceeded 1.6 m in the channels, it was 0.6-0.8 m on most of the higher areas and 0.4-0.5 m close to the coast. Where the foreland is higher due to the presence of salt marshes, the maximum wave height was 0.12-0.4 m. In the Ems estuary, the waves reached 1.5 m in the main channel, 1.1 m in the Bocht van Watum and 0.8 m on the Hond Paap. The lowest values (0.1-0.6 m) occurred near Rysumer Nacken. At the entrance of the Dollard and in the Emden Fahrwasser, the maximum wave height was 1.5 m. In the smaller channels, this decreased to 1-1.3 m, whereas on the flats the waves were 0.6-0.8 m. Near the salt marshes, 0waves did not exceed 0.35 m.

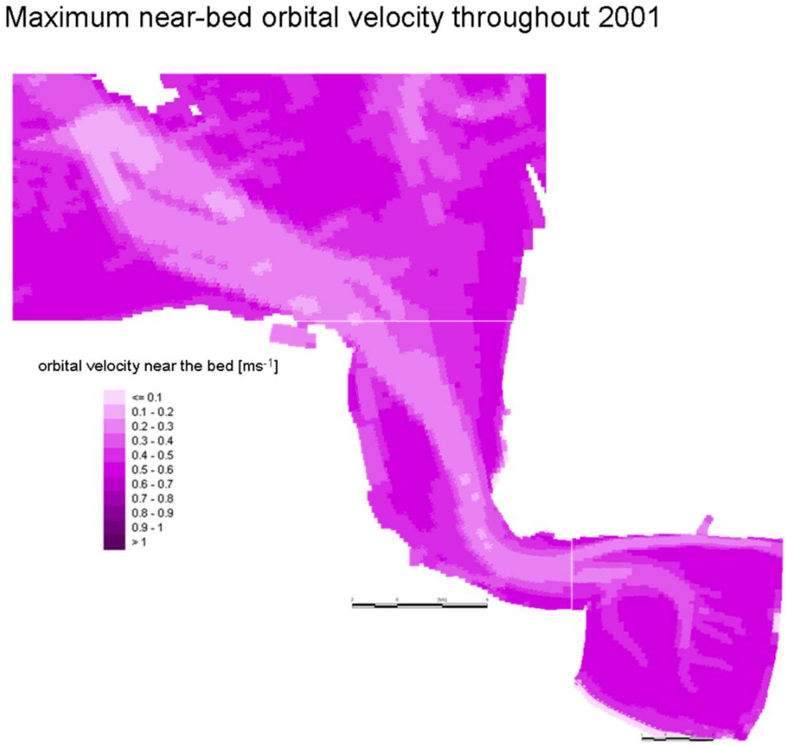


Figure 4.4 The estimated maximum near-bed orbital velocity in 2001.

The maximum orbital velocity (Fig 4.4) is related to the maximum wave height, but also to the wave period, wave length and water depth. For shallow water, Van Rijn (2011) gives the following approximation: Orbital velocity = 0.5 × (Wave height × Wave length)/ (Wave Period × Water depth). For these four input parameters, the maximum values throughout the year were used, as these are likely to occur at the same time, e.g. a storm with setup. Using the average water depth would have lead to an overestimation of the velocities. Note that the approximation is not valid for deep water –there it overestimates the orbital velocity-, but that does not pose a problem for this study because here the species that are related to the orbital velocity, are unlikely to occur in deep water.
The lowest orbital velocities occurred in the deepest areas: The Randzelgat (0.18 ms^{-1}) and the entire main channel (0.2-0.3 ms^{-1}). The highest values occurred on the intertidal areas in the Dollard and Outer area and on the Hond-Paap; up to 0.54 ms^{-1}.

4.2.3 Flow velocity

The depth averaged flow velocity was highest in the large channels in the outer area and the Ems estuary: 0.4 ms^{-1} and higher (Fig. 4.5). In the Dollard, the velocity in the channels was lower. The tidal flats in the outer area as well as the Dollard were characterised by flow velocities between 0 and 0.2 ms^{-1}, versus 0.1 to 0.2 ms^{-1} in the Ems estuary. This means that the Ems estuary has little low-dynamic area compared to the other to regions. The depth averaged flow velocity was practically the same for all seasons.

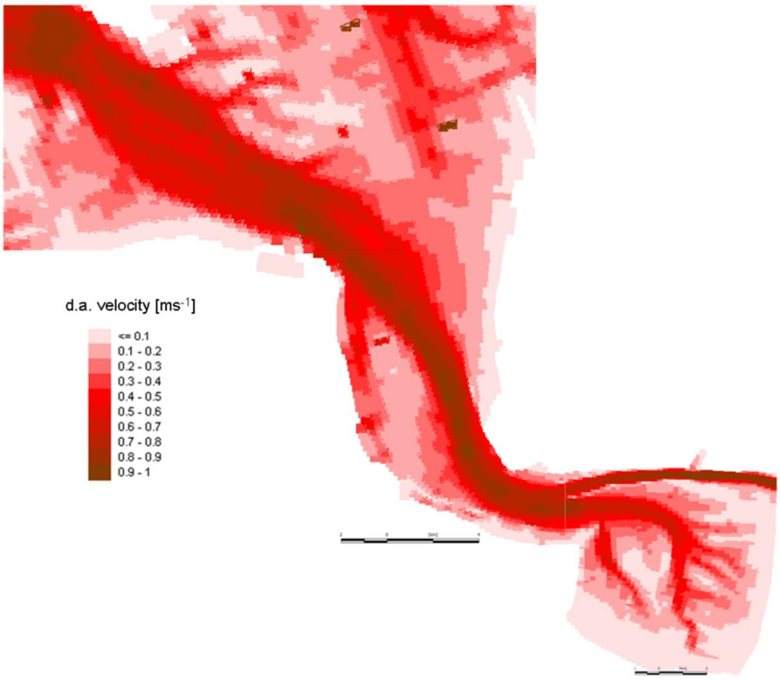


Figure 4.5 Depth-averaged velocity throughout 2001.

4.2.4 Sediment concentration

The sediment concentration (Fig. 4.6) varied little throughout the year when averaged over monthly periods (Fig. B9). Also, the concentration was quite uniform for most of the Ems-Dollard (below 50 gm^{-3}), except for the edges and shallow areas where concentrations were orders of magnitude higher.

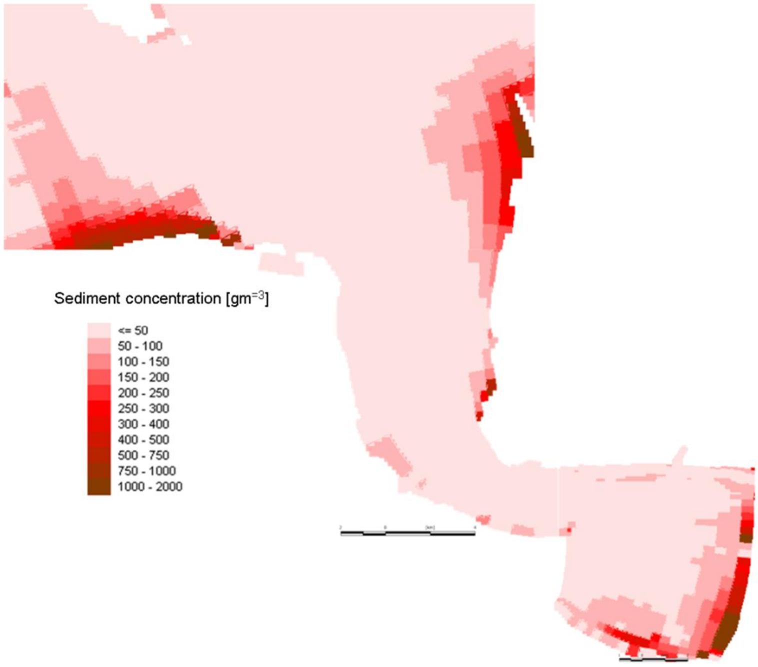


Figure 4.6 Depth-averaged sediment concentration throughout 2001.

The historical concentration (Fig B11) was clearly lower, which is best visible in the shallow areas in the Dollard and outer area, as well as the south part of the Bocht van Watum. However, the legend is somewhat biased towards these higher concentrations, which means that variations in the range below 50 gm^{-3} (50 mgl^{-1}) are not visible. The spatial differences in low sediment concentrations show better in the maps of turbidity

4.2.5 Turbidity (Secchi depth)

The turbidity, expressed as Secchi depth because that matches the habitat suitability curves, varied considerably spatially and moderately temporally. Throughout the year, the main channel at the North Sea side was the clearest (around 0.7 m visibility; Fig. 4.7)), the entire Dollard area (0.1-0.3 m) was the most turbid, i.e. had the lowest Secchi depth. Other turbid areas were the shallow areas in the Wadden Sea and around Rysumer Nacken (0.15-0.4 m). Generally, the deeper areas had slightly better visibility than shallow areas. The water just in front of the harbour of Delfzijl was remarkably clearer than the surrounding water.

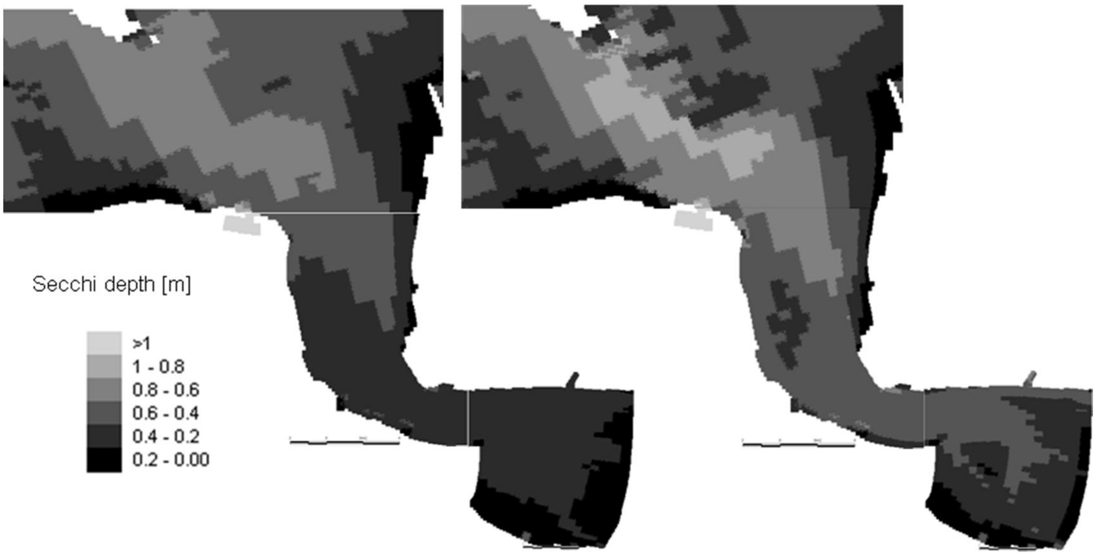


Figure 4.7 Secchi depth throughout 2001(left panel) and in the historical scenario (right panel)

Overall, the water was slightly clearer in winter and slightly more turbid in spring. The difference between the seasons was small however; typically less than 10 cm on most locations. Because the data are averaged over an entire month, possible peaks in turbidity due to short-term events such as storms are not visible. The increased turbidity in spring is likely due to algal growth: The Secchi depth is a function the light extinction coefficient and the Poole-Atkins coefficient; the latter depends on the concentration and composition of suspended sediment, phytoplankton and humic acids.

4.2.6 Historical turbidity

As a result of the lower imposed suspended sediment concentration in the historical scenario, the turbidity throughout the Ems-Dollard is lower than in the 2001-scenario (Fig. 4.7). In the outer area, in spring, the difference in Secchi depth between the two scenarios was between circa 10 cm in the main channel to 2 cm on the intertidal areas. In the Ems estuary, the difference was slightly more apparent: in the 2001 scenario, the Secchi depth ranged from 48 cm at the sea side to 24 cm near the Dollard, whereas in the historical situation the range was 60 to 44 cm. The shallow areas Hond-Paap and Bocht van Watum were the most turbid with 33 cm (2001: 24 cm). In the Dollard, the channels and intertidal areas had a Secchi depth of

15-22 cm in 2001, whereas the water near the saltmarshes in the back was more turbid. The historical model results also showed high turbidity near the marshes, but considerably clearer water (25-40 cm Secchi depth) in most of the area.

4.2.7 Temperature

The depth-averaged water temperature was spatially homogeneous, for all areas (Figure B.16). Stratification is unlikely due to the strong mixing. The maximum temperature was 22 ºC, the minimum slightly below 1 degree and the average over the year 10.7 ºC. In winter, the average temperature was lowest: 1 ºC. In spring, summer and fall the average temperatures were 7.5, 19.6 and 14.2 ºC, respectively.

4.2.8 Salinity

In all seasons, a clear gradient in salinity existed (Fig. 4.8): The part of the outer area closest to open sea was saltiest with 30-31 gl^{-1} throughout the year, whereas the area around Nieuw Statenzijl in the Dollard was freshest with 0-2 gl^{-1} throughout the year. The Ems River, as well as areas in and near the harbour of Delfzijl were also relatively fresh (5-13 gl^{-1}). Spring showed the lowest salinity values, fall the highest. Winter was comparable to spring and summer to fall. The outer area showed the most constant salinity during the year: ± 2 gl^{-1}. The Dollard was the most variable over the year with a variation of around ± 3.5 gl^{-1}.

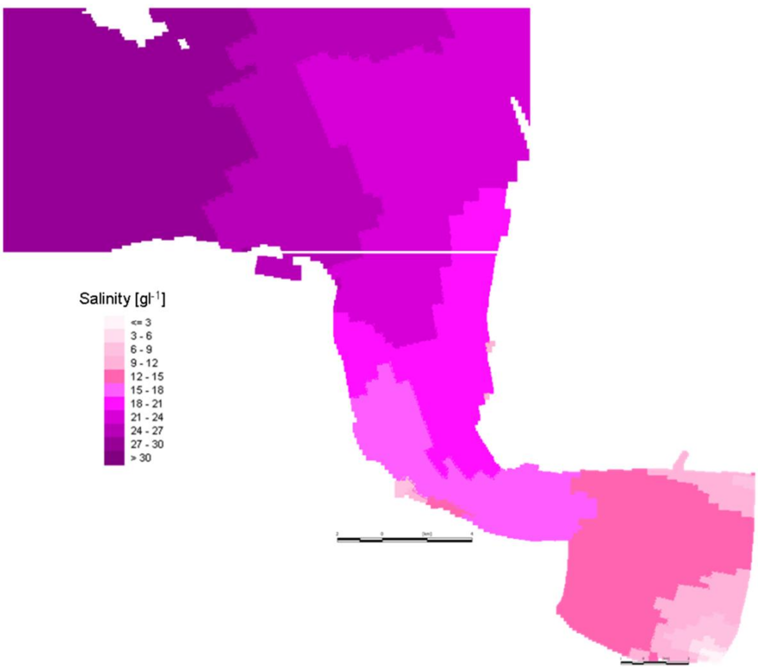


Figure 4.8 Salinity throughout 2001

4.2.9 Oxygen

The oxygen concentration was one of the most variable parameters throughout the year: In all areas, the concentration was highest in winter and lowest in summer (Figs. B20-22). In winter, the concentration was spatially nearly homogeneous, with just over 13 mgl^{-1} throughout the relatively fresh Dollard decreasing to slightly less than 13 mgl^{-1} in the salty outer area, independent of bathymetry. In other seasons, the influence of the bathymetry was more apparent, as deeper parts generally exhibited a lower oxygen concentration. In summer, the lowest time-averaged concentration in the Dollard was 8.5 mgl^{-1}, though the concentrations in the Emder Fahrwasser and Emden harbour were 7.4 and 5.5 mgl^{-1}, respectively. The summer concentrations in the deeper parts of the Oostfriesche gaatje were between 6.5 and 7 mgl^{-1}, whereas in the vicinity of the Hond-Paap and near the German coastline they ranged between 10 and 12 mgl^{-1}. The water in the deep main channel in the outer area contained around 6 mgl^{-1} oxygen, in the shallower areas between 9 and 11 mgl^{-1}.

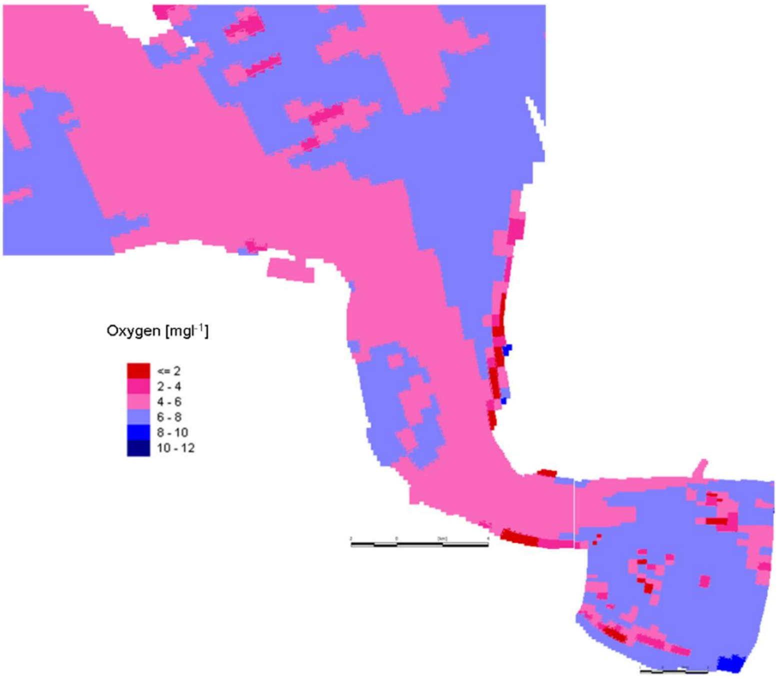


Figure 4.9 2nd percentile oxygen concentration throughout 2001.

Because these monthly-averaged oxygen concentrations are generally sufficient for most biota, the lowest values (i.e., minimum concentrations) that occurred during shorter periods in summer are of interest too. The minimum concentrations resulting from the water quality simulations can be minima that occurred for one hour (one time step) only. Consequently, it is uncertain whether they were harmful or not. Therefore, the 2nd percentile is used (Figure 4.9): This is the value below which 2% of the observations in one year fell, which is equivalent to 8.3 days and as such long enough to be possibly harmful.

The shallow and lower intertidal areas (behind the barrier islands, the Hond-Paap and Bocht van Watum, the Dollard) showed the highest 2nd percentile oxygen levels; between 6-8 mgl^{-1}. The higher intertidal areas a bit farther from the North Sea displayed lower levels: 5-6 mgl^{-1} on the middle of Hond-Paap, and 1-5 mgl^{-1} in the Dollard and near Rysumer Nacken. In the deeper areas, i.e. the main channel, the 2nd percentile oxygen concentration rose very gradually from 4.4 mgl^{-1} on the seaward side to 5.8 mgl^{-1} near Emden Harbour.

4.2.10 Bed composition

The lowest percentages of fines in the bed (Fig. 4.10) were found in the seaward part of the main channel: Around the Randzelgat and the Doekegat there were no more than 2% fines in the bed. The main channel from Borkum to the Gaatjebocht contained slightly more fines; 2-8%. Similar values were encountered in parts of the intertidal areas behind Borkum, whereas the intertidal areas behind Rottumeroog contained considerably more fines (ca. 6-20 %). The borders of the main channel contained a very high percentage of fines (40-80%), especially near Uithuizerwad and Dukegat Plate. The shallow areas near both the German and Dutch coasts and the area (north)west of Greetsieler Nacken had the most fine material however, with 70-100 % fines.

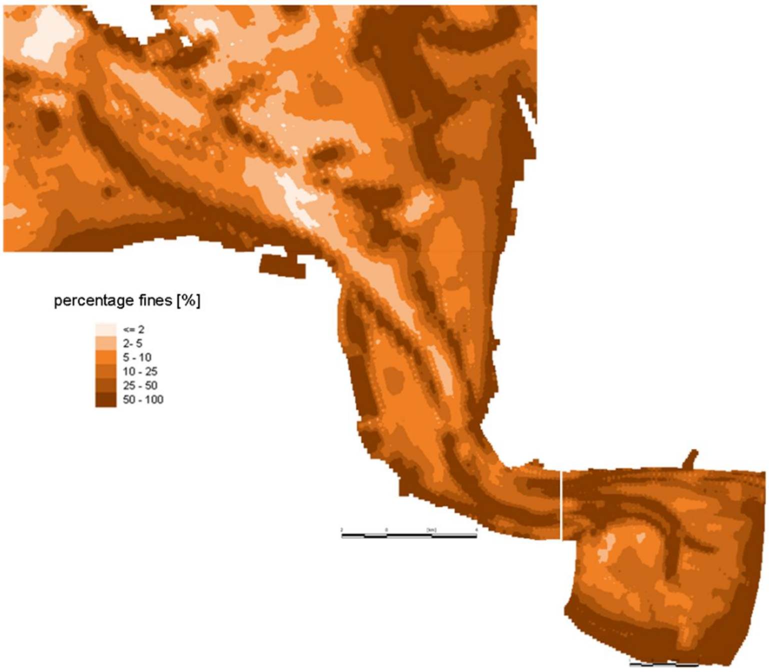


Figure 4.10 Percentage fine material in the bed throughout 2001.

Further in the estuary, the Bocht van Watum, the area around Delfzijl Harbour and the convex (outer) bend of the Gaatjebocht were characterized by high percentages of fines (15-90%). The largest part of the Hond-Paap had between 7-9% fines, with slightly more fine material (10-15%) around the edges and on the Hond. At the back of the estuary, the channels contained more fines than the intertidal flats: Both the Groote gat and Emder Fahrwasser had between 20-95% fines, versus typically 6-20% for most of the intertidal area. Only the higher intertidal areas close to the dikes and saltmarshes contained more fines: 60-99%.
The variability of the fraction of fine material throughout the year was very small: the standard deviation was just 1.2% of the mean.

4.2.11 Summary of conditions

Table 4.1 below summarizes the variability of the physical and chemical conditions that determine habitat suitability in the Ems-Dollard. Note that spatial differences over the vertical are not taken into account.

Table 4.1 Spatial and seasonal variability of representative conditions

Parameter	Spatial variability	Seasonal variability
Depth	Large	None
Inundation time	Large	Limited
Wave height	Large	Only maximum; average is constant
Orbital velocity	Large	Only maximum
Flow velocity	Large	Limited
Sediment concentration	Large	Limited
Secchi depth	Considerable / large	Moderately
Temperature	None	Large
Salinity	Considerable	Considerable
Oxygen	Large	Considerable
Bed composition	Large	None

4.3 Determination of Habitat Suitability Index per species

For every studied species, the habitat suitability index (HSI) is determined using the representative conditions discussed in the previous section (Section 4.2) and the response curves presented in the previous reports (Dijkstra et al., 2010, 2011). To aid the reader of the present report, the response curves and the corresponding set of representative conditions to be used are repeated here, but without discussing their construction again in detail.

For several species in the Ems-Dollard estuary, knowledge about their habitat requirements is readily available in the Habitat-database (http://public.deltares.nl/display/HBTDB). When response curves were not available or not sufficient, information from multiple sources was used to construct these curves: scientific papers, reports from institutes such as Imares and the former RIKZ and online resources (e.g. soortenbank.nl, fishbase.org). The report of Meesters et al. (2008) on an 'indicator system for biodiversity in Dutch marine waters' was especially useful.

As discussed during the construction of these response curves, food availability is not taken into account as a factor that can determine habitat suitability. The main reason for this exclusion is the lack of existing response curves with respect to food; most studies used the flow velocity or transparency as an indicator for how much food is can be available or can be caught, respectively.

The total habitat suitability index is defined as the minimum of the suitability indices of all individual parameters; the higher the total HSI, the higher the suitability seems. For discussion purposes, a HSI of 0 implies completely unsuitable, a HSI between 0 and 0.5 stands for moderately suitable, a HSI between 0.5 and 0.8 implies reasonably suitable and a HIS above 0.8 implies very suitable. Subsequently, the outcome is discussed and compared to available field data of 2001. This comparison is used to discuss the validity of the model and make suggestions for improvements where applicable. The findings mentioned under 'results' are the results using the original habitat suitability. To avoid confusion and too many figures, the map of these first results is not shown for all species; the map that is included for each species is the one after calibration. An exception was made for Eelgrass and the Blue mussel, because here the suitability depends on many parameters; a map of suitability per parameter was considered to be very informative.

4.3.1 Suitability for Eelgrass (ZM)

Representative conditions and response curves (Fig. 4.11)

Parameter	Layer	Season	Temporal	Remark
Salinity	AV	YR	AV	
Flow velocity	AV	YR	MX	
Orbital velocity	AV	YR	MX	Seems overestimated by model
Fines in bed	Bed	YR	AV	
Pct Surface Irradiance	n.a.	SM	AV	Only important in growing season
Emersion time	n.a.	YR	AV	Waddensea-specific response curve required

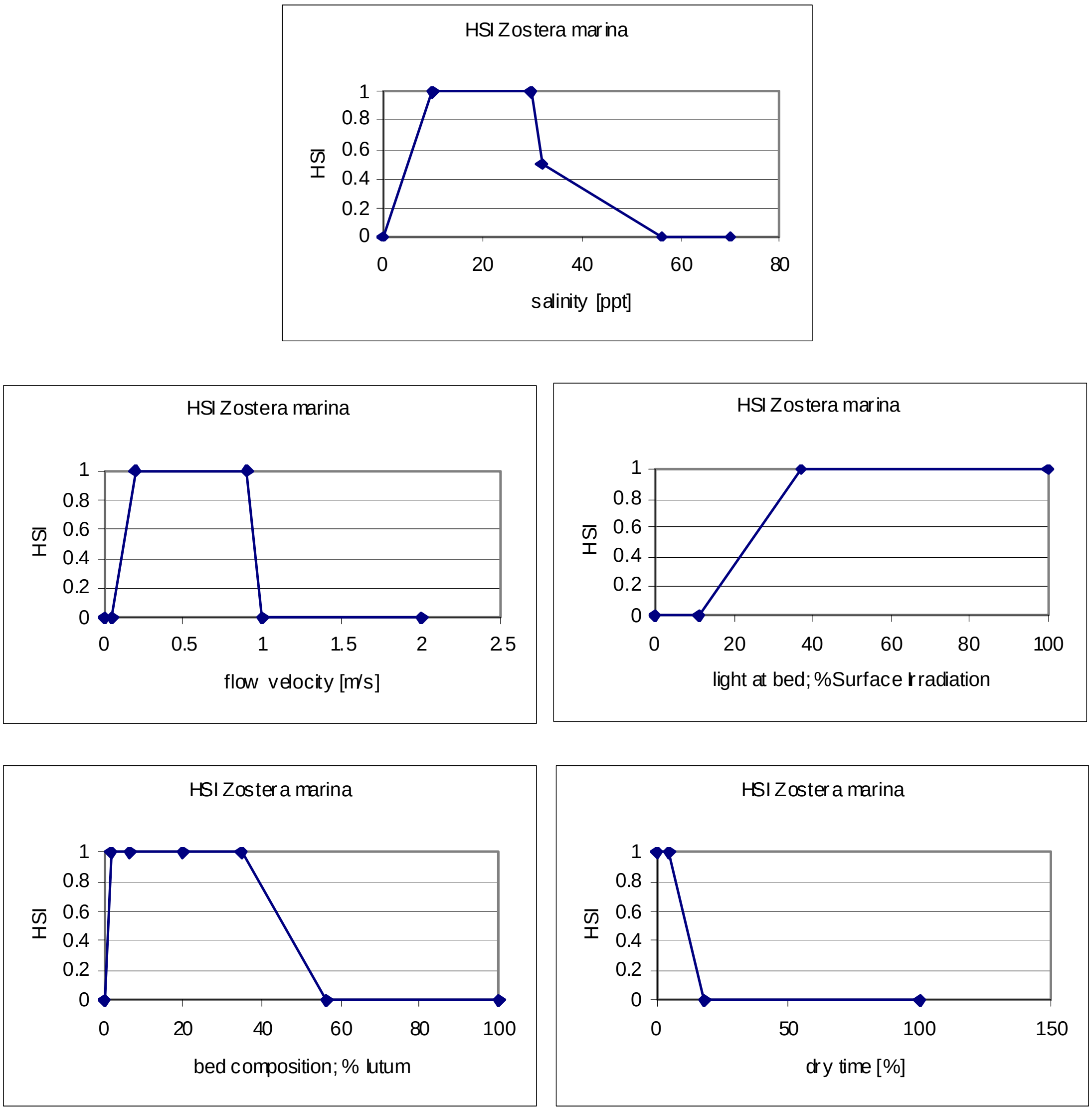


Fig 4.11 Response curves of Zostera marina (Eelgrass)

Results

The suitability per parameter is indicated in Figure 4.12. Salinity was not a limiting factor for eelgrass in Ems-Dollard, except in the immediate vicinity of the freshwater discharges near Delfzijl and Nieuwe Statenzijl. High flow velocities lead to unsuitable conditions in the main channels and lower suitability in some shallow areas, but for most shallow areas the flow velocity was not a limiting factor. Contrary to the unidirectional flow velocity, the orbital velocity was a limiting factor on the shallow parts (HSI=0), while the conditions in the channels were fine.

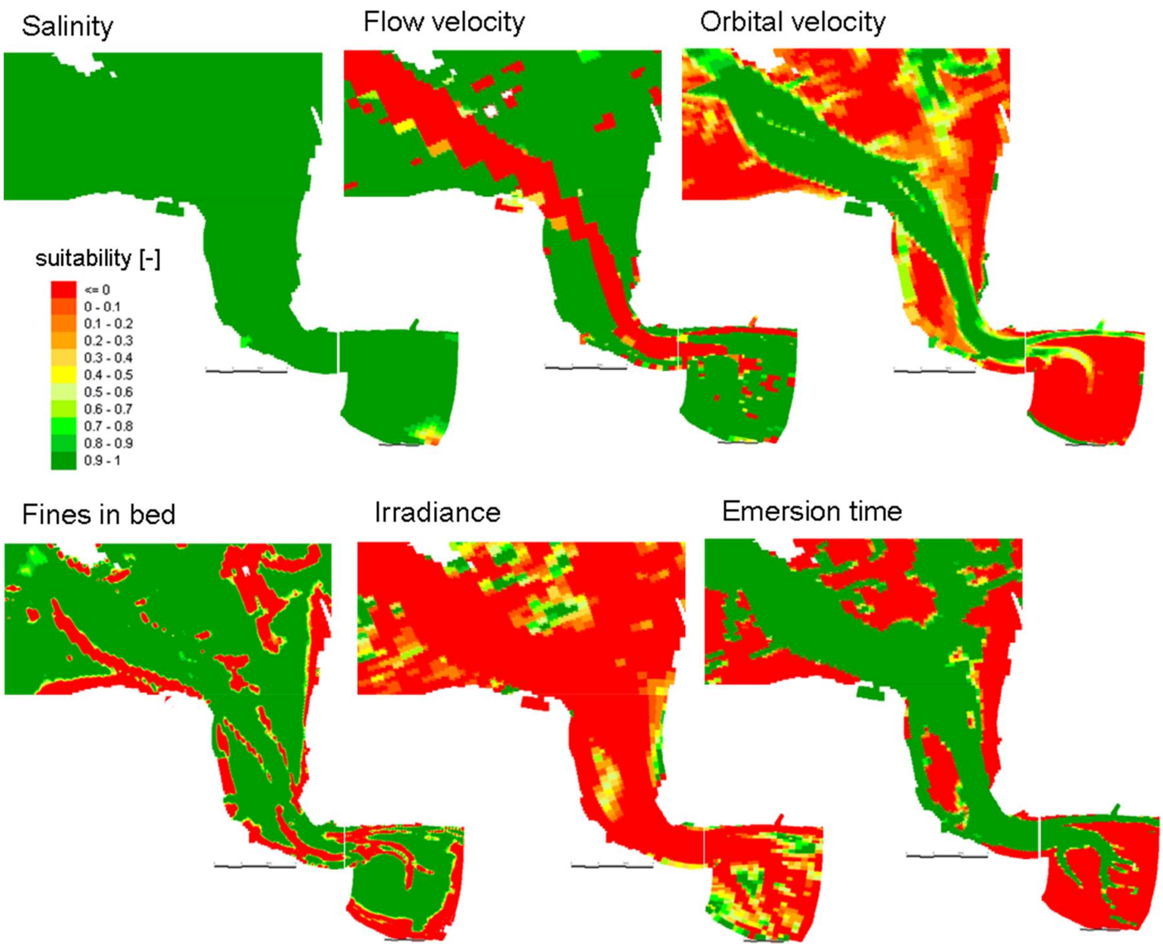


Figure 4.12. Habitat suitability for Zostera marina *per parameter.*

Most of the higher intertidal to supratidal areas near the dikes had a too high percentage of fines in the bed to be suitable, which also held for the more seaward part of the Uithuizer wad, Dukegat Plate and the area west of Greetsieler Nacken. In the outer and middle area, the bed composition in the channels would be suitable for eelgrass, whereas the borders of the channels and the Bocht van Watum were not. In the Dollard, the channels were too muddy but the intertidal area had an appropriate composition. As the percentage of irradiation that reaches the bed is a function of depth, inundation time and turbidity, all suitable areas were high and intertidal, but not all high areas were suitable. Only those directly behind the barrier islands, the Voolhok, near Rysumer Nacken, the middle of the Heringsplaat, the southwest corner of the Dollard and the area near the Geiseleitdam had high suitability values. The middle of the Hond-Paap had a suitability between 0.2 and 0.6; deeper areas were unsuitable. The short emersion time (<18%, preferably less than 10%) allowed by the 'worldwide' response curve rendered practically all intertidal areas unsuitable and the channels suitable.
Overall, the whole study area was unsuitable for eelgrass based on these criteria.

Discussion

Eelgrass did occur on the Hond-Paap in 2001 (Figure 4.14), whereas the model calculated that the entire Ems-Dollard was unsuitable for eelgrass. The meadow on the Hond-Paap was not an example of a thriving and healthy seagrass meadow, so conditions were likely not optimal but at least suitable for survival. In the model, two factors were limiting for eelgrass on Hond-Paap: the orbital velocity and the emersion time.
On the Hond-Paap, the maximum orbital velocity according to the model was just over 0.5 ms^{-1}, which is too high for eelgrass according to the response curves of De Jong et al. (2005) that limit the occurrence to orbital velocities below 0.4 ms^{-1}. Either the present model overestimates the orbital velocities, or the eelgrass is more resilient to wave motion than the applied response curves indicate. The first reason is most likely, as the modeled orbital velocities used by De Jong et al. (2005) on the Hond-Paap are lower and they more or less correctly predicted the suitability for eelgrass. De wave model used by De Jong et al. (2005; SWAN) is more realistic than the fetch length approach used in the present study.
The emersion time on Hond-Paap was between 16-47% in the model, whereas the response curve that was calibrated for the Venice lagoon (Erftemeijer and Van de Wolfshaar, 2006) had a maximum emersion time of 18%. Since eelgrass in the Waddensea has adapted to longer emersion times and depends on these dry and therefore light periods for photosynthesis (Ochieng and Erftemeijer subm.), the response curve of De Jong et al. (2005; Figure 4.13) with emersion times between 40-65% might be more realistic.

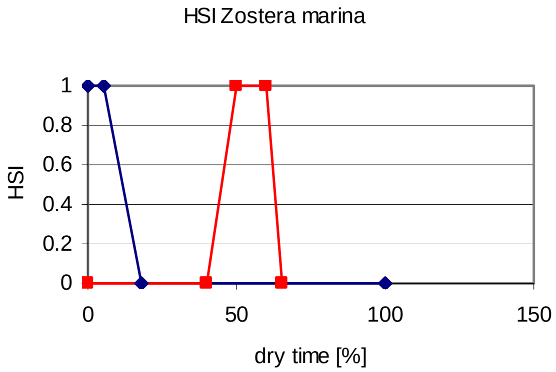


Figure 4.13. Originally used (dark blue line) and adapted, Waddensea-specific (red line) relation between emersion time and habitat suitability for eelgrass.

Figure 4.15 shows that if orbital velocities are not taken into account and the Waddensea-specific response curve for emersion time were to be used, a substantial part of the Hond-Paap would indeed be moderately to reasonably suitable for eelgrass. Some of the area around Rysumer Nacken and intertidal areas behind the barrier islands would have even better suitability, which is not the case in the study of De Jong et al. (2005) due to higher orbital velocities. These higher velocities seem realistic, as Rysumer Nacken is more exposed to (north)westerly winds than Hond-Paap. The intertidal flats in the Dollard were also identified as reasonably suitable, whereas the study of De Jong et al. (2005) identified them as unsuitable due to high ammonium concentrations in combination with relatively low salinity; ammonium was not taken into account in the present study.

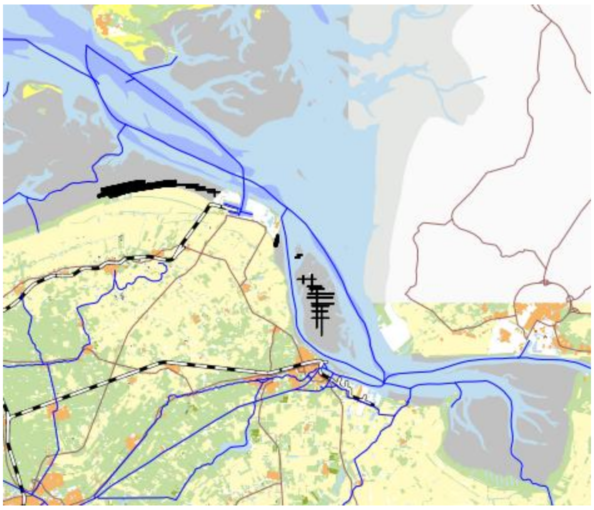


Figure 4.14. Black lines indicate approximate locations of Zostera marina on Hond-Paap and Zostera noltii at Uithuizerwad. The year is unknown, but in the Dutch part of this region Z. marina did not occur in meadows other than on Hond-Paap in recent decades. Source: Watlas 2011.

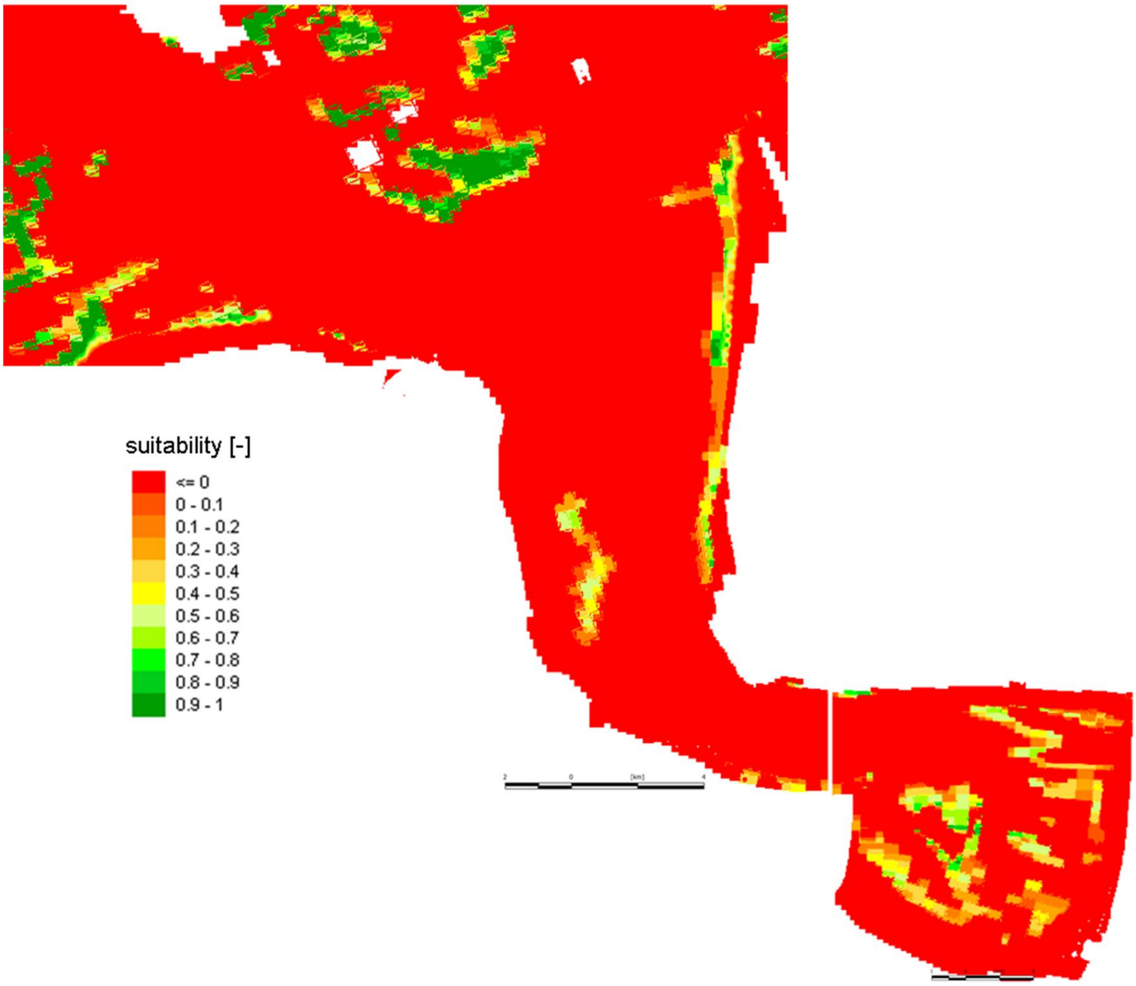


Figure 4.15. Habitat suitability for Zostera marina in 2001, based on salinity, flow velocity, bed composition, percentage surface irradiance and a Waddensea-specific relation for the emersion time. The orbital velocity (wave action) is not included as a criterion, as this would render the entire area unsuitable.

4.3.2 Suitability for salt marshes (H1330), pioneer zones (H1310) and *Spartina* swards (H1320)

Representative conditions and response curves (Fig. 4.16)

Parameter	Layer	Season	Temporal	Remark
Salinity	AV	YR	AV	
Emersion time	n.a.	YR	AV	
Dynamics	n.a	YR	MX	Assumed suitable if orbital motion < 0.25 ms^{-1}

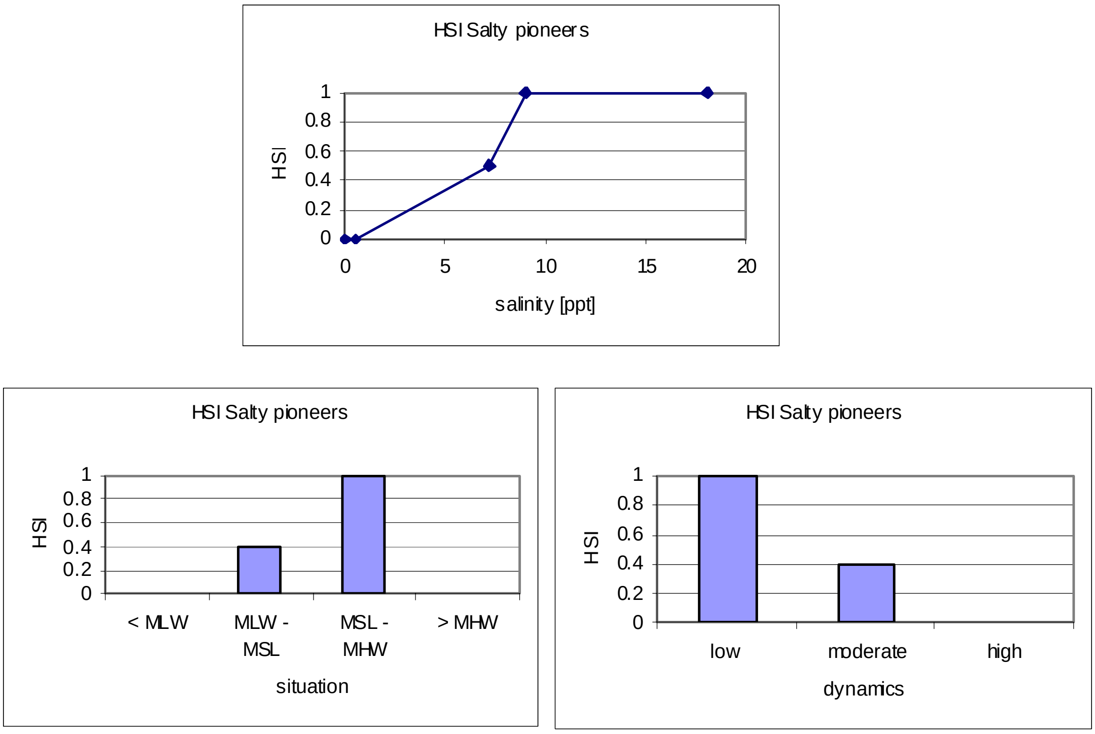


Figure 4.16 Response curves for H1301 Salty pioneers.

Spartina swards and salty pioneers were treated together in this study because the only difference in their habitat preferences is the unspecified parameter ‘Dynamics’; *Spartina* can cope with moderate dynamics, whereas other pioneers require low dynamics. As the quantification for dynamics is unknown, areas with orbital velocities lower than 0.25 ms^{-1} were considered suitable and areas with higher orbital velocities unsuitable.

Results

Salinity only decreased habitat suitability for salty plants close to the pumping stations near Delfzijl and Nieuw Statenzijl. The emersion time limited the occurrence of suitable plant habitats to the higher areas close to the dikes and the highest intertidal flats: Fringing the barrier islands, the Uithuizerwad, the strip from Greetsieler Nacken to Rysumer Nacken, the Voolhok and a small area just north of it, the Hond-Paap, near Termunterzijl, Knockster Watt and large parts of the Dollard, including most of the Heringplaat. The highest areas, i.e. the ones not flooded daily, were unsuitable for salty pioneers and *Spartina*, whilst only the upper intertidal areas were suitable for salt marshes. High orbital velocities (when only 80% of the originally calculated value was used) made only the deeper areas (channels) and the highest areas suitable.

Overall, moderately suitable areas for salt marshes were found at the back of Rottumeroog and Borkum, just west of the Eemshaven, near the intake/outfall constructions of the Eemscentrale, Pilsumer Watt, Rysumer Nacken, Knockster Watt, de Punt van Reide and most extensively at the landward margins of the Dollard (Figure 4.17). Much smaller moderately suitable areas for *Spartina* or other salty pioneers were found just a bit lower at these same locations (Figure 4.18). Only very small reasonably or very suitable areas were identified, for pioneers near the Eemscentrale and Rysumer Nacken.

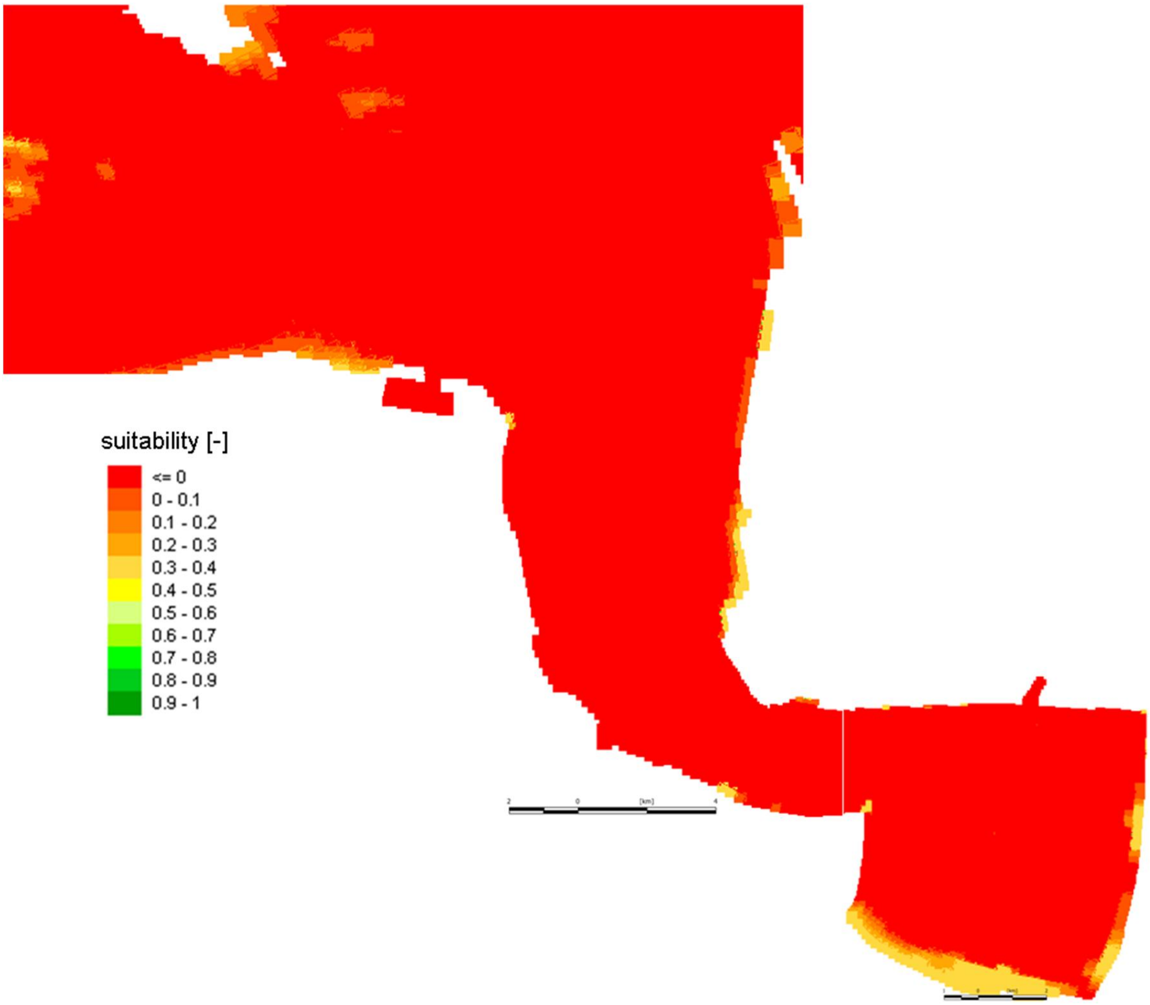


Figure 4.17. Habitat suitability for saltmarshes in 2001, based on salinity, emersion time and 'dynamics', i.e. orbital velocity.

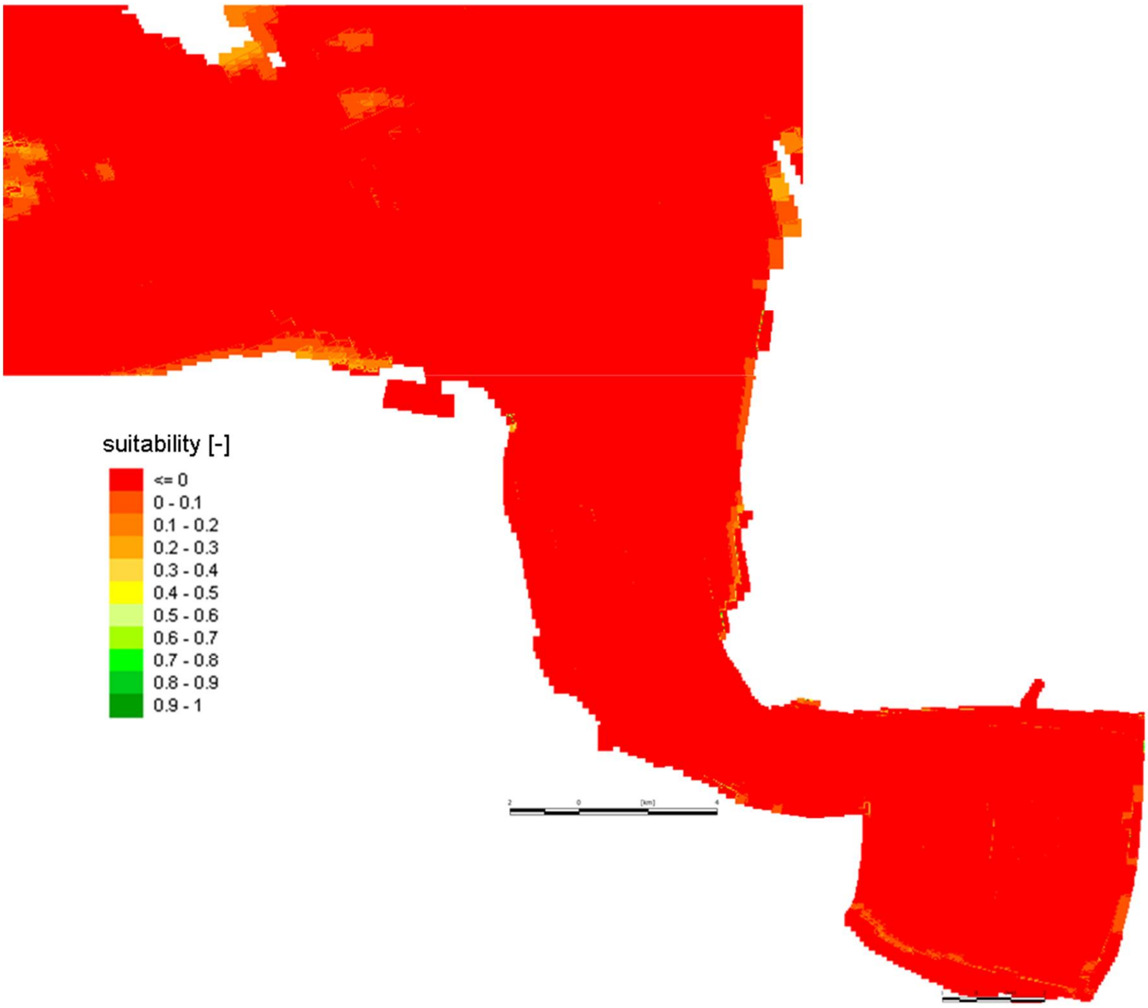


Figure 4.18. Habitat suitability for Spartina and salty pioneers in 2001, based on salinity, emersion time and 'dynamics', i.e. orbital velocity.

Discussion

The easiest way of verifying the actual presence of salt marshes is by means of GoogleEarth, although the years may not exactly match. The resemblance of the modeled habitat suitability to the actual presence is remarkably good: On all locations that are indicated as moderately suitable salt marshes were visible on the images, and all visible salt marshes were identified as moderately suitable. This indicates that the habitat suitability curves in combination with the modeled physical parameters are all right, though perhaps some fine-tuning with respect to the orbital velocity, which was now artificially lowered to 80%, is possible.
The presence of *Spartina* or other salty pioneers specifically cannot be verified easily via imagery, as these are difficult to distinguish from the salt marshes. Usually, pioneers and salt marshes occur together unless there is severe erosion, so this does give some confidence that the modeled locations are correct. The modeled extent of these pioneer zones is very small however.

4.3.3 Suitability for Blue mussel (ME)

Representative conditions and response curves (Fig. 4.19)

Parameter	Layer	Season	Temporal	Remark
Oxygen	AV	YR	2nd pctle	~effective minimum in summer
Salinity	AV	YR	AV	
Temperature	AV	YR	MX, MN	Spatially uniform; therefore not included
Depth	n.a.	YR	AV	
Wet time	n.a.	YR	AV	
Flow velocity	AV	YR	MX	
Orbital velocity	AV	YR	MX	
Suspended sediment	AV	YR	AV	Relations for short-term

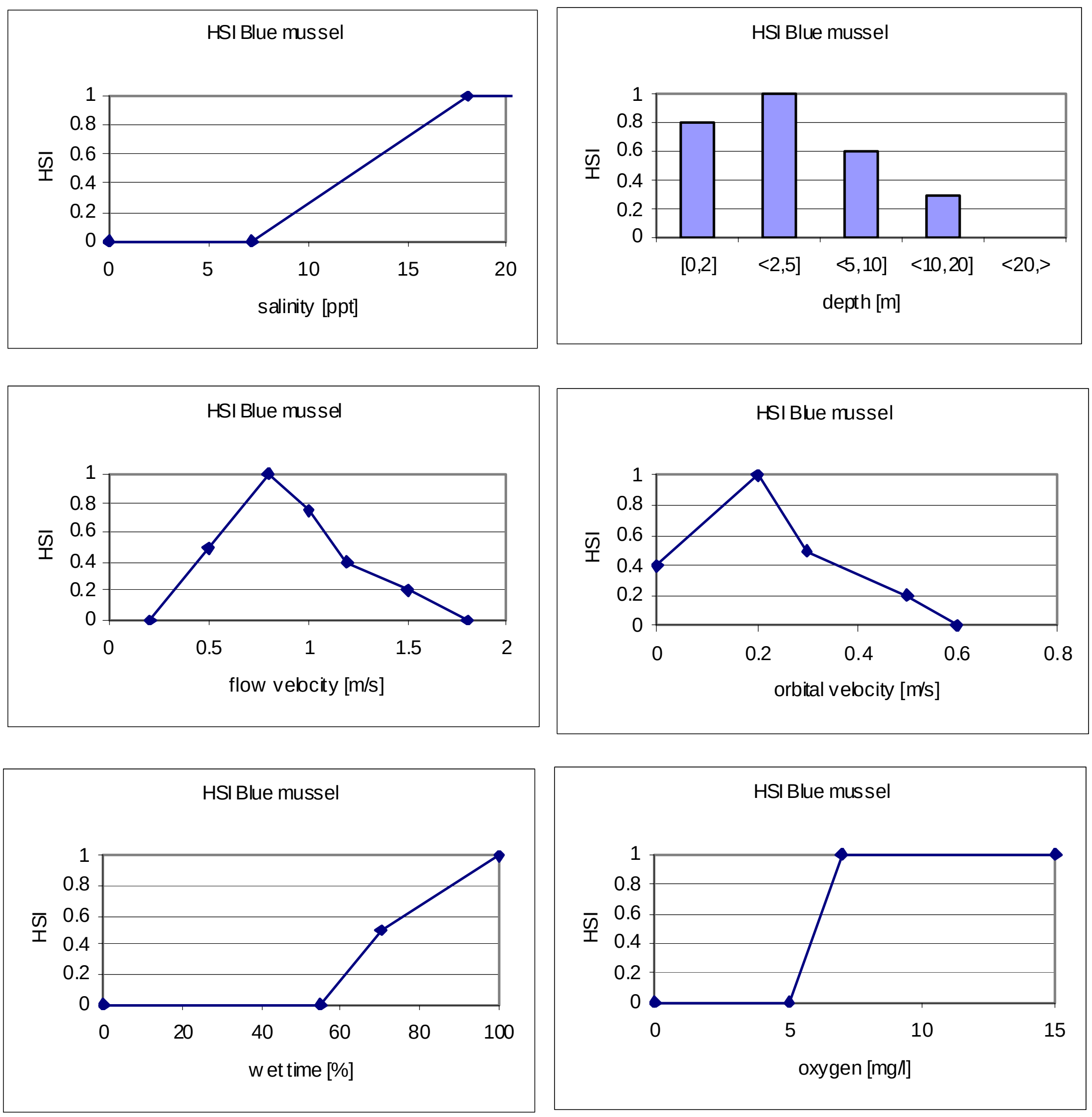


Figure 4.19. Response curves for Mytilus edulis (Blue mussel)

As discussed during the making of the response curves (see Dijkstra et al., 2010), the availability of food (phytoplankton) is not used as a parameter that determines habitat suitability for the Blue mussel. The main reason for this, is that the amount of food actually present in the estuary is considered to be more than sufficient to support mussels and other molluscs, especially since they have always occurred there. If food would become a limiting factor, the amount of molluscs likely decreases first, before areas become unsuitable. A second reason is that the feedback between food and predators, which determines the amount of available food, cannot be modeled using the static Habitat model, but asks for a different modeling approach.

Results

The habitat suitability per parameter is indicated in Figure 4.20, except for salinity, which was only limiting in a very small part of the area. The suitability index for the oxygen concentration showed high values on most lower intertidal and shallow subtidal areas. Some higher intertidal areas as well as the main channels were unsuitable to moderately suitable due to low oxygen concentrations, likely resulting from algal respiration. Salinity was not a restrictive factor for the occurrence of Blue mussels in Ems-Dollard, except for the freshwater outlets near Nieuw Statenzijl and Delfzijl.

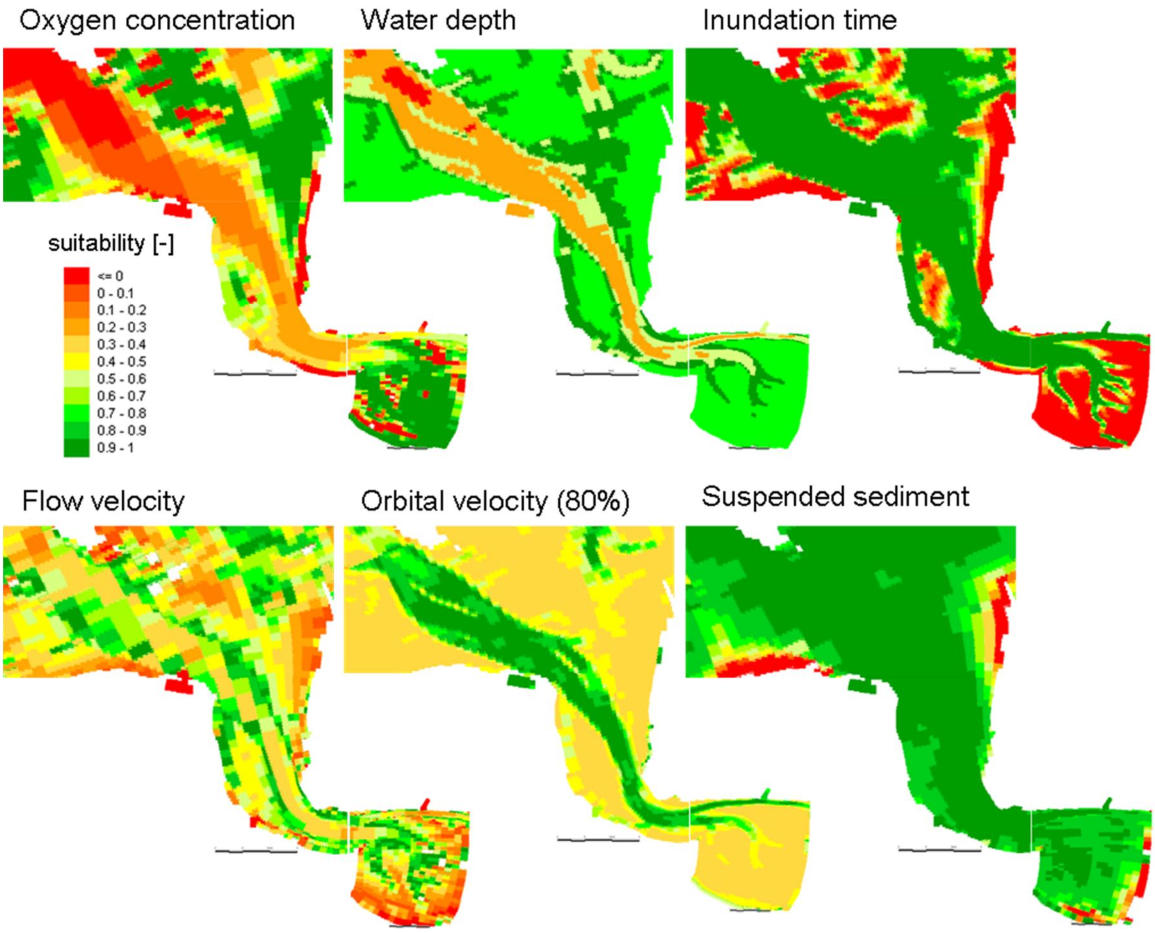


Figure 4.20. Habitat suitability for Mytilus edulis *per parameter. Instead of the originally calculated orbital velocity, which was considered unrealistically high in Section 4.3.1, a 20% lower value was used.*

Areas deeper than 20 m –the main channels near the North Sea- were unsuitable. Shallower channels (10-20m) were moderately suitable, with suitability increasing to very suitable as the area is shallower than 5 m. Intertidal areas such as large parts of the Dollard, behind the

barrier islands, along the dikes and Hond-Paap had a HSI of 0.8. The criterion for inundation time rendered many intertidal areas as unsuitable to moderately suitable, while the suitability in the channels was good. The improved criterion (see Discussion below) made the lower intertidal areas reasonably to very suitable, the channels reasonably suitable (HSI=0.6) and only the highest intertidal areas unsuitable. The demand for a substantial flow velocity (as an indication of possible food supply and refreshment) made most of the shallow areas moderately suitable, especially in the Dollard, whereas the channels were reasonably to very suitable. Most shallow intertidal areas (behind the barrier islands, Hond-Paap, the majority of the Dollard) were at best moderately suitable for mussels due to high orbital velocities. High suspended sediment concentrations decreased suitability only in the areas very close to the coastline, especially just in front of the salt marshes near Uithuizer Wad, Manslagter Nacken and in the Dollard; other areas were very suitable.
Overall, the suitability for Blue mussels given these criteria is moderate at best. The most suitable areas (HSI=0.4-0.5) are the channel side of the Emshorn Plate, the Hamburgersand, the northern part of the Bocht van Watum, and the Groote Gat in the Dollard.

Discussion
The modeled habitat suitability for *Mytilus edulis* differed substantially from the locations were mussels were encountered in the field: No mussels are reported in the Dollard, whereas they are present on Hond-Paap, Voolhok and on numerous locations in the intertidal areas behind the barrier islands (Fig. 4.21). In general, the model over predicted suitability in deep areas, and under predicted suitability in shallow areas. The most likely reasons for these differences, as well as suggestions for improvement are discussed below.

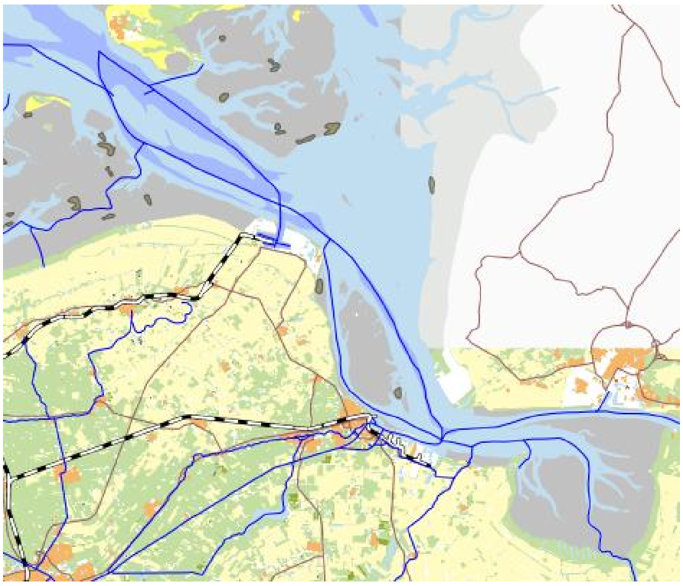


Figure 4.21. Natural mussel banks (ochre patches) in Ems-Dollard. Year unknown. Source: Watlas 2011

The response curve for the oxygen concentration has not been validated in earlier studies, but rather derived from relations for *Crassostrea gigas* and the *Dreissena polymorpha.* Although the long-term oxygen requirements of *Mytilus edulis* might be similar to these bivalves, neither of the studies in which these curves were derived concerned tidal waters, so the duration of low oxygen conditions likely differs: a long period just below 5 mgl^{-1} might be lethal, but several hours with 4 mgl^{-1} followed by several hours with 8 mgl^{-1} might be fine. These dynamics are difficult to capture with the present model setup, but the 2nd percentile oxygen concentration in combination with a slightly less strict response curve, seem to work well. Apart from that, the water quality model simulated very low oxygen concentrations in single cells on intertidal areas while the adjacent cells contained plenty of oxygen, which seems unrealistic.

Because mussels can stand extreme temperatures (-10 to + 40°C) for a while, and because the monthly averaged temperature from the water quality model is spatially uniform and between +1 and +20°C –inside the comfort limits of the Blue mussel-, temperature is not included in the calculation of habitat suitability. Perhaps using an upper and a lower percentile that represent extreme values that last for several days, just like the parameterization of oxygen conditions, would provide limiting conditions. In Ems-Dollard, moving ice might be the more damaging for mussel beds than extreme temperatures, however ice was not modeled.

The original criterion for inundation time may have been a bit too restrictive, as it excluded areas were mussels are known to occur. Therefore, the relation in Figure 4.22 (a) is proposed, which is more akin to the relations derived by Brinkman et al. (2002). Also, the depth at which mussels can occur might be overestimated: The depth itself is not a limiting factor in waters 5-20 m deep, but because this considers the main channels the sedimentation/erosion likely is substantial and will therefore hamper mussels. For this reason, the adapted depth-suitability relation in Figure 4.22 (b) was tested.

The calculated orbital velocities might be too high (see Discussion in Eelgrass section; 4.3.1), thereby limiting the suitability on shallow areas whilst overestimating the suitability in deeper water. For example, a reduction of the orbital velocity on Hond-Paap from the present 0.51 ms^{-1} to the more realistic 0.4 ms^{-1} in line with de values calculated by De Jong et al. (2005) would increase the HSI from 0.16 to 0.35. Therefore, as a test, the modelled orbital velocity was reduced to 80% of the original.

High sediment concentrations are known to hamper mussel feeding and consequently habitat suitability (Widdows et al., 1979; Essink et al., 1990). The filtering behaviour of mussels not only depends on the sediment concentration, but also on the sediment size/type. The experiments performed in these two studies were limited in assessment of mussel growth and in tested conditions, respectively. Widdows et al. (1979) report a maximum intake of useful food at 11 mgl^{-1}; at higher concentrations filtering requires more energy but does not produce more useful food. At 300 mgl^{-1} filtering would require more energy than it would yield, so mussels stop filtering. Essink et al. (1990) compared mussel growth at concentrations between 50 and 125 mgl^{-1} and found little difference, which lead to the conclusion that this highest concentration does not hamper feeding, but also mentioned that mussels growing in lower concentrations elsewhere grew faster. Therefore, the renewed curve in Figure 4.22 (c) was tested. With these adapted curves, Figure 4.23 was made.

The study of Brinkman et al. (2002), on which most HSI-curves are based, also identified the median grain size for all particles larger than 16 μm (M16) as a factor that determines habitat suitability for the Blue mussel. Values around 170 μm are optimal; very coarse sands or siltier beds are not preferred. This parameter is not computed by the morphological model, nor can it be derived from other parameters. Therefore it is not used in this habitat suitability study, even though it might be the limiting factor for mussels in the Bocht of Watum and in the channels of the Dollard, for which the present model seems to over predict the suitability.

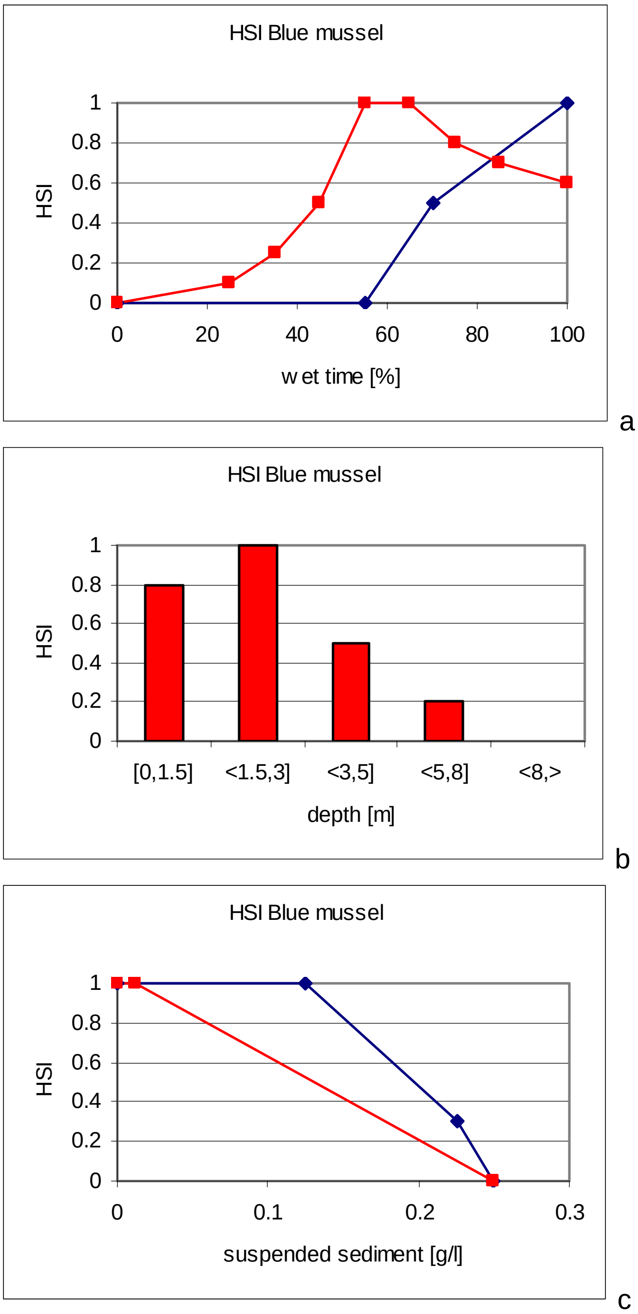


Figure 4.22. (a) Originally used (dark blue line) and improved (red line; according to Brinkman et al., 2002) relation for inundation time, (b) Adapted depth criterion and (c) Originally used (dark blue line) and renewed (red line) relation between suspended sediment concentration and habitat suitability for the Blue mussel.

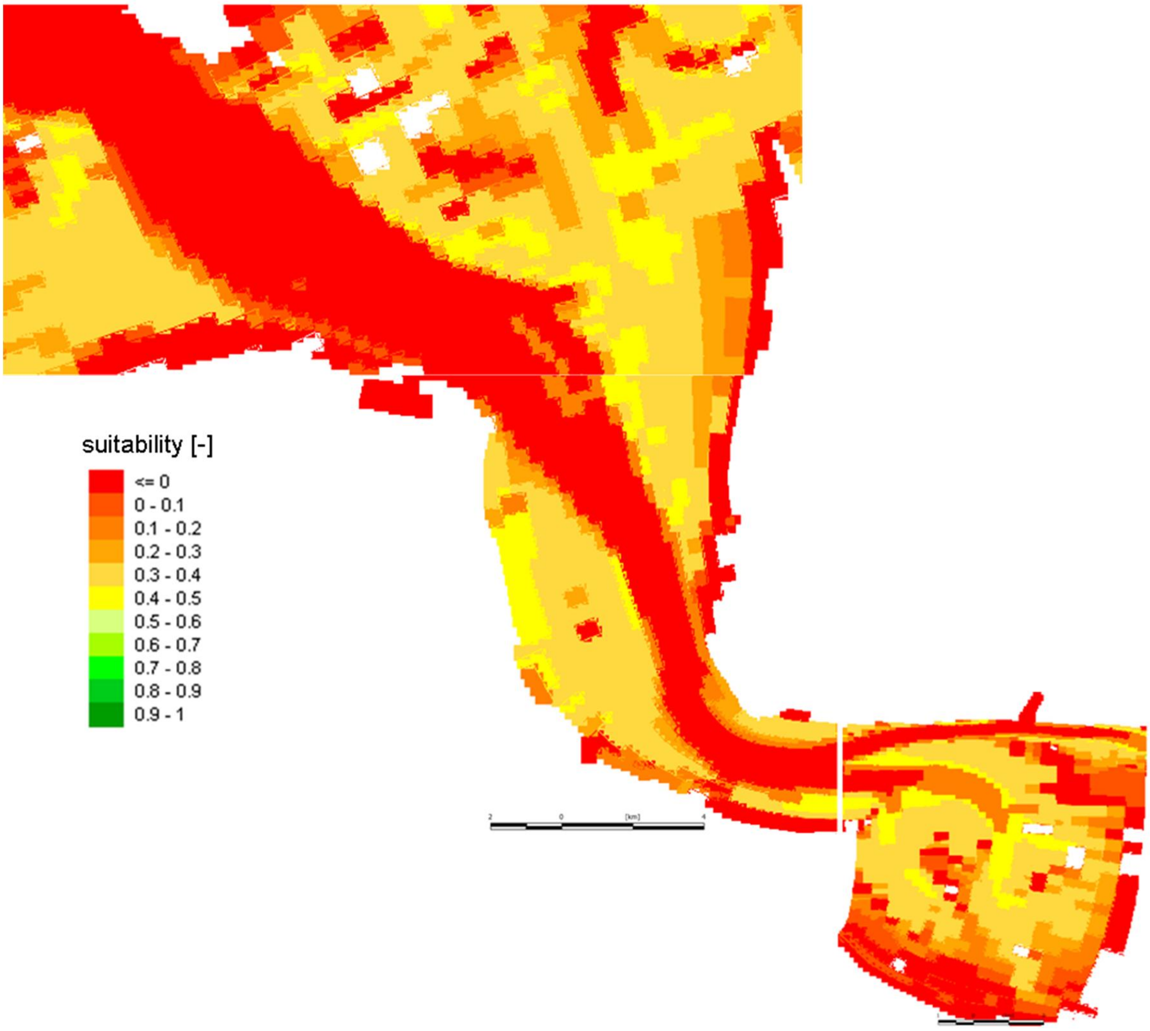


Figure 4.23. Suitability for the Blue mussel throughout 2001, using an improved response curve for emersion time, a reduced maximum occurrence depth, a relation between suitability and suspended sediment concentration and a 20% reduction of the orbital velocity

4.3.4 Suitability for Cockle (CE)

Representative conditions and response curves (Fig. 4.24)

Parameter	Layer	Season	Temporal	Remark
Oxygen	BL	YR	2nd pctle	2D, so depth averaged instead of bottom layer. 2nd Percentile ~ effective minimum in summer
Salinity	AV	YR	AV	
Temperature	*AV*	*YR*	*MN,MX*	*Not assessed: is spatially uniform and within tolerance limits*
Depth	n.a.	YR	AV	
Dry time	n.a.	YR	AV	
Grain size	*Bed*	*YR*	*AV*	*Not modelled*
Sediment concentration	AV	YR	AV	

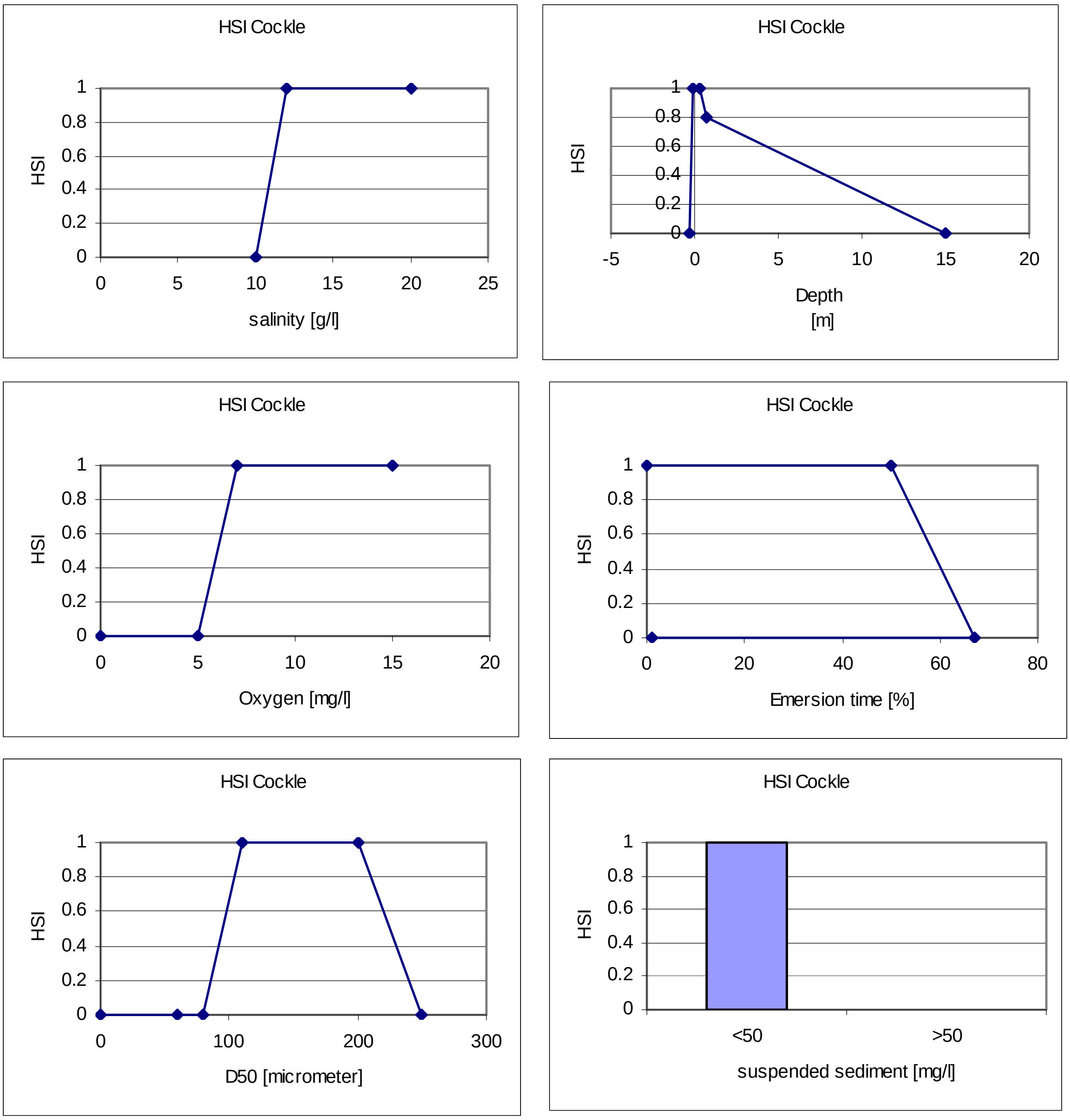


Figure 4.24. Response curves for Cerastoderma edule (Cockle)

For the same reasons as for the Blue mussel (Section 4.3.3), food supply is not considered to be a limiting factor for Cockles.

Results

In the deepest areas (i.e., the main channel) as well as some areas in the higher intertidal (Heringsplaat, Hond-Paap, Uithuizerveld, behind Borkum), low oxygen concentrations negatively affected the habitat suitability for Cockles. Salinity was not an inhibiting factor, except for the mouth of the Ems River, the area near Nieuw Statenzijl and a small area near Delfzijl. Temperature was not an issue for Cockles because the lowest modeled water temperature was 1°C throughout the study area; in reality frost might be an issue on intertidal areas in particular. High intertidal and very deep (>15m) areas were unsuitable for Cockles; most of the area was moderately to very suitable. Likewise, the high intertidal areas (Heringsplaat and fringes of the Dollard, Rysumer Nacken, strips close to the dikes in the

outer area) were rendered unsuitable due to the criterion for emersion time. The grain size was not assessed because this parameter was not modeled explicitly. The 'Discussion' treats a possible manner of including the role of the grain size and the consequences thereof. High suspended sediment concentrations (>500 mgl^{-1}) were barely limiting.
Overall, without accounting for bed composition, large parts of the study area were reasonably to very suitable for Cockles. Particularly the lower intertidal to shallow subtidal areas were suitable; higher intertidal areas were unsuitable and deeper channels at best moderately suitable.

Discussion
In the real Ems-Dollard, cockle presence was only reported on Voolhok, Uithuizerwad and behind Rottumeroog (Figure 4.25). Dekker and Waasdorp (2002) did not find any cockles in their samples from the Heringsplaat. The model results differed strongly from these field results. Cockle presence in Germany is unknown, so results cannot be validated.

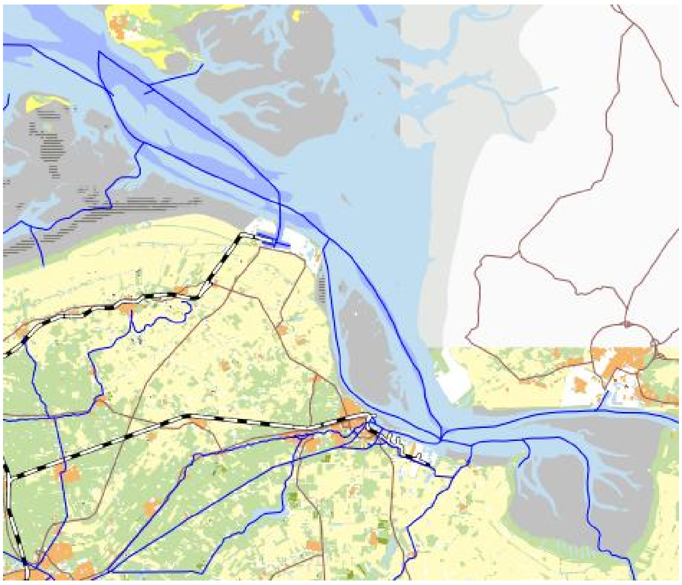


Figure 4.25. Cockle beds in the Dutch part of Ems-Dollard. Year unknown. Source: Watlas 2011.

Given the fact that the Cockle requires a sandy rather than a muddy substrate, the Dollard does not seem to be favourable area. Consequently, incorporating the bed composition (grain size) in the habitat modeling should give more realistic results. Because the sediment transport model does not have information concerning the grain size, the grain size needs to be estimated from the modeled percentage of fines in the bed. Assuming that the material in the fine fraction has a D_{50} of 20 μm, and that the coarse material (i.e., the rest) has a D_{50} of 200 μm, the representative D_{50} [μm] (Appendix B) can be calculated according to:

$$D_{50}=[20\times Percentage\_fines + 200\times(1-Percentage\_fines)]/100$$

Note that this is not a correct map of the exact D_{50}, as the sand in the Dollard likely is finer than that in the outer area, but it does roughly indicate where the bed consists mainly of fine material.
By adding the criterion for grain size to the overall habitat suitability assessment, parts of the Dollard and the Bocht van Watum become unsuitable (Figure 4.26) but the overall suitability seems still overestimated when compared to the –limited- monitoring data.

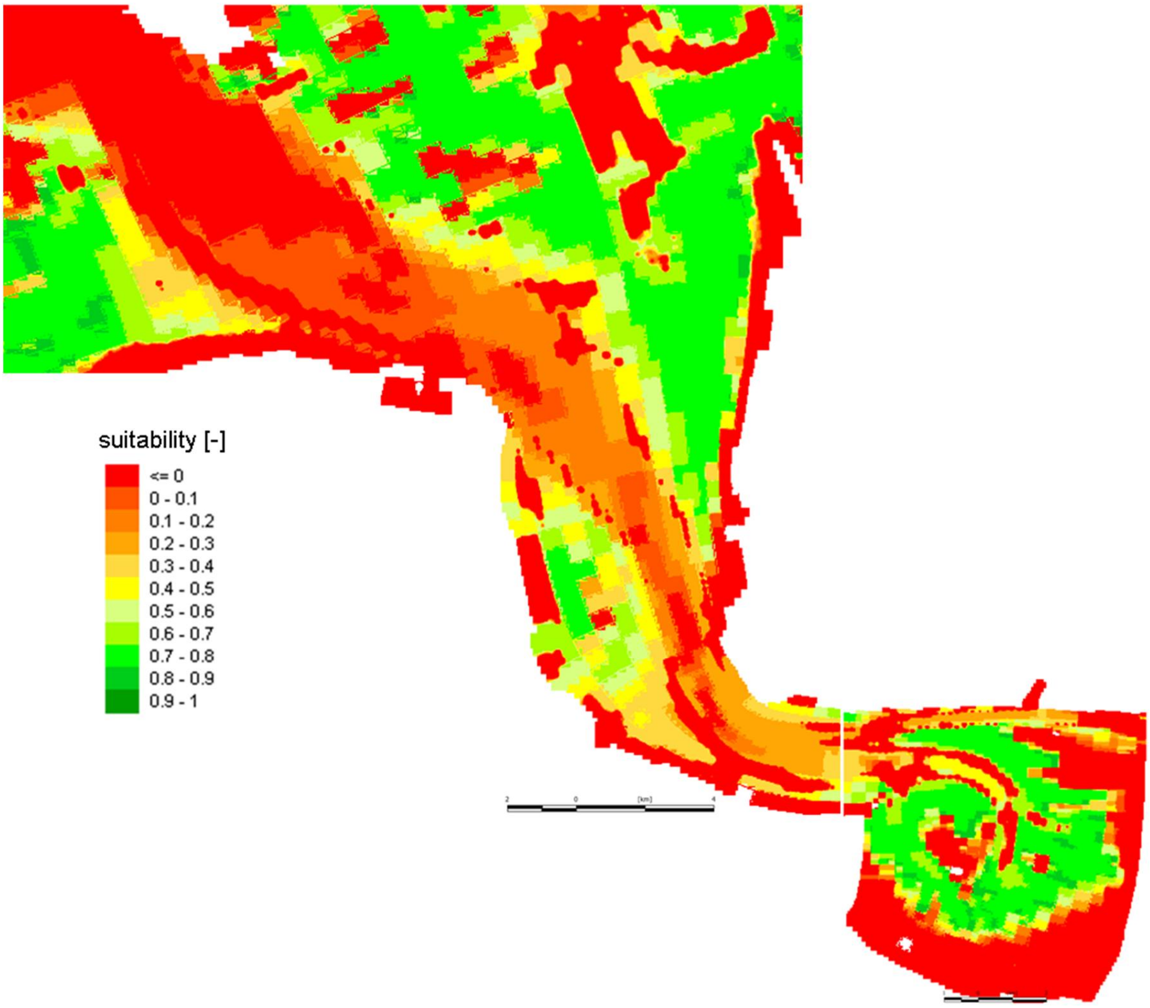


Figure 4.26. Suitability for the Cockle throughout 2001, using the 2nd percentile oxygen concentration, salinity, the depth, the emersion time, the suspended sediment concentration and an estimation of the grain size.

The criterion for suspended sediment concentrations (unsuitable if >500 mgl^{-1}; suitable if lower) does not seems very decisive, and compared to that for Blue mussels (optimal until 12 mgl^{-1}; acceptable until 225 mgl^{-1}) it seems too high. The flow velocity was not used as a criterion, but could be important as a proxy for the flux of food. As such, it could render most of the Dollard, but also the Voolhok and other intertidal areas, unsuitable. Apart from ice, rapid sedimentation and erosion, e.g. due to storms, can be a threat for cockles because they have a very limited ability to migrate vertically through the sediment.

4.3.5 Suitability for juvenile Herring (CH)

Representative conditions and response curves (Fig. 4.27)

Parameter	Layer	Season	Temporal	Remark
Oxygen	AV	SP	MN	
Turbidity	AV	SP	AV	
Temperature	AV	SP	AV	
Depth	n.a.	YR	AV	

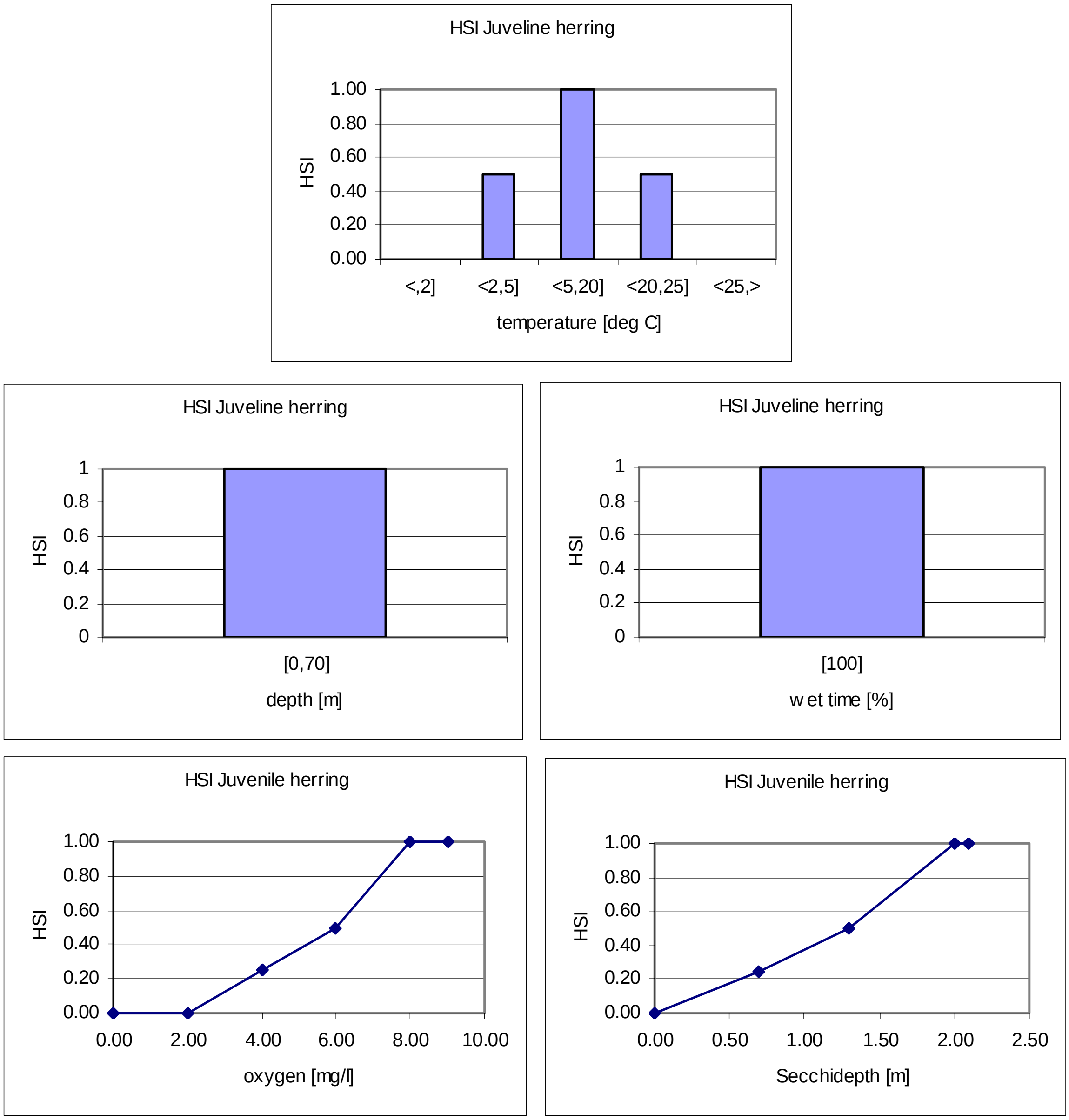


Figure 4.27. Response curves for Clupea harengus (juvenile herring)

As for the bivalves in the previous sections, the availability of food is not used as possibly limiting factor, but considered to be sufficient all the time. Also, the possible effect of predation is not taken into account in this study: Predation will limit the number of fish, but not the suitability of their habitat. Modelling predator-prey interactions requires a different type of model than the stationary Habitat model.

Results

In spring (April), the minimum oxygen concentration was not a limiting condition anywhere except in a few very isolated cells. In summer (July) however, even the time-averaged concentration would have affected the suitability for herring in the deeper channels, from Oostfriese Gaatje seawards.

The turbidity affected the suitability for the herring, which needs to see the zooplankton it feeds on, the most: In the deeper parts in the outer area, suitability was reasonable (HSI=0.5-0.6), whereas the intertidal areas behind the islands were moderately suitable (0.2-0.3) to

practically unsuitable (HSI<0.05) close to the coastline. In the Ems estuary, all areas were moderately suitable but with a clear spatial gradient, from 0.45 on the seaward side to 0.05 on the Dollard side. The turbid Dollard itself was practically unsuitable.
As the average temperature in spring was a spatially uniform 7.5°C, the entire Ems-Dollard was very suitable (HSI=1). In summer, hence likewise later in spring, the average temperature was 19.6°C, which would render the entire area unsuitable (HSI=0). The effect of depth on suitability was simple: all areas deeper than 2 m were very suitable, all shallower (intertidal) areas were reasonably suitable (HSI=0.5).
Overall, the deeper and clearer parts in the outer area were reasonably suitable in spring (April; Figure 4.28); the deeper into the estuary, the water was more turbid and therefore less suitable. The Eemshaven was remarkably suitable according to the (limited) criteria applied here. In summer, when the water would become warmer and oxygen concentrations would decrease, the entire study area would become unsuitable.

Discussion
As there is hardly any data on the spatial distribution of *Clupea harengus*, a thorough comparison and calibration with field data is impossible. The study of Hadderingh & Jager (2002) did report juvenile clupeids (herring or sprat, which species exactly was impossible to identify) in spring in the Doekegat, their only study location, in 1993 and 1997. According to the model, this is a reasonably suitable area, so that is a promising sign.
The recorded presence of birds that extensively feed on juvenile herring might be used as a proxy for the presence of herring. A substantial colony of Common terns is present near Delfzijl; smaller colonies existed on Rottumeroog, near Eemshaven and near the Punt van Reide and Polder Breebaart (Willems et al. 2005). Rottumeroog and Eemshaven are close to moderately to very suitable areas for herring, whereas the latter two are near apparently unsuitable sites (HSI<0.1). The south side of the Hond-Paap and Gaatjebocht, as well as some small areas near Delfzijl harbour were moderately suitable for herring, which might be enough for these birds.
One has to be careful with such indirect indicators however: The fact that De Boer et al. (2002) observed that the highest number of Terns occurred in July and August, when conditions are unsuitable for juvenile herring, and the herring itself might be a bit bigger than the optimal prey size, indicates that other factors (other prey, but also environmental conditions and behaviour) play a role.

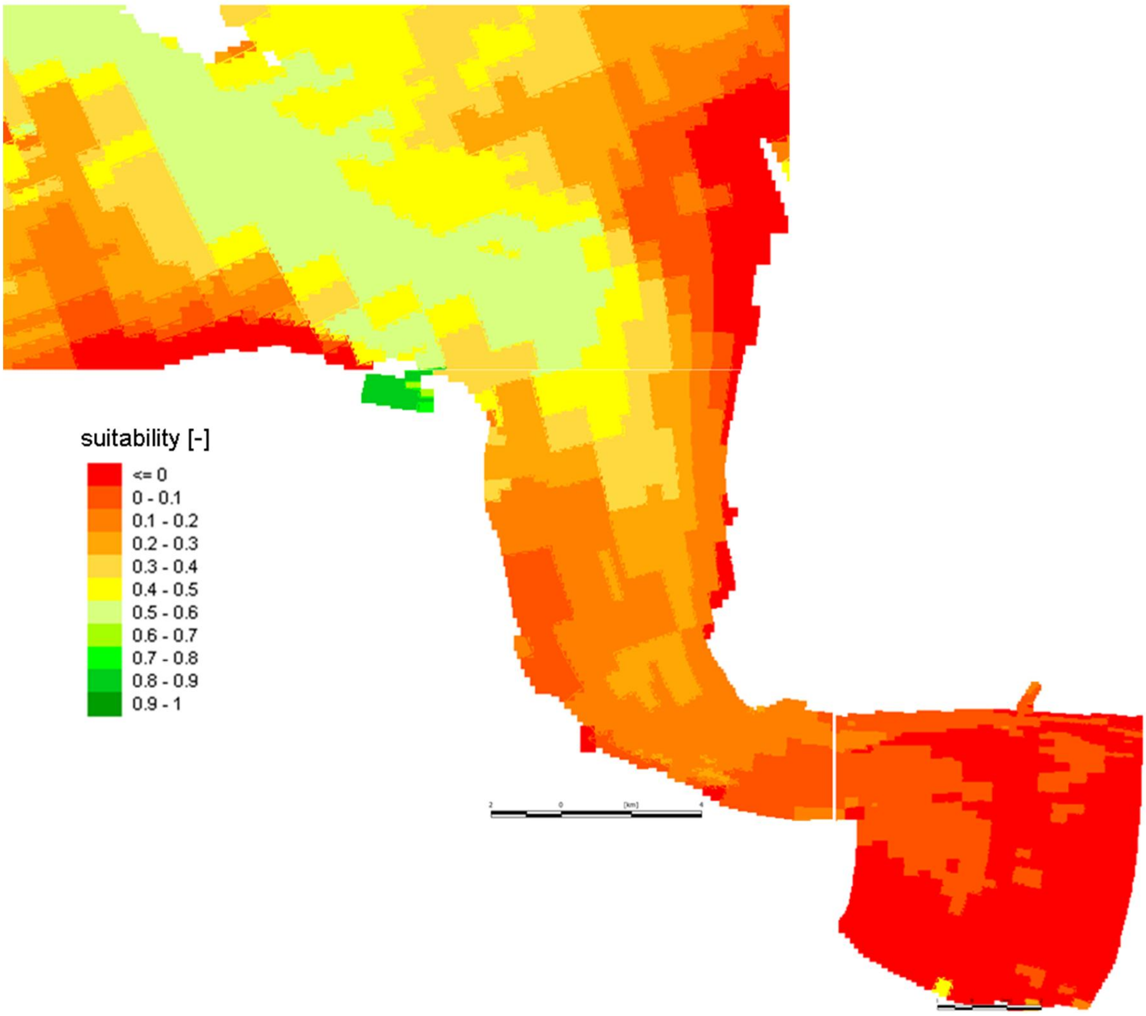


Figure 4.28 Suitability for juvenile Herring in spring 2001, based on oxygen, temperature, turbidity and depth.

4.3.6 Suitability for Common tern (SH)

Representative conditions and response curves (Fig. 4.29)

Parameter	Layer	Season	Temporal	Remark
Depth	n.a.	YR	AV	
Turbidity	SL	SP	AV	2D model, so turbidity is depth-averaged
Flow velocity	AV	YR	AV	
Colony distance	*n.a.*	*n.a.*		*Requires locations of colonies*

In Ems-Dollard, the number of Common terns is highest in spring. Therefore, the turbidity in this season is used; the time-averaged flow velocity and depth are practically constant throughout the year.

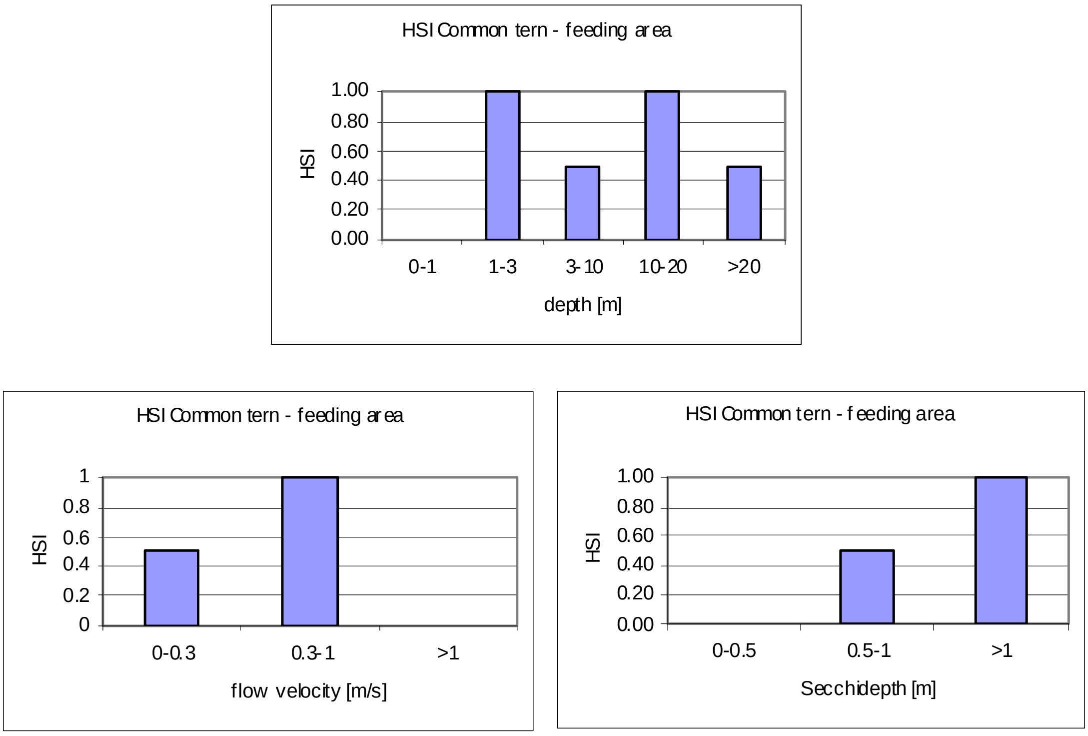


Figure 4.29. Response curves for Sterna hirundo (Common tern)

Results

Very shallow areas (<1 m) are not suitable for the Common Tern, which disqualified most of the Dollard, the intertidal flats in the outer area, the Hond-Paap and Rysumer Nacken. The channels and shallow areas (depth 1-3 m) were moderately to very suitable with respect to depth. Turbidity was an inhibiting factor in the entire Dollard, the Ems estuary and on the shallow parts of the outer area. The main channel in the outer area was moderately suitable. With respect to flow velocity, the entire study area had a HSI of at least 0.5, as the threshold of 1 ms^{-1} was not exceeded. The deeper (therefore carrying faster flow) parts of the channels reasonably suitable, whereas the some sides of the channels were very suitable.
Overall, only the channels in the outer area and the Eemshaven were moderately suitable; the rest was unsuitable.

Discussion

In reality, a substantial colony of Common terns was present near Delfzijl harbour, with smaller colonies on Rottumeroog, near the Eemshaven, the Punt van Reide and Polder Breebaart (Willems et al. 2005). According to the habitat model, only the ones on Rottumeroog and near Eemshaven would be near suitable feeding waters. This cannot be true, so either the conditions in the estuary are not modeled right or the used suitability curves are incorrect.
The principal limiting factor in the Ems estuary and the Dollard is the turbidity. Although turbidity seems a logical factor to include for a visually hunting bird, Schwemmer *et al*. (2009) did link the presence of terns just to water depth and flow velocity. If turbidity is not used as a criterion, the overall suitability for Common tern feeding areas increases considerably (Fig. 4.30): Parts of the main channels become reasonably to very suitable, even up in the Dollard. The area near the large colony at Delfzijl is moderately to very suitable; likewise for other locations were the presence of Common terns has been reported. Therefore, using only flow velocity and depth as habitat suitability indicators seems a viable option.

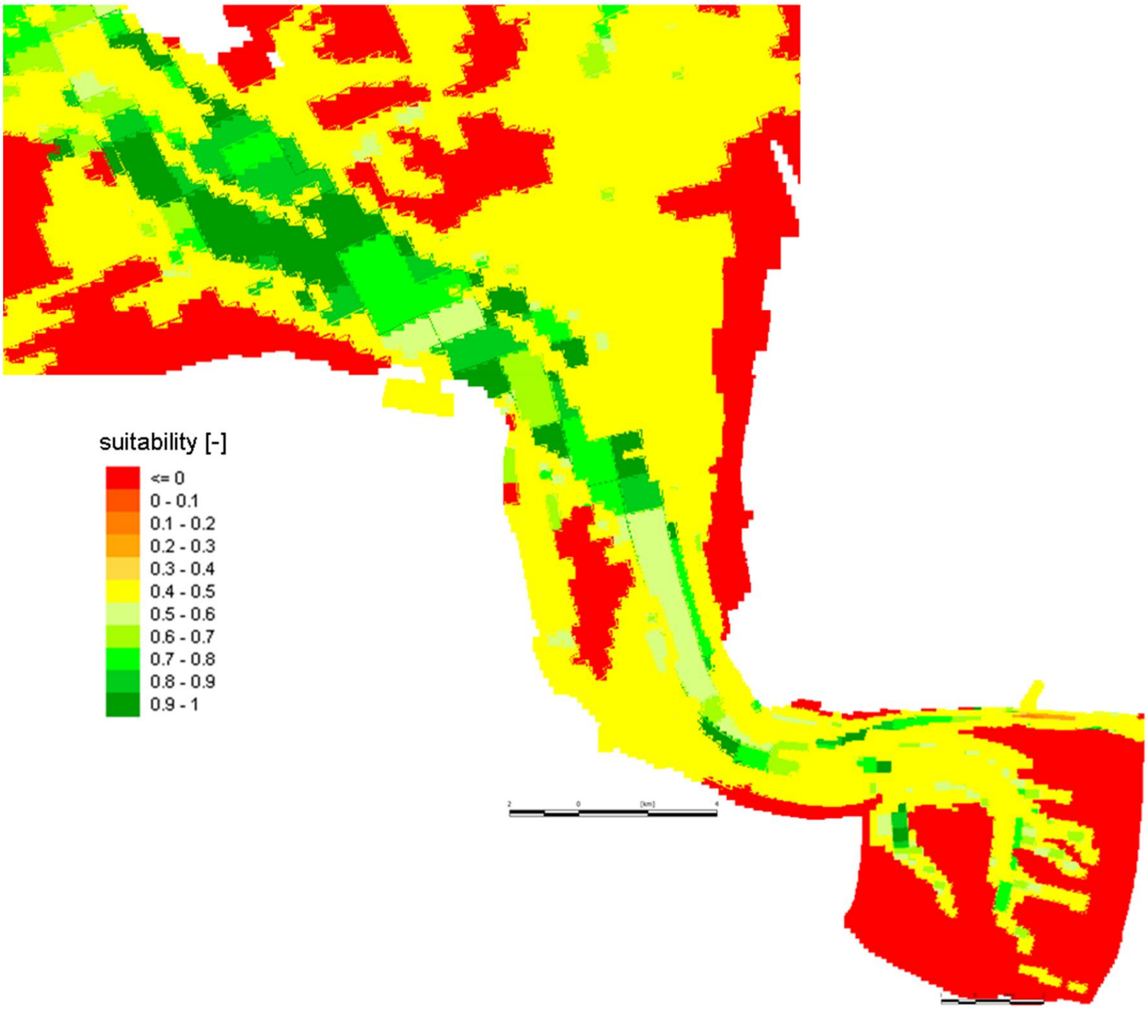


Figure 4.30. Habitat suitability for Common tern in 2001, based on depth and flow velocity.

4.3.7 Suitability for mammals

The mammals that possibly occur in Ems-Dollard are not discussed in this report: Their habitat suitability depends on physical factors (depth, dry areas for resting, temperature) that can be considered as suitable in Ems-Dollard without modeling them. Their actual presence however is determined by the presence of food, absence of disturbance and their behavior, which cannot be modeled well.

4.4 Historical scenario: Juvenile Herring

This section discusses the differences in habitat suitability between the 2001-scenario and the historical scenario for juvenile herring. The main difference between these scenarios is the suspended sediment concentration (see Section 3.3.1 for more details on the paramerisation of both scenarios). Historically, the sediment concentration was lower (Figs. 4.6 and 4.7) in the entire area, but with a spatial gradient. The sediment concentration was substantially lower in the Dollard, but only slightly lower in the Outer area. As a consequence, historical turbidity was also lower, and there was a small effect on oxygen concentrations. As a further consequence, the primary production in the historical scenario differed from 2001:

algae started to grow earlier but their growth became limited by nutrients, and their growing season lasted longer (Section 3.3.5). This would imply that food for fish was present for a longer period, but this factor is not taken into account.
Juvenile herring was used for this example mainly because its habitat suitability depended strongly on turbidity (Section 4.3.5), so changes in turbidity should clearly reflect in a change in habitat suitability. Moreover, herring was not restricted to a limited part of the study area by an other parameter, except for the shallowest areas. Furthermore, juvenile herring is an interesting species because it is an important prey for other fish and birds, which may also thrive better if more herring is present.
Figure 4.31 indicates the habitat suitability for juvenile herring in the historical conditions. Similar to the 2001-scenario, the deep parts of the outer area were the most (HSI=0.5-0.6; reasonably) suitable if the Eemshaven is not considered. But in the historical situation, this reasonable suitability extended well into the middle reaches of the Ems estuary, which was moderately suitable according to the 2001 model results. Most of the Dollard was moderately suitable in historical conditions, but unsuitable in 2001.
The difference in suitable habitat area between the two scenarios was apparent (Fig. 4.32): From the historical conditions to 2001, the amount of unsuitable area for juvenile herring more than doubled. The sum of moderately suitable areas decreased 6.5%, the amount of reasonably suitable areas decreased 32% and the area indicated as very suitable declined by 9%.

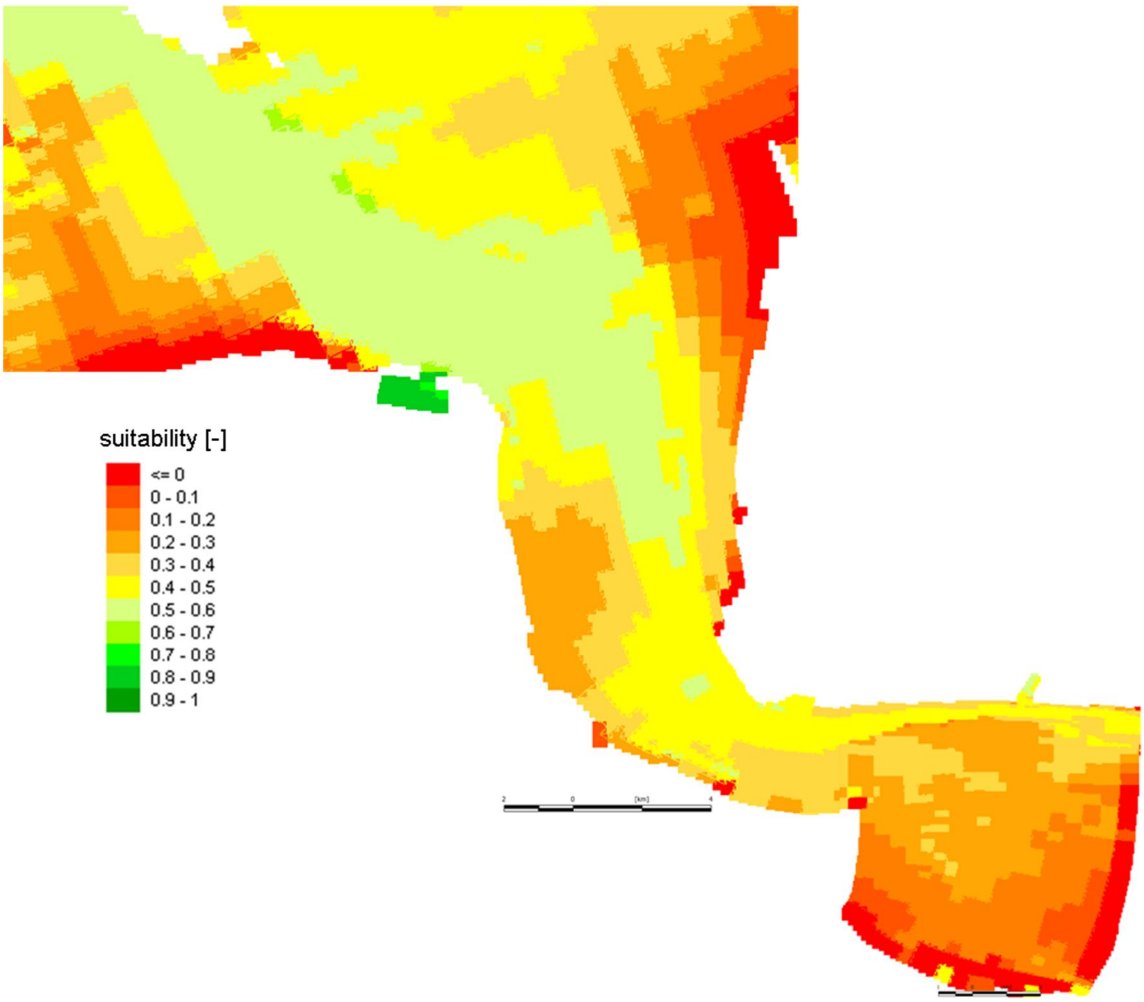


Figure 4.31. Habitat suitability for Juvenile herring in historical conditions.

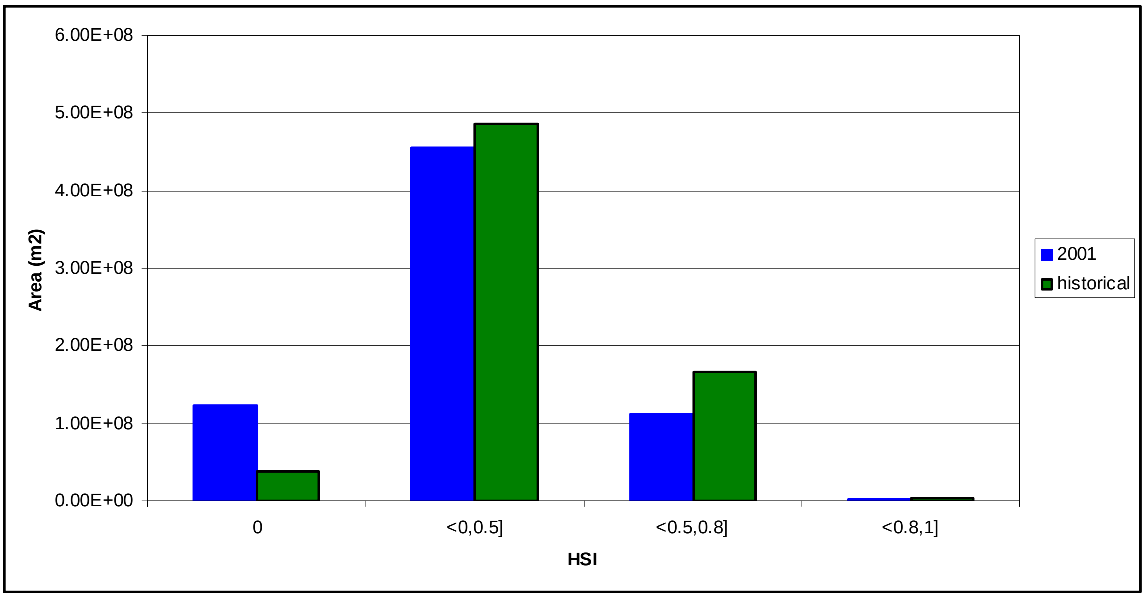


Figure 4.32. Difference in habitat suitability for Juvenile herring between historical conditions with reduced turbidity (green) and more turbid 2001 conditions (blue).

4.5 Discussion

Given the limited resemblance of most modeled habitat suitability maps to the available field data, it cannot be concluded that the model in its present state performs well. For several species however, the habitat model did indicate the suitability correctly, and in most cases the likely source(s) of the differences could be indicated. These differences can be categorized into unknown or insufficiently calibrated habitat suitability relations and missing or incorrectly modeled environmental parameters.

The overestimated orbital velocity is the clearest example of an incorrectly modelled physical parameter. In this study, the maximum orbital velocity was not calculated by a wave model but derived from the statistics of depth and wave properties that were the result of a simple fetch length approach. As a consequence of using these statistical values (maxima), this post-hoc maximum orbital velocity may not have occurred at all because not all the determining parameters arose at the same time. This issue could be resolved by applying a real wave model such as SWAN, which would also improve the directional properties of waves. For a limited number of typical or extreme conditions this could be done relatively easy, and is therefore recommended for a similar or follow-up study.

Another issue is the duration and severity of anoxic or hypoxic conditions, which applies both to modeled parameters and to unknown suitability relations: The minimum oxygen concentration in some cells was zero or even negative due to the applied model time step. If this happens only once, and only for several minutes, it is not meaningful for habitat suitability but it does show up in the results. Using the 2nd percentile (the concentration that was undercut 2% of the modeled time, i.e. 7.2 days in total) partially overcame this problem, but then other unknowns arise: Is 2% more realistic than 1%? Was this 2% a continuous period, or was it made of many but very short intervals of hypoxia? And which of these is the most harmful to the species of interest?

The use of a two-dimensional instead of a three-dimensional water quality model meant that only depth-averaged values could be used. For most parameters this seemed not problematic as the estuary is considered to be usually well-mixed. A comparison of the two-dimensional water quality with a later three-dimensional version (Section 3.3) showed that the oxygen concentrations in both versions were similar, and neither model calculated time-averaged concentrations below the usually critical 4 mgl^{-1} where the other did not.

A central shortcoming of the morphodynamic and water quality models is that they do not incorporate small-scale features that can be very relevant for ecology. For example, the presence of a mussel bed or small levee on an intertidal flat can attenuate waves and thereby make an area suitable for the establishment of other organisms, or determine how quickly and area is drained. Such features and their effects are typically not included in the other models, either due to the grid size used, the effort involved in including them or the fact that their existence is unknown. However, this is also an example of the fundamental difference in scale between the large-scale hydrodynamics and water quality throughout the estuary, and the (sometimes) small-scale demands of organisms: Increasing the accuracy of the habitat model in this respect would require a disproportional effort in hydrodynamic modeling. Using expert judgment to fill these gaps seems a more efficient method.

Combining the static Habitat model with models for the dynamic feedback between available food (i.e. plankton) and feeders (i.e. mussels, cockles, herring, etc.) would overcome the weakness of not including food availability as a factor for habitat suitability in this study. Thereby, such an extension would strengthen the interaction between the links of the effect chain. This hybrid modeling approach requires substantial development and validation data though.

The quality of habitat suitability curves might be further improved by more extensive literature study or further calibration, but probably the best way of getting better relations is to derive them from local or nearby field data via a GIS, complemented with knowledge of (local) experts. Using augmented reality, a technique that shows the output of the model directly in the field on a mobile device, would be an interesting practice for calibration and validation. This study started with the ambition of modeling habitat suitability for many species, but given the effort involved and the time available, only a few suitability curves could be calibrated to some extent.

5 Conclusions

The effect chain model for the Ems-Dollard is operational and has been applied to several scenarios. For 2012, a historical scenario with reduced suspended sediment levels was compared with the present situation. For the historical scenario, a significantly higher primary production is computed than for the present situation.

The current model for water quality and phytoplankton growth is performing technically well and includes both phytoplankton and phytobenthos. It uses a full-year hydrodynamics and modelled suspended sediment concentrations. There is a 2D-version available for fast exploration of a large number of scenarios, and a 3D-version which is performing better in terms of accurately describing phytoplankton biomass. Also, the 3D model predicts the timing of the spring bloom better than the 2D model. During the vertical aggregation, also the suspended sediment fields from the sediment model need to be aggregated vertically. In the current model, this is done by averaging concentrations over all layers. Possibly, a better result with the the 2D model is obtained when only suspended sediment concentrations from the top layer are used, which should be lower than the average and result in earlier spring bloom.

With regard to habitat suitability, the habitat model in its present state does not yet perform satisfactorily. Most of the modeled habitat suitability maps show a limited resemblance to available field data. For several species however, the habitat model did indicate the suitability correctly, and in most cases the likely source(s) of the differences could be indicated. These differences can be categorized into unknown or insufficiently calibrated habitat suitability relations and missing or too inaccurately modeled environmental parameters.

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- 2010: Dijkstra, J., T. van Kessel, B. van Maren, C. Spiteri, W. Stolte (2011). Setup of an effect-chain model for the Eems-Dollard. Results 2010. Deltares report 1202298.
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B Conditions determining habitat suitability

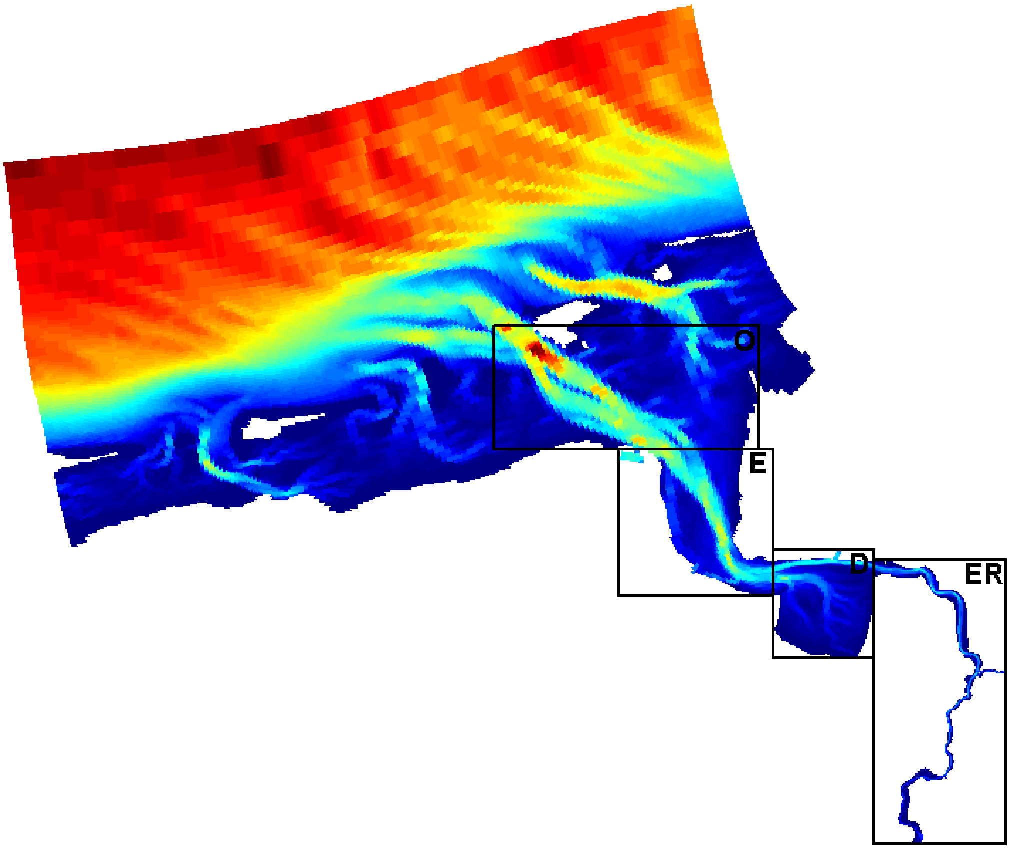

Figure B.0 Division of the Ems-Dollard study area into four subbasins: Outer area (O), Ems estuary (E), Dollard (D) and Ems River (ER; not studied).

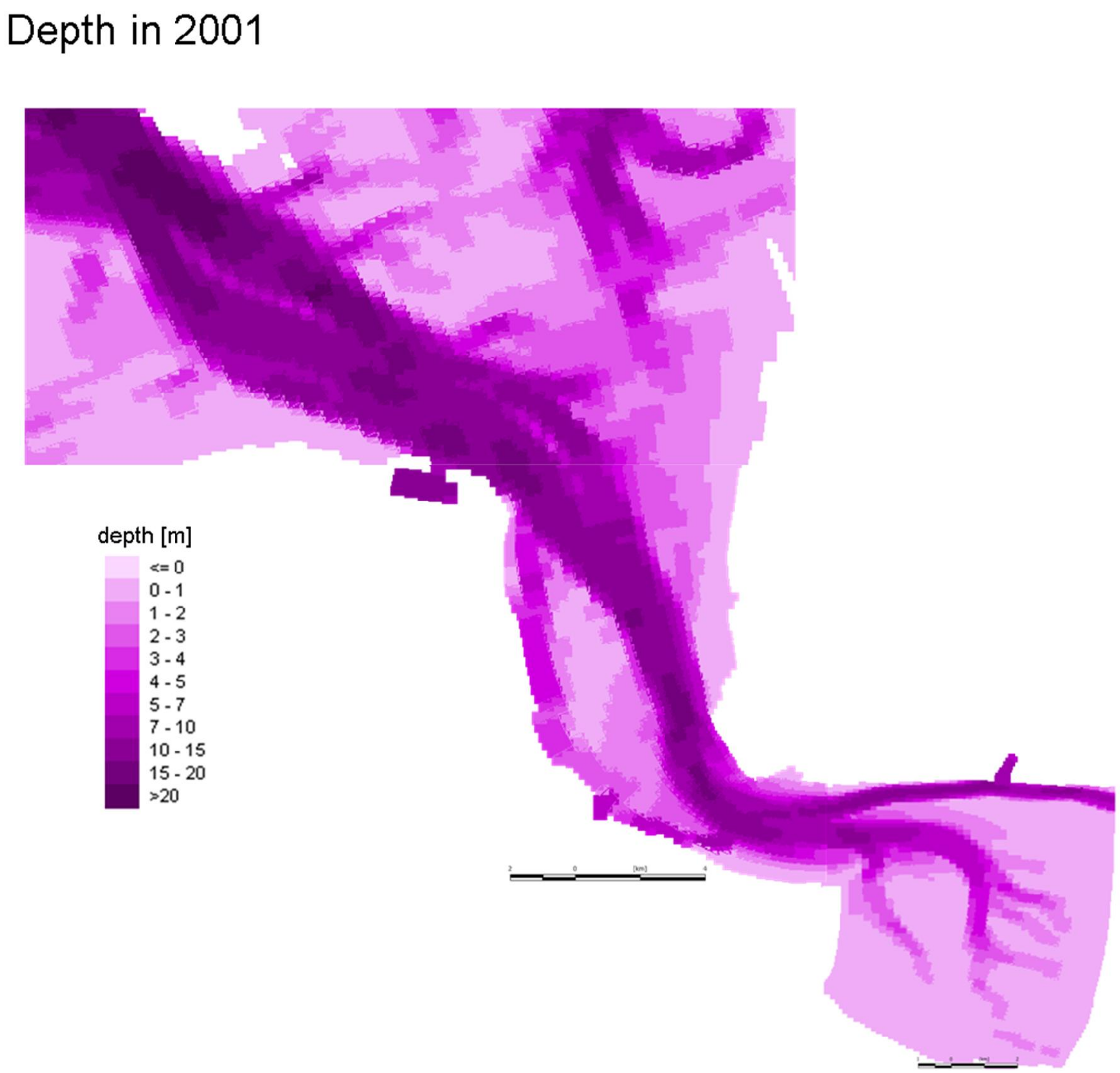

Figure B.1 Average depth in 2001.

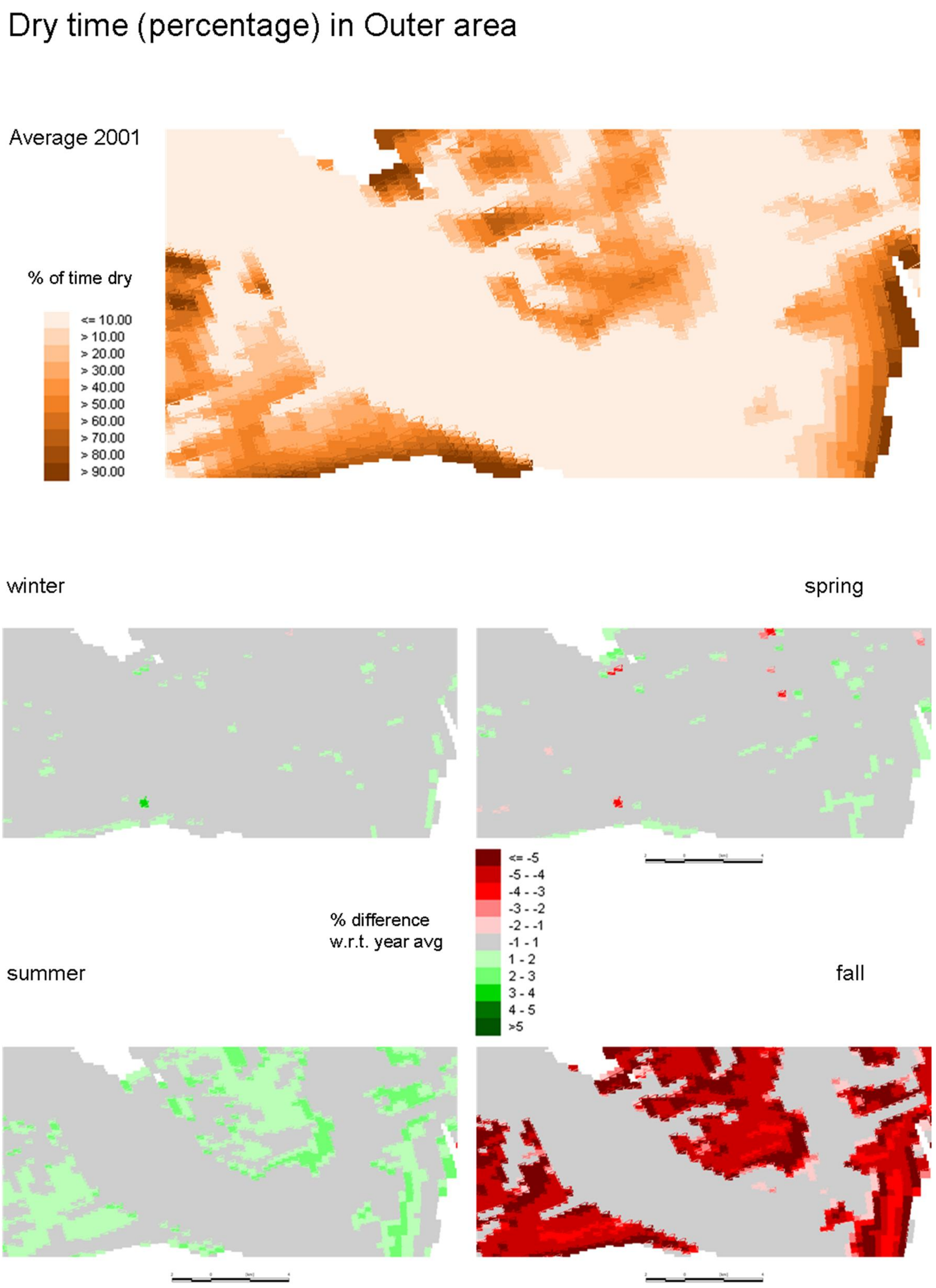

Figure B.2 Percentage of dry time in the outer area. Averaged over the year (upper panel) and the difference per season (lower panels).

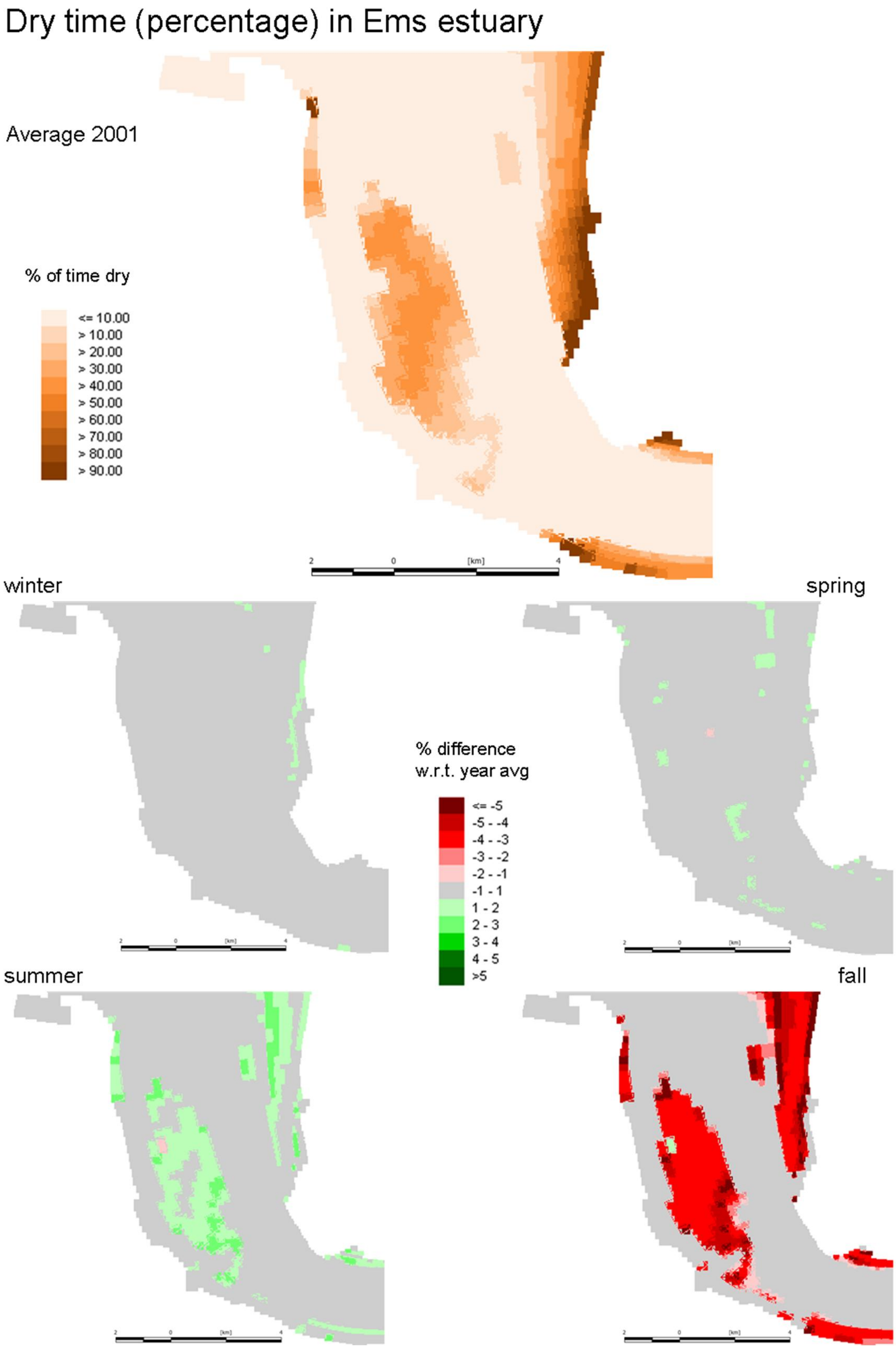

Figure B.3 Percentage of dry time in the Ems estuary. Averaged over the year (upper panel) and the difference per season (lower panels).

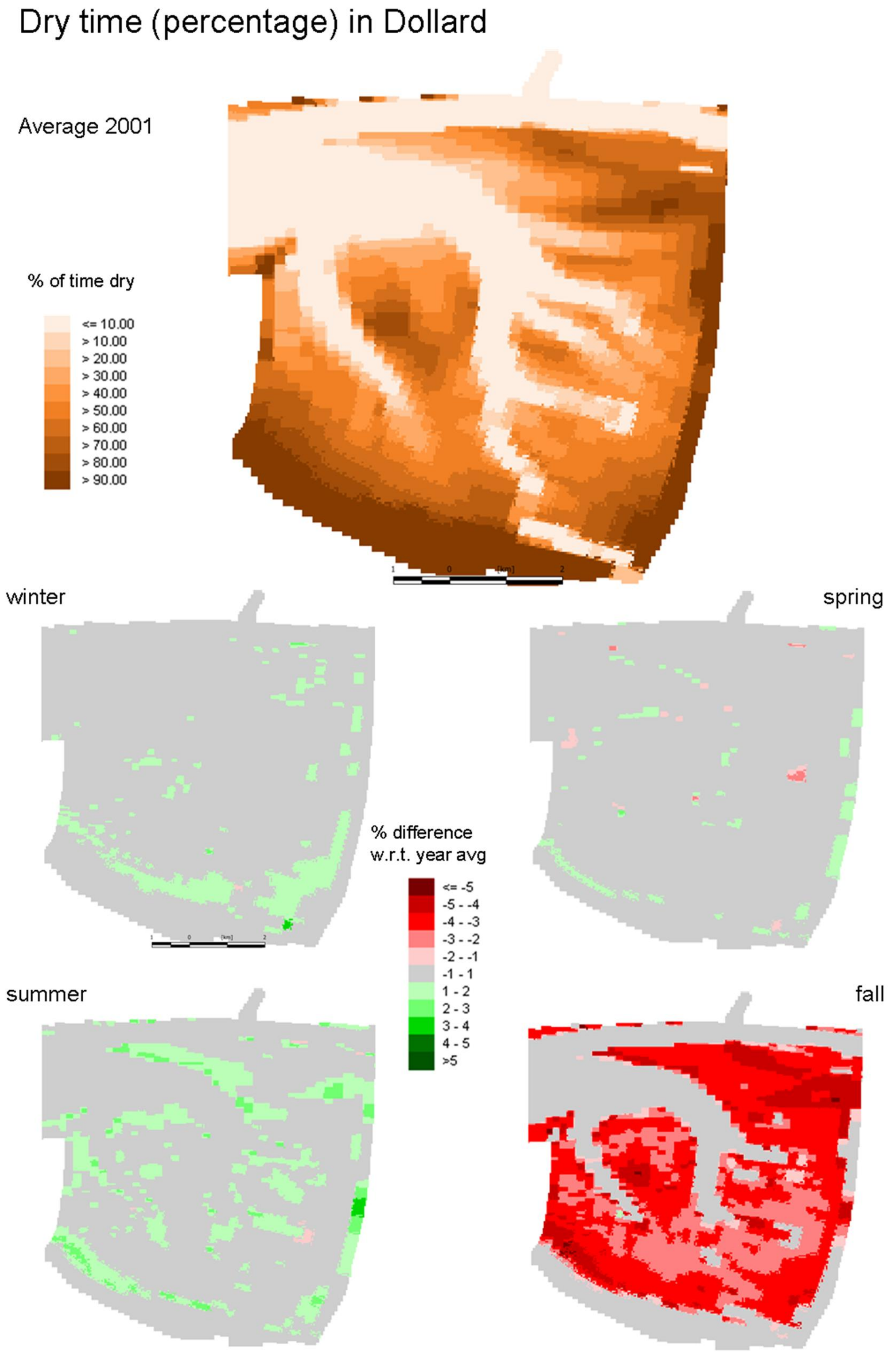

Figure B.4 Percentage of dry time in the Dollard. Averaged over the year (upper panel) and the difference per season (lower panels).

Maximum wave height throughout 2001

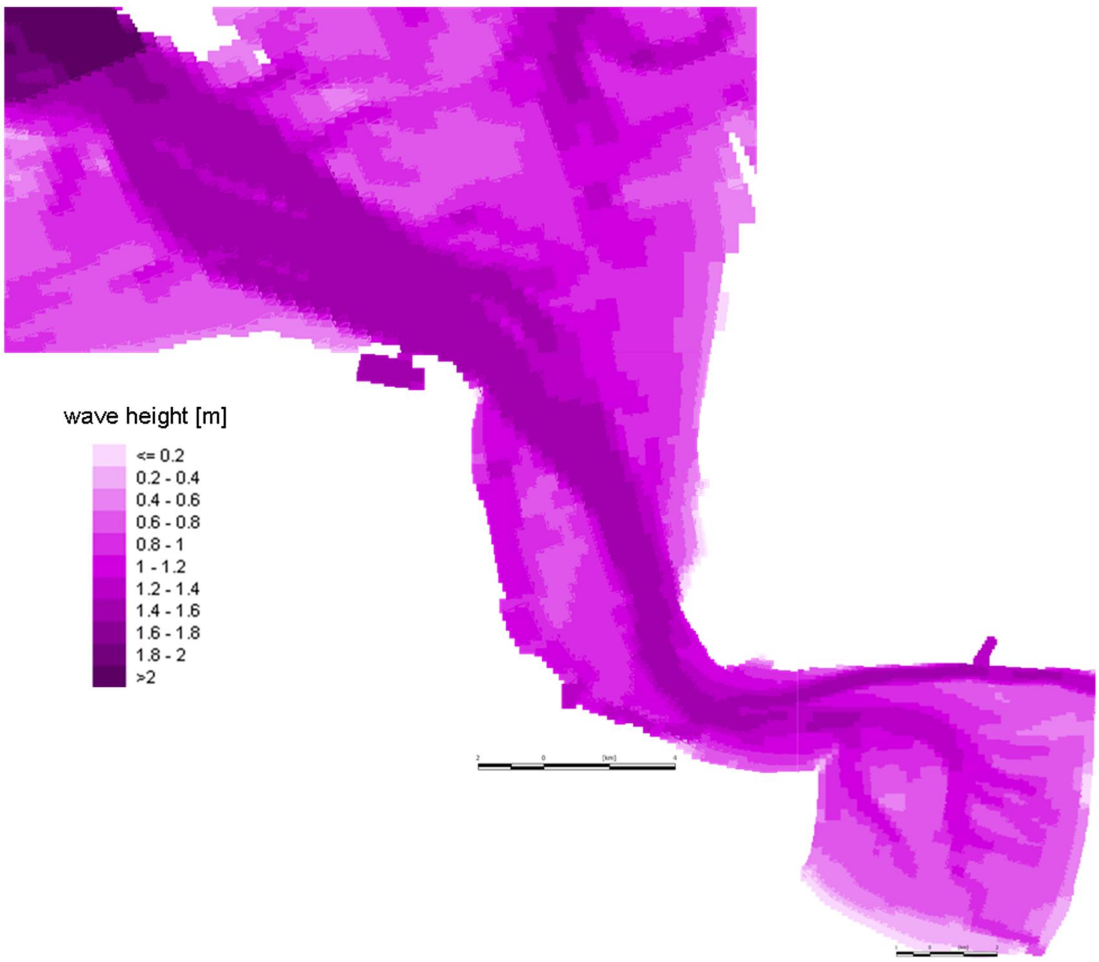

Figure B.5 Maximum wave height throughout 2001.

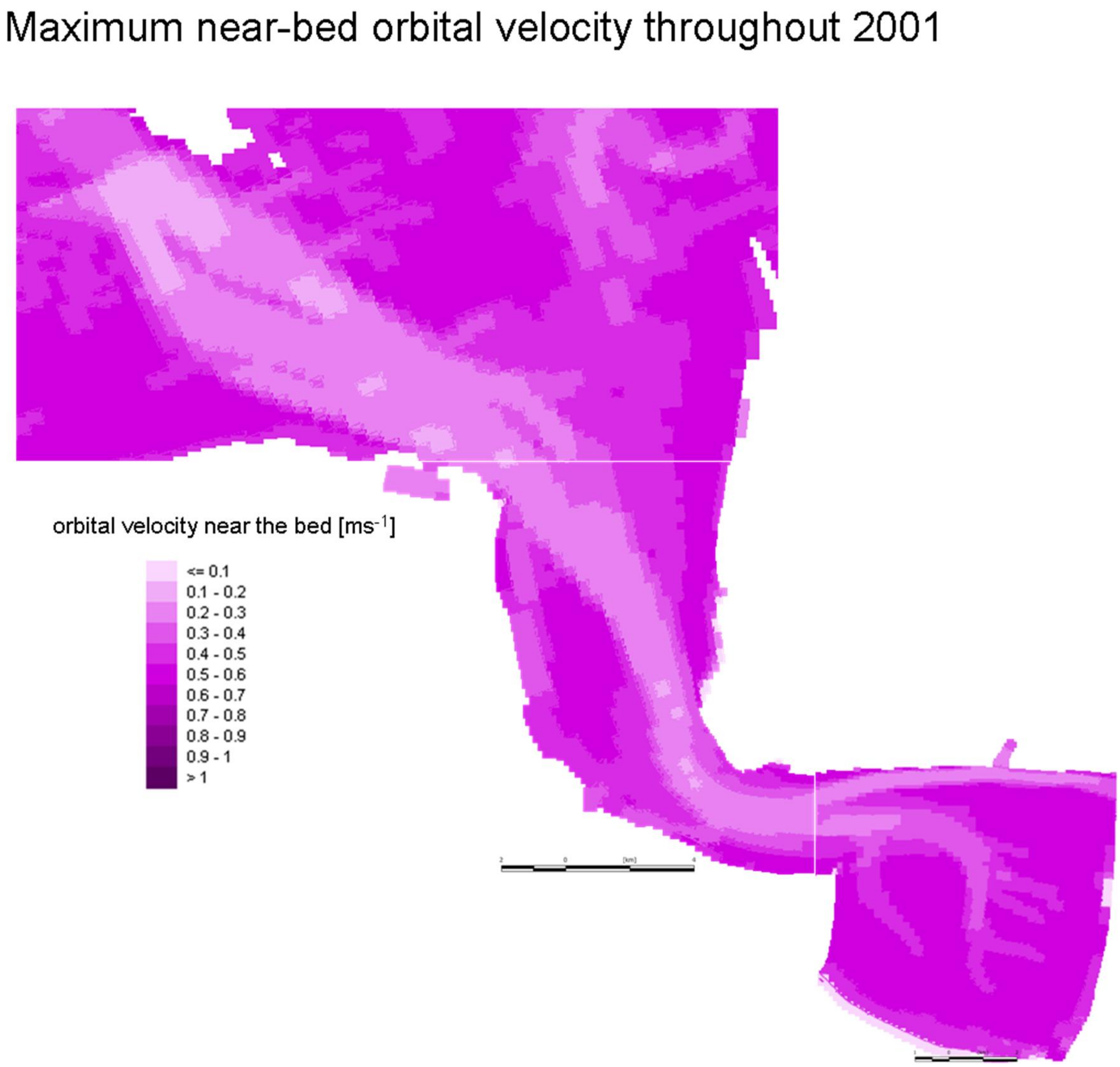

Figure B.6 Maximum orbital velocity throughout 2001.

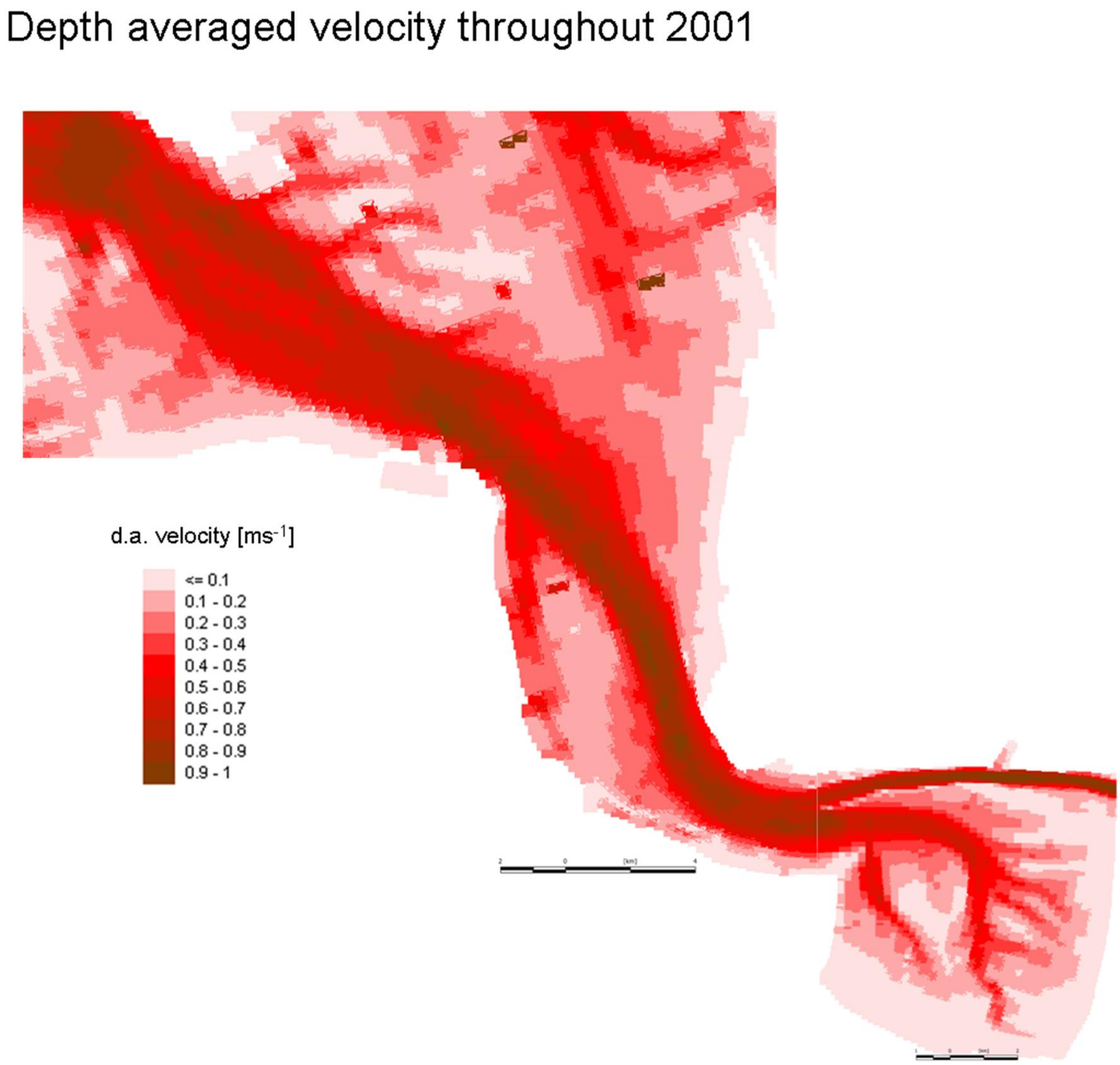

Figure B.7 Depth averaged velocity throughout 2001.

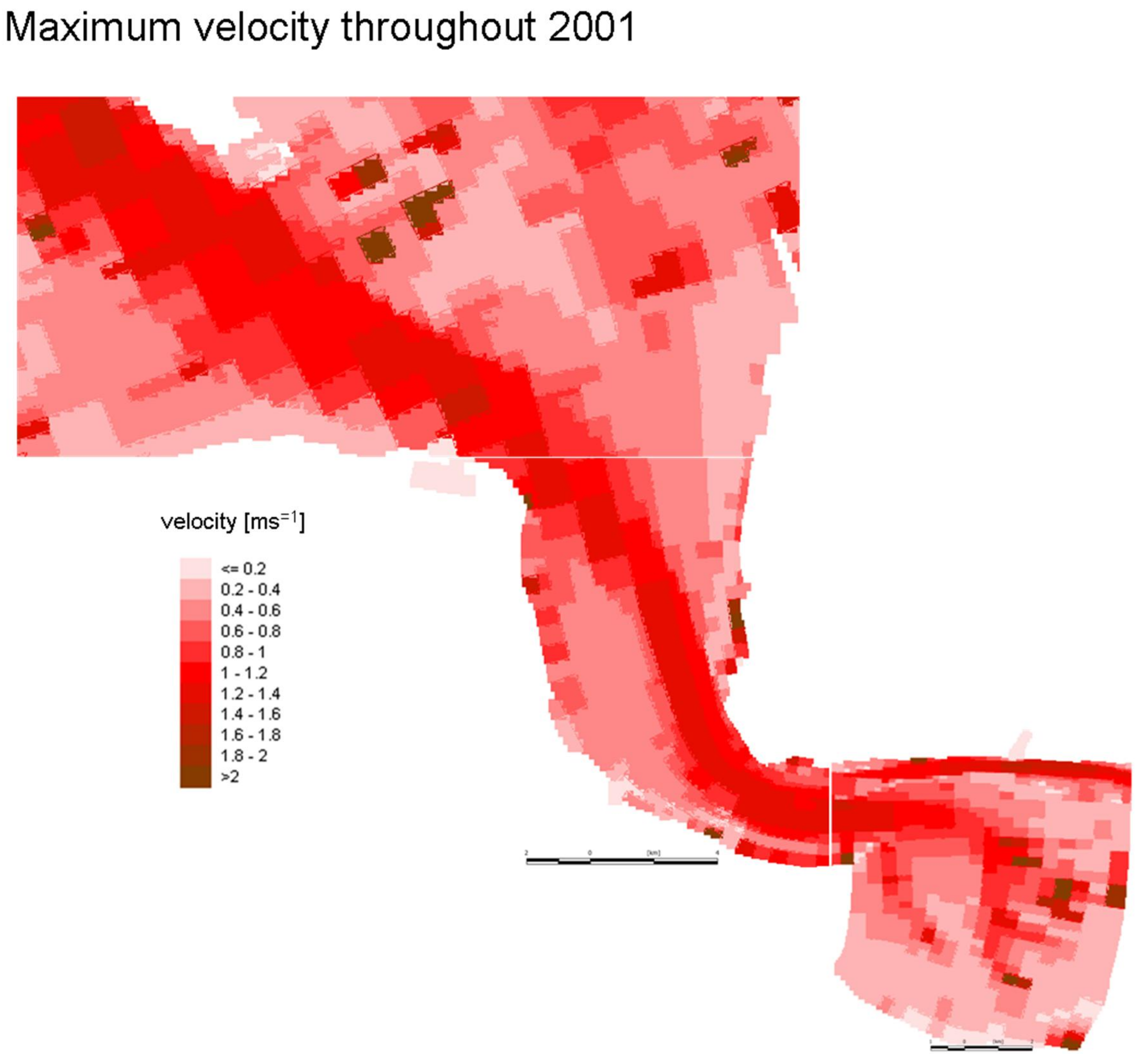

Figure B.8 Maximum depth averaged velocity in 2001 (NB different scale than time-averaged velocity).

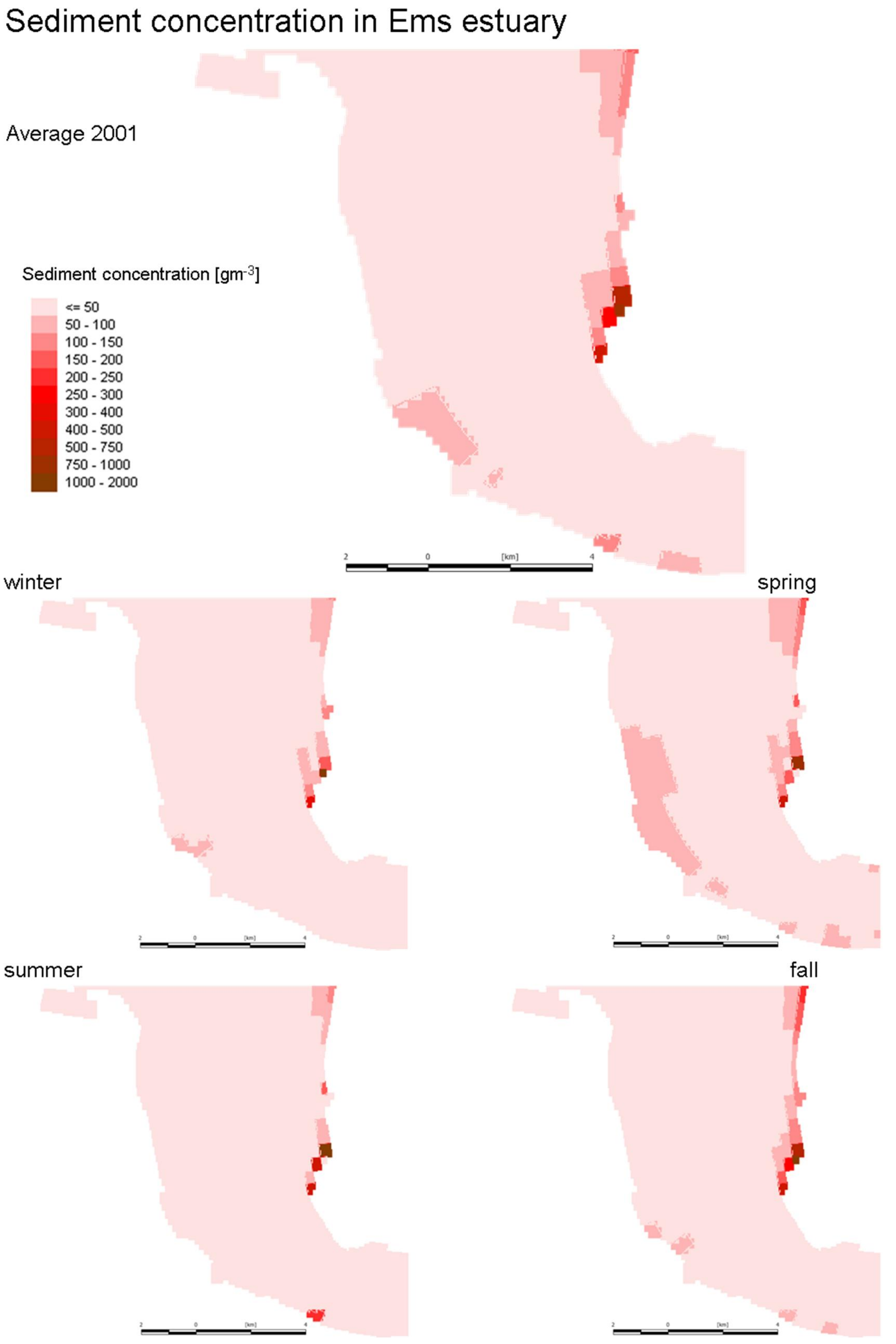

Figure B.9 Sediment concentration in the Ems estuary. Averaged over the year (upper panel) and the values per season (lower panels).

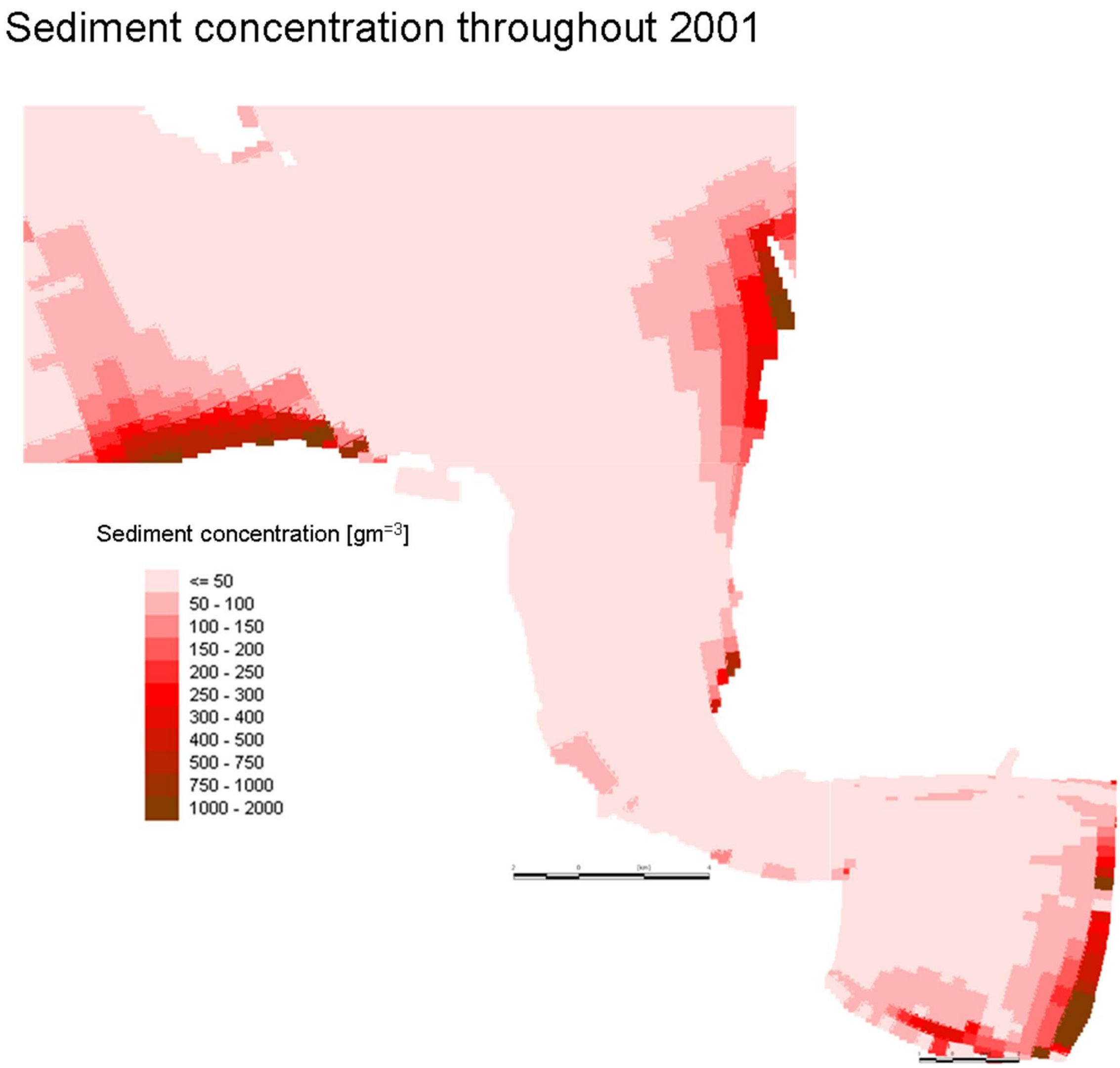

Figure B.10. Sediment concentration over 2001.

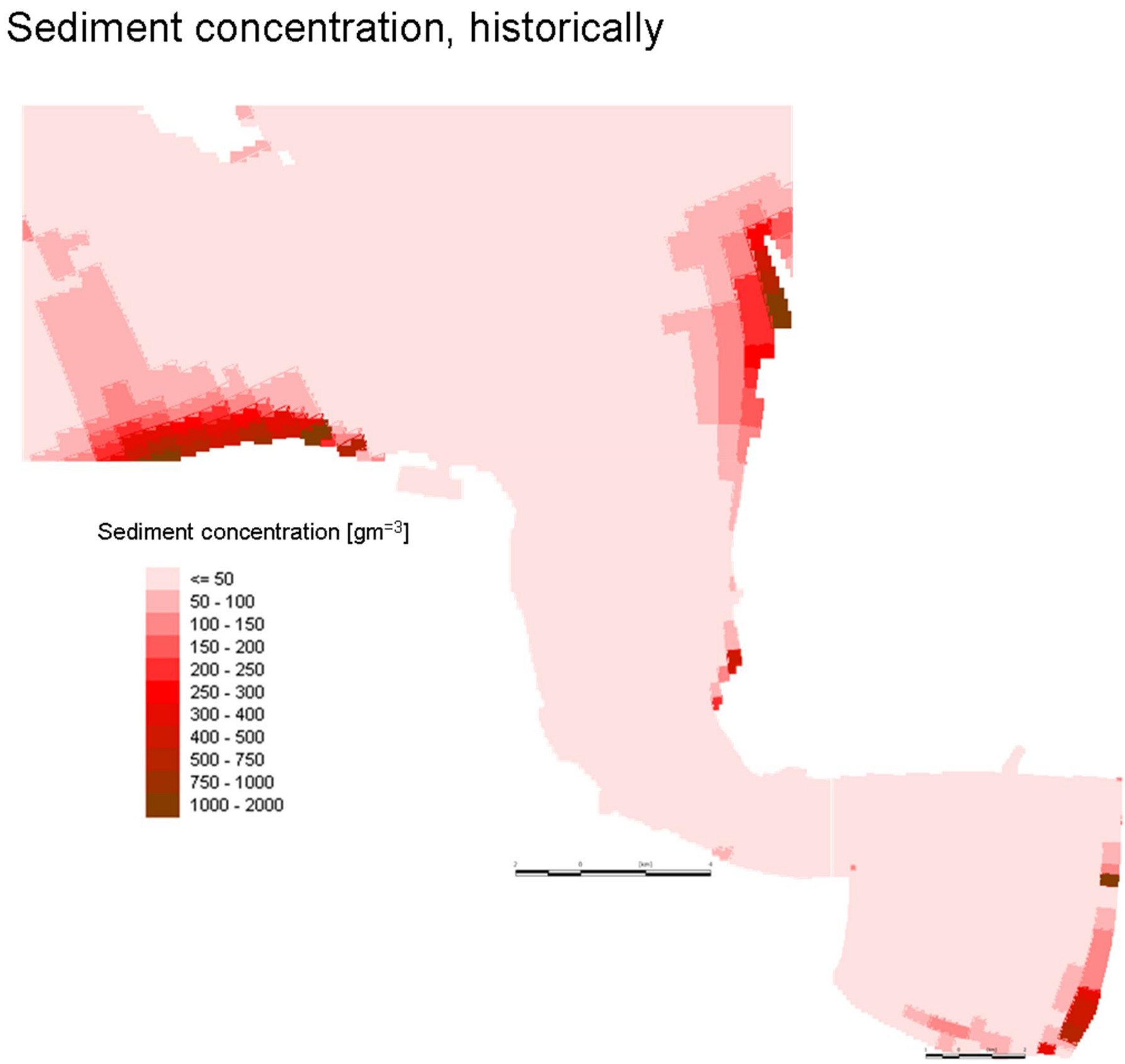

Figure B.11. Sediment concentration in the historical scenario. As Figure B.10 shows, the sediment concentration is fairly constant throughout all seasons for the shallow areas where concentrations are high. The sediment concentration dynamics in deeper areas, with concentrations below 50 gm^{-3}, can better be observed via the Secchi depth (Figures B.14-15).

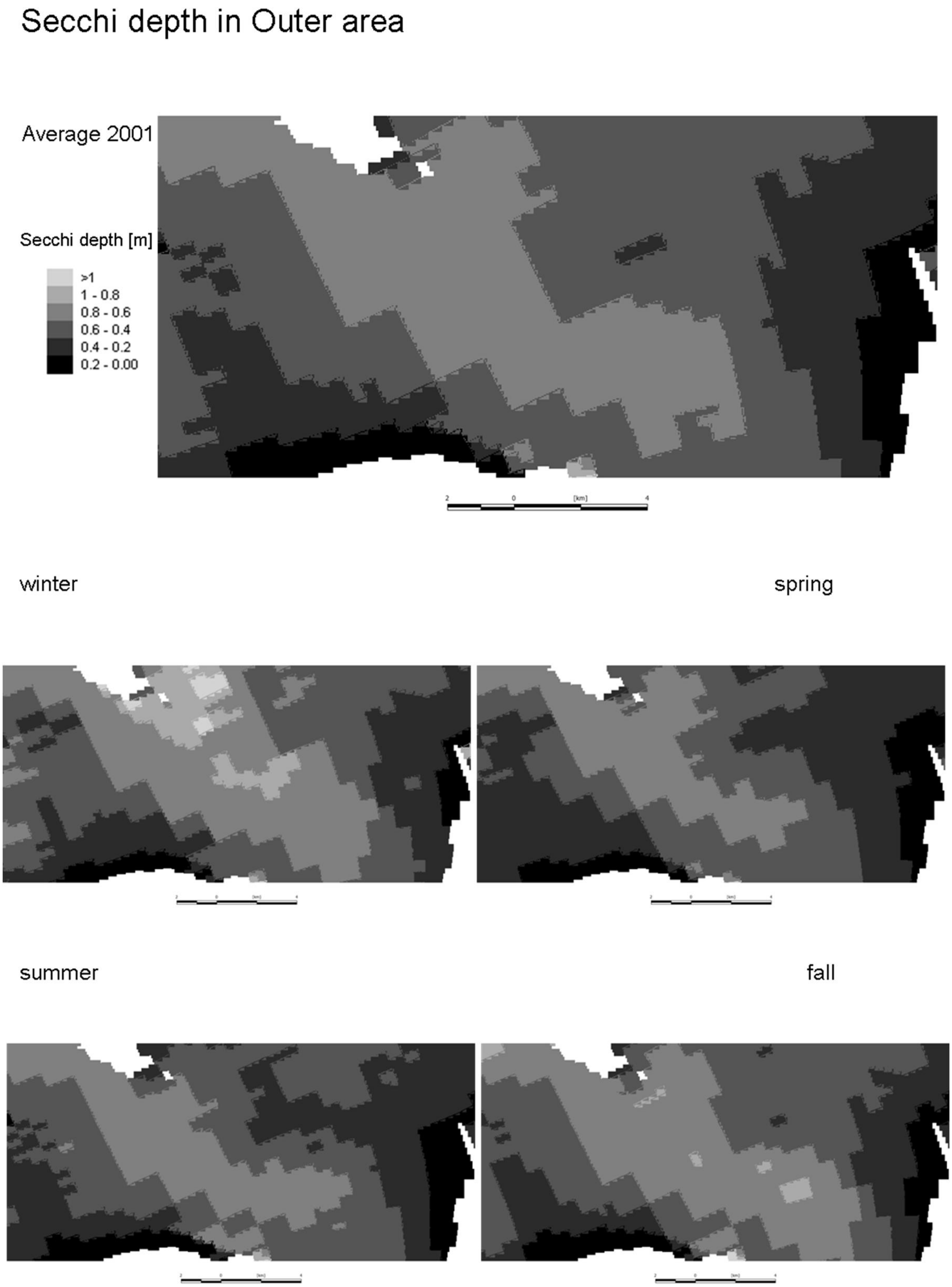

Figure B.12. Secchi depth in the outer area in 2001. Averaged over the year (upper panel) and the values per season (lower panels). The darker the colour, the more turbid the water.

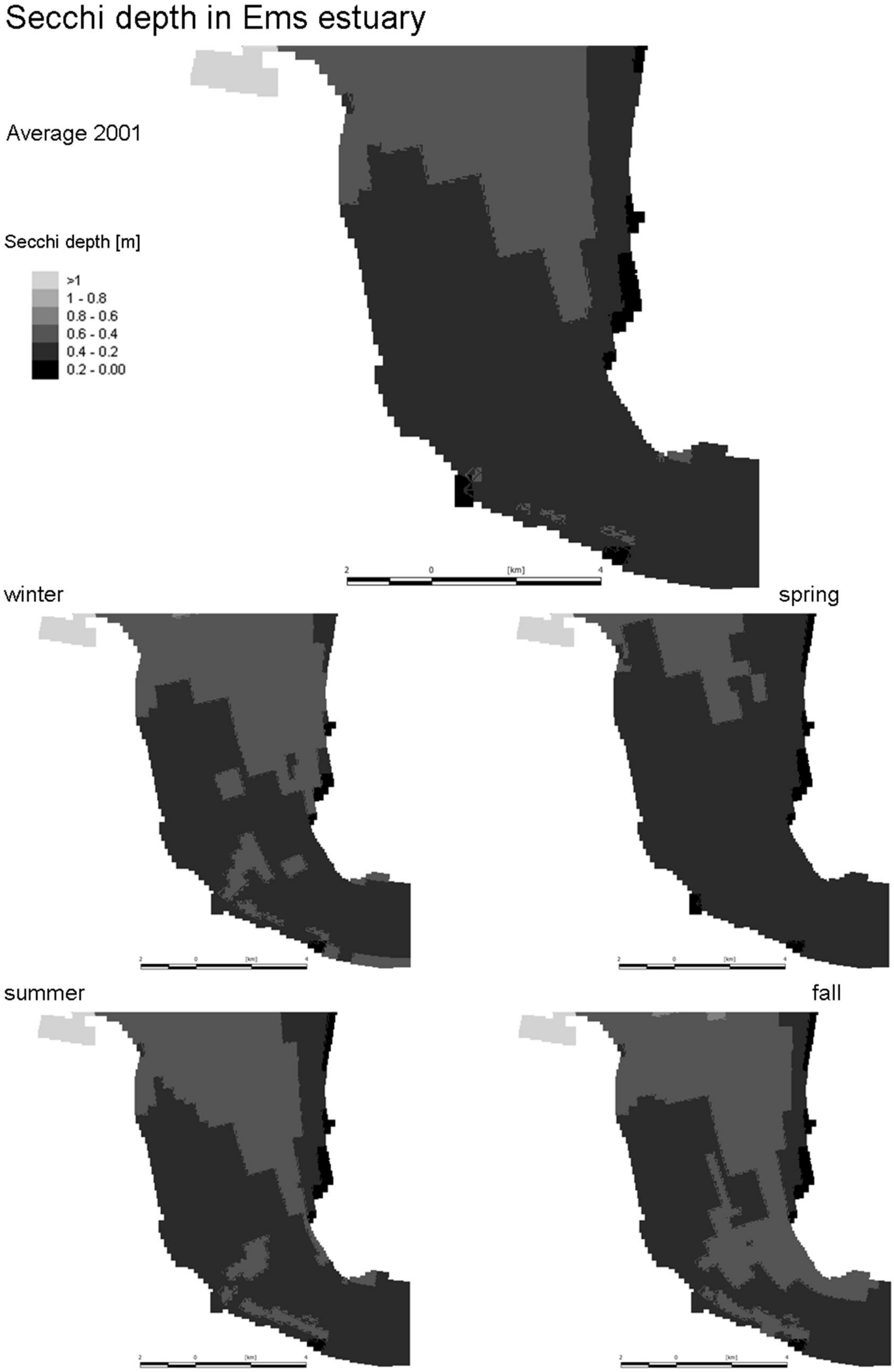

Figure B.13. Secchi depth in the Ems estuary in 2001. Averaged over the year (upper panel) and the values per season (lower panels). The darker the colour, the more turbid the water.

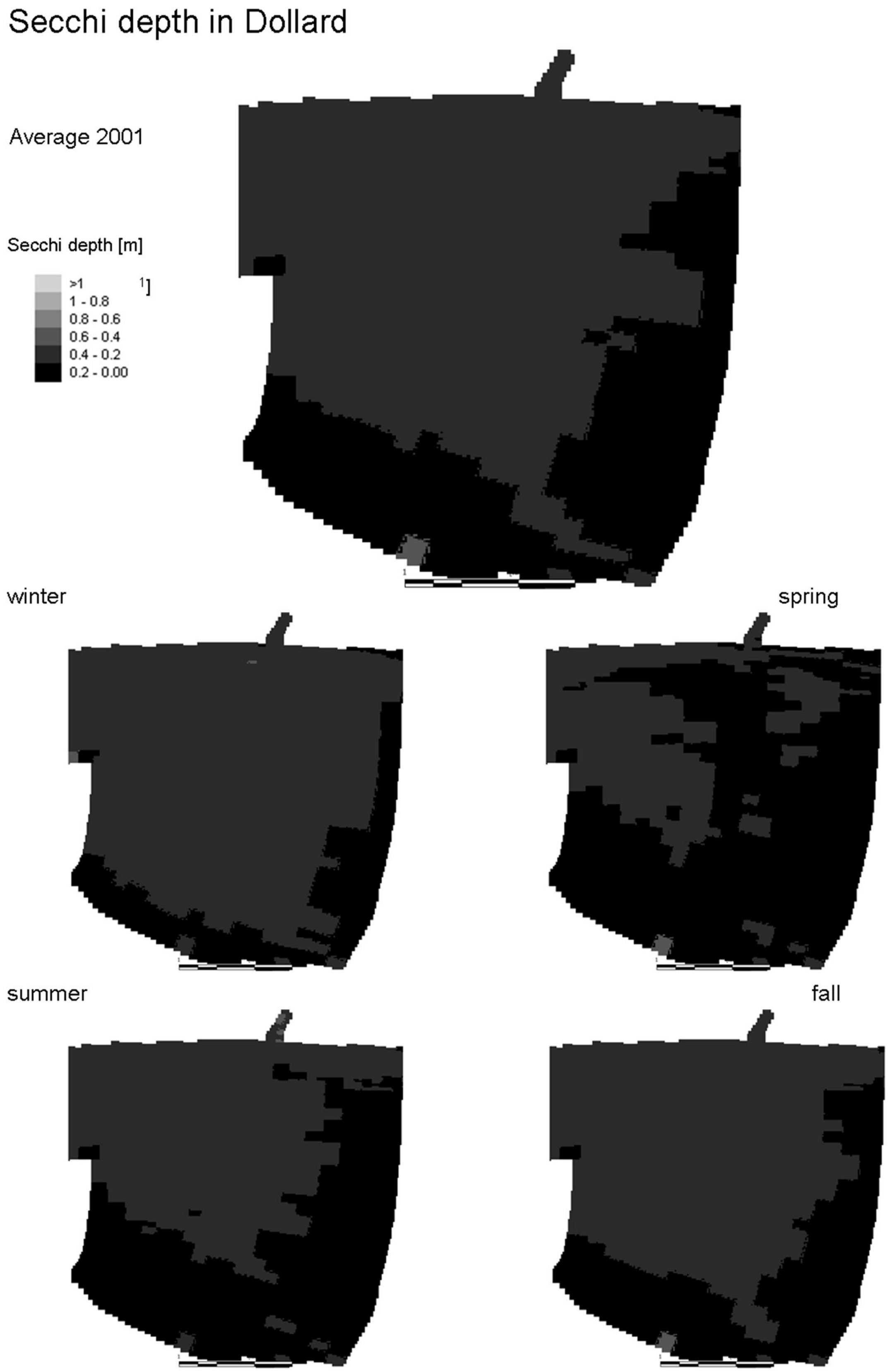

Figure B.14. Secchi depth in the Dollard in 2001. Averaged over the year (upper panel) and the difference per season (lower panels). The darker the colour, the more turbid the water.

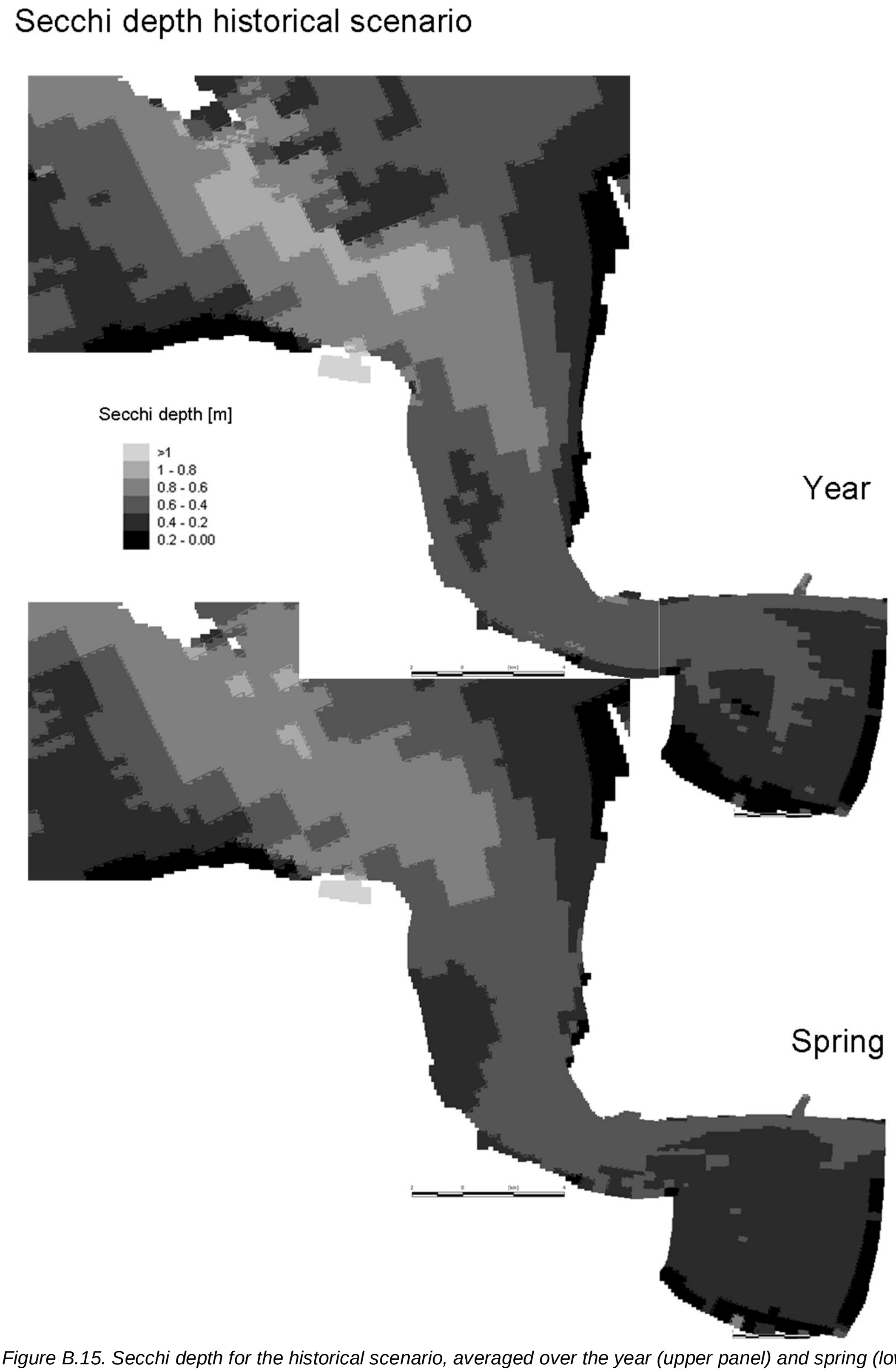

Figure B.15. Secchi depth for the historical scenario, averaged over the year (upper panel) and spring (lower panel). The dynamics throughout the year are the same as for 2001.

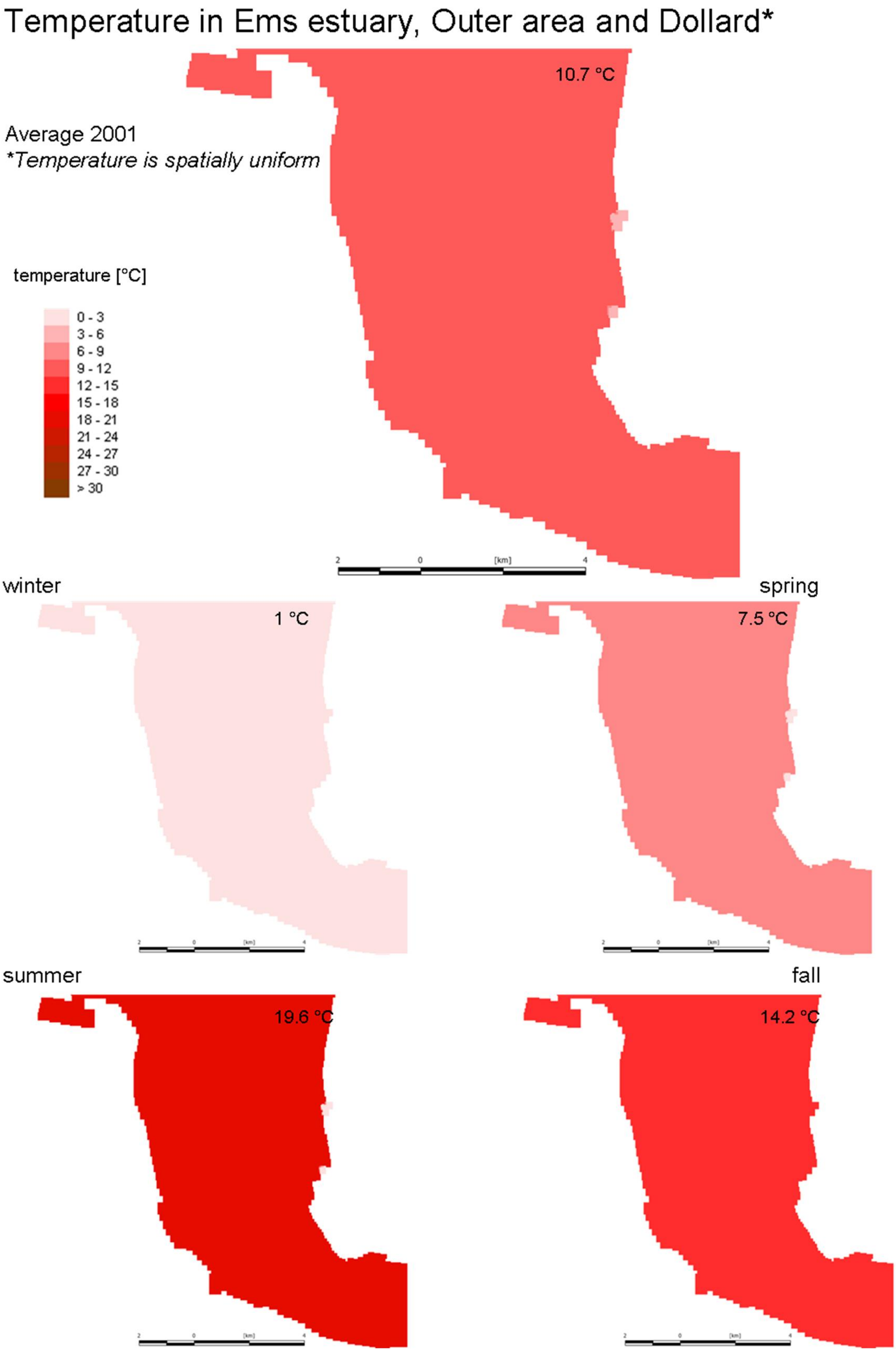

Figure B.16. Temperature in the Ems estuary. The time-averaged values for the outer area and the Dollard are the same; there is very little spatial variability.

Figure B.17. Salinity in the outer area. Averaged over the year (upper panel) and the values per season (lower panels).

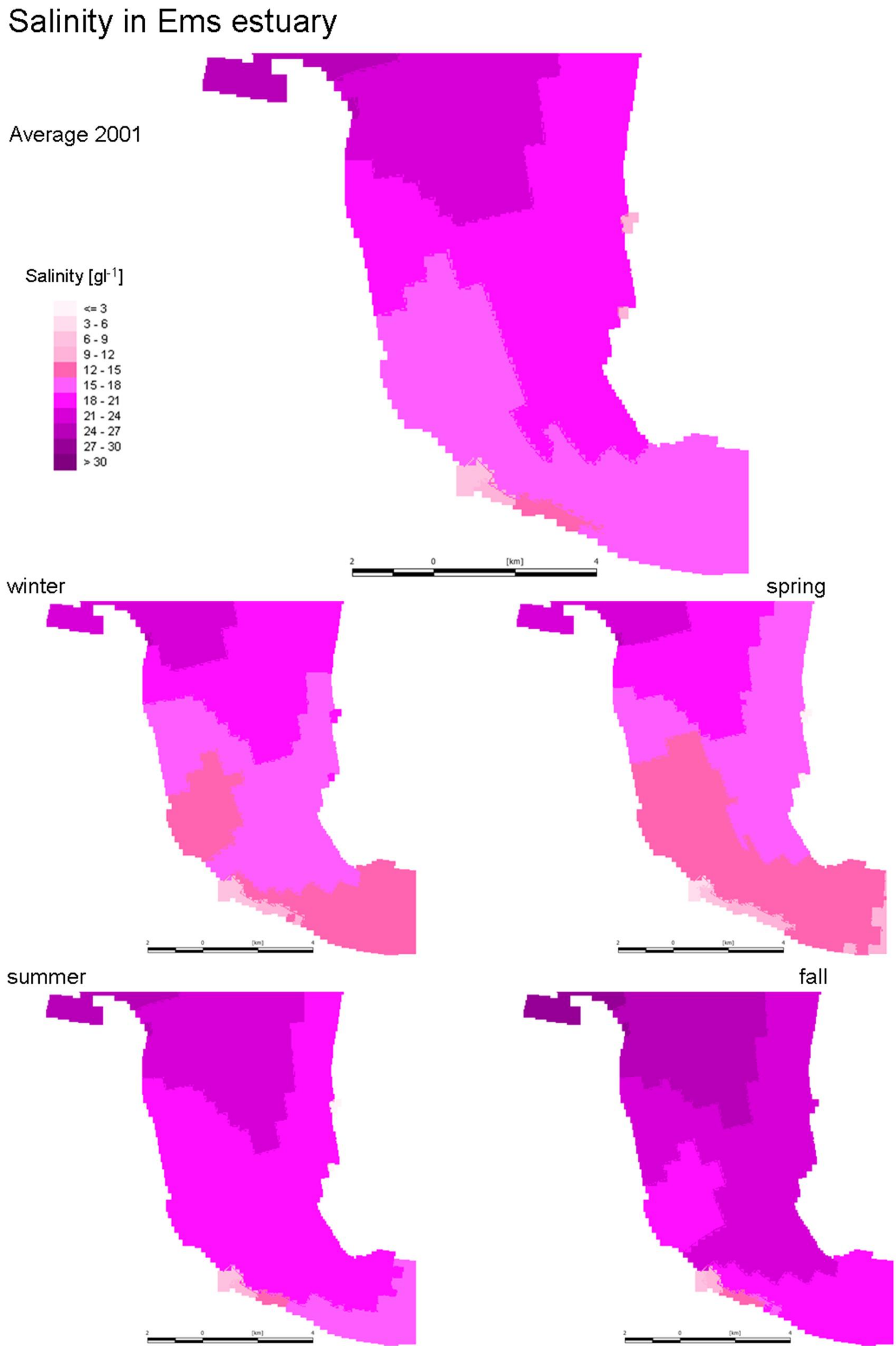

Figure B.18. Salinity in the Ems estuary. Averaged over the year (upper panel) and the values per season (lower panels).

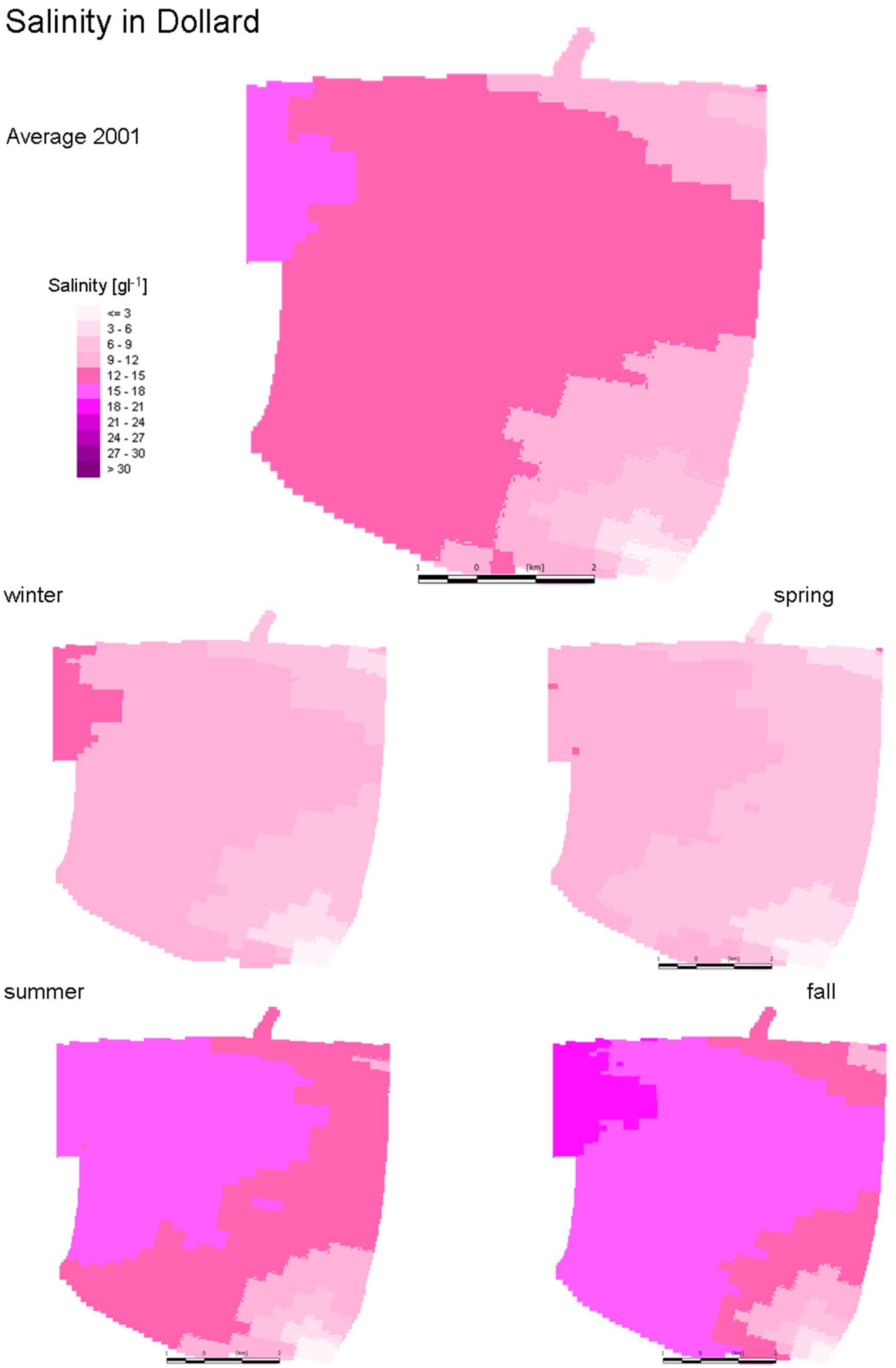

Figure B.19. Salinity in the Dollard. Averaged over the year (upper panel) and the values per season (lower panels).

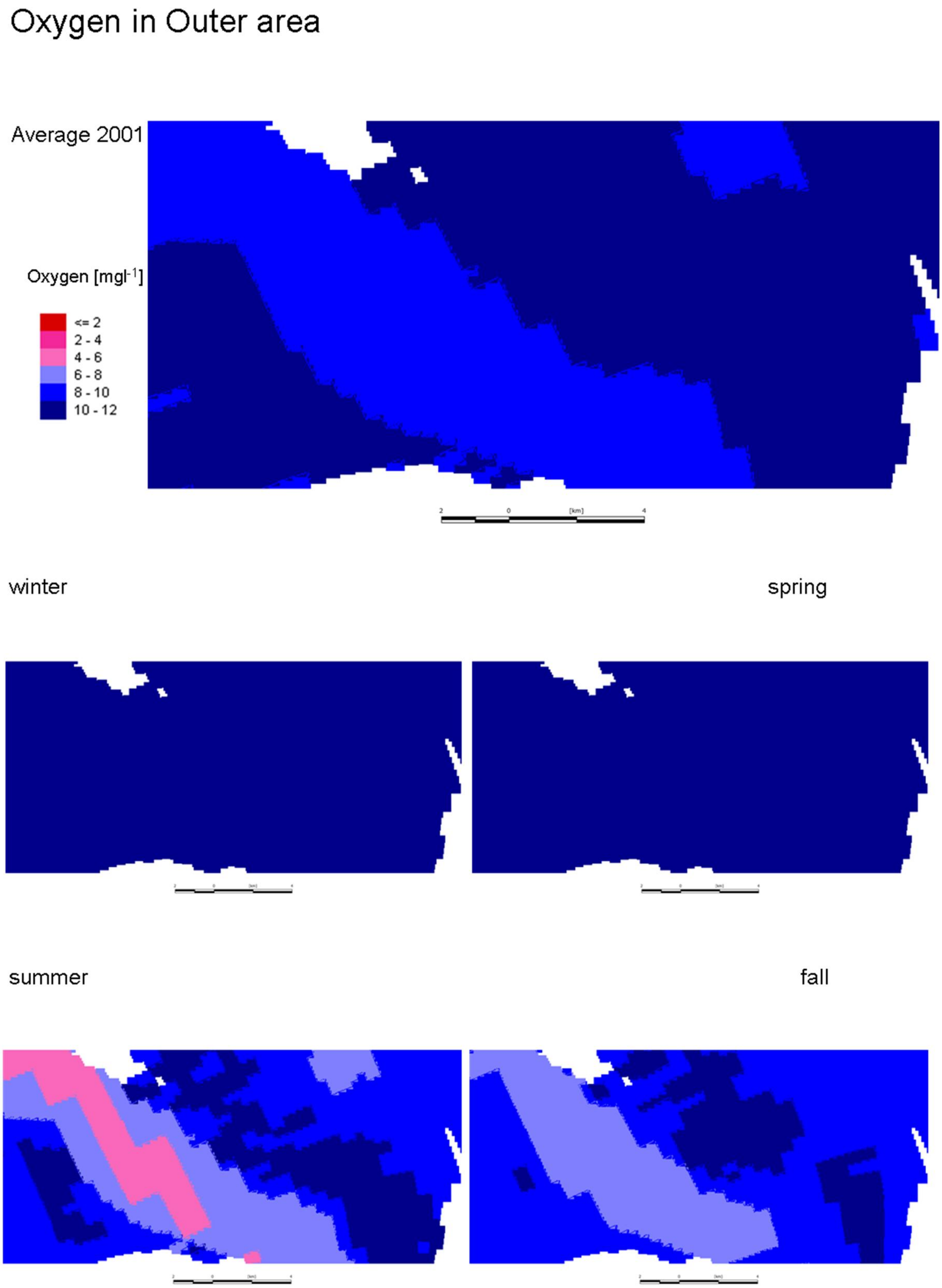

Figure B.20. Oxygen concentration in the outer area. Averaged over the year (upper panel) and the values per season (lower panels).

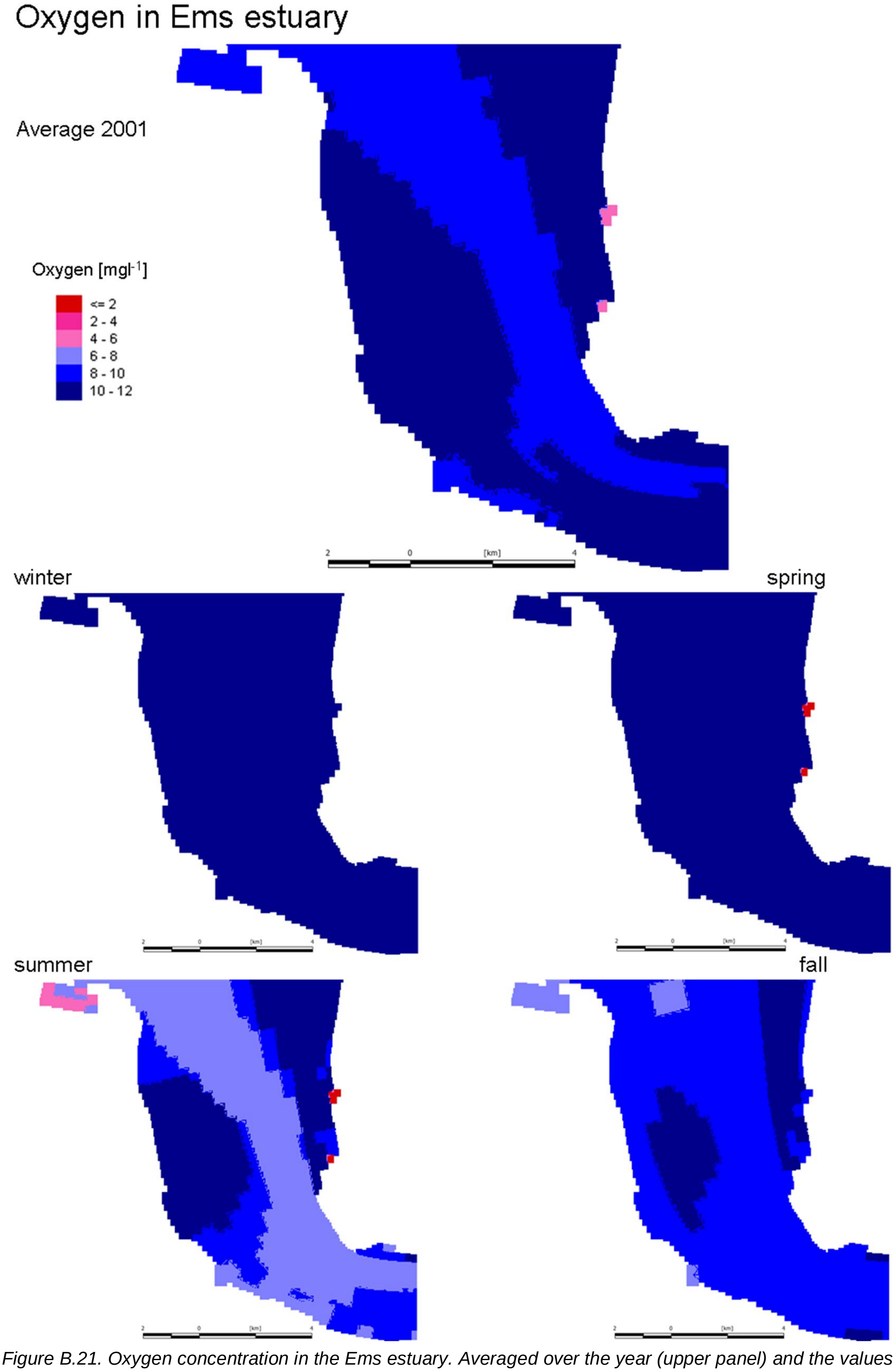

Figure B.21. Oxygen concentration in the Ems estuary. Averaged over the year (upper panel) and the values per season (lower panels).

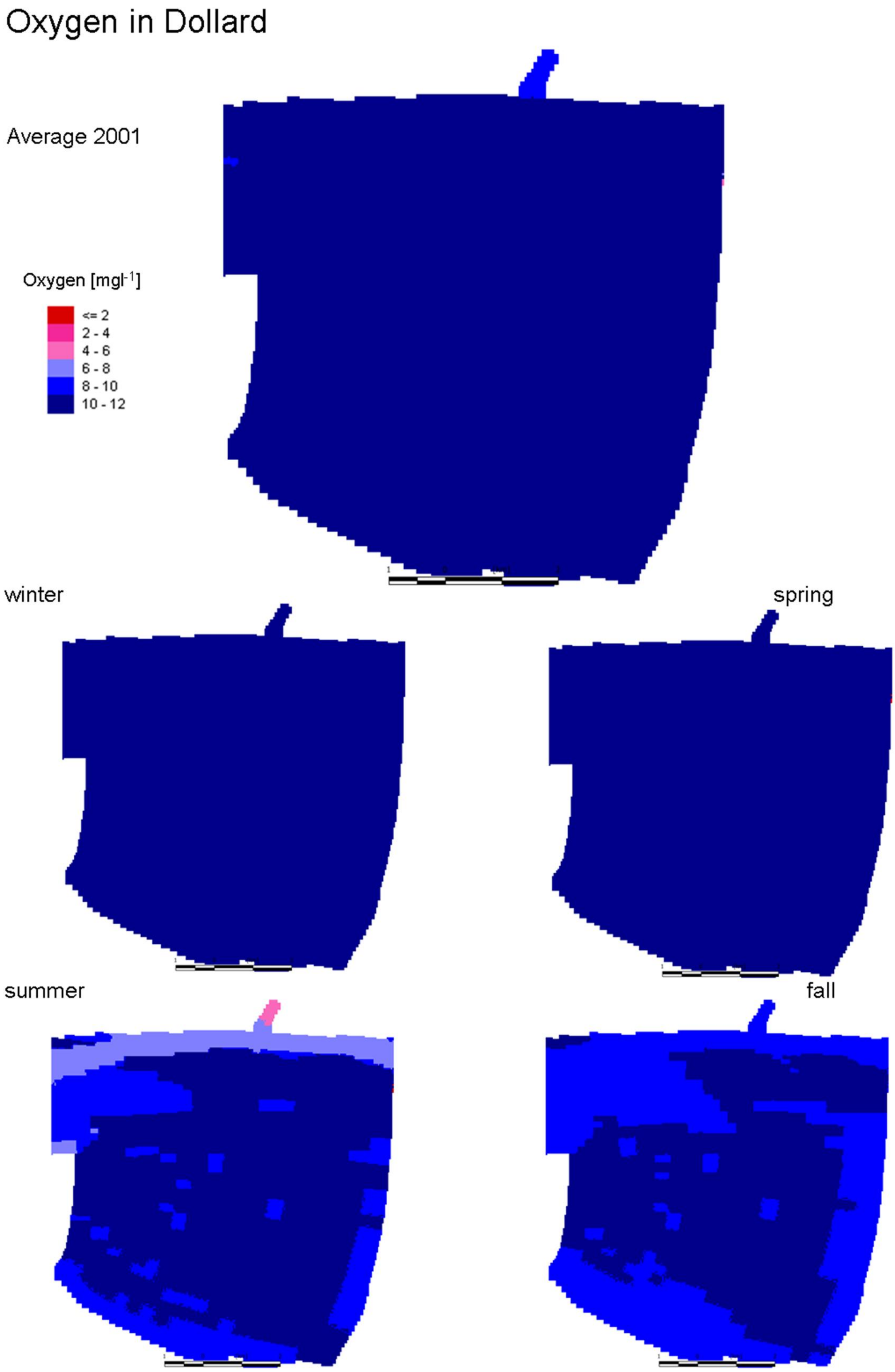

Figure B.22. Oxygen concentration in the Dollard. Averaged over the year (upper panel) and the values per season (lower panels).

Lowest (2nd percentile) oxygen concentration 2001

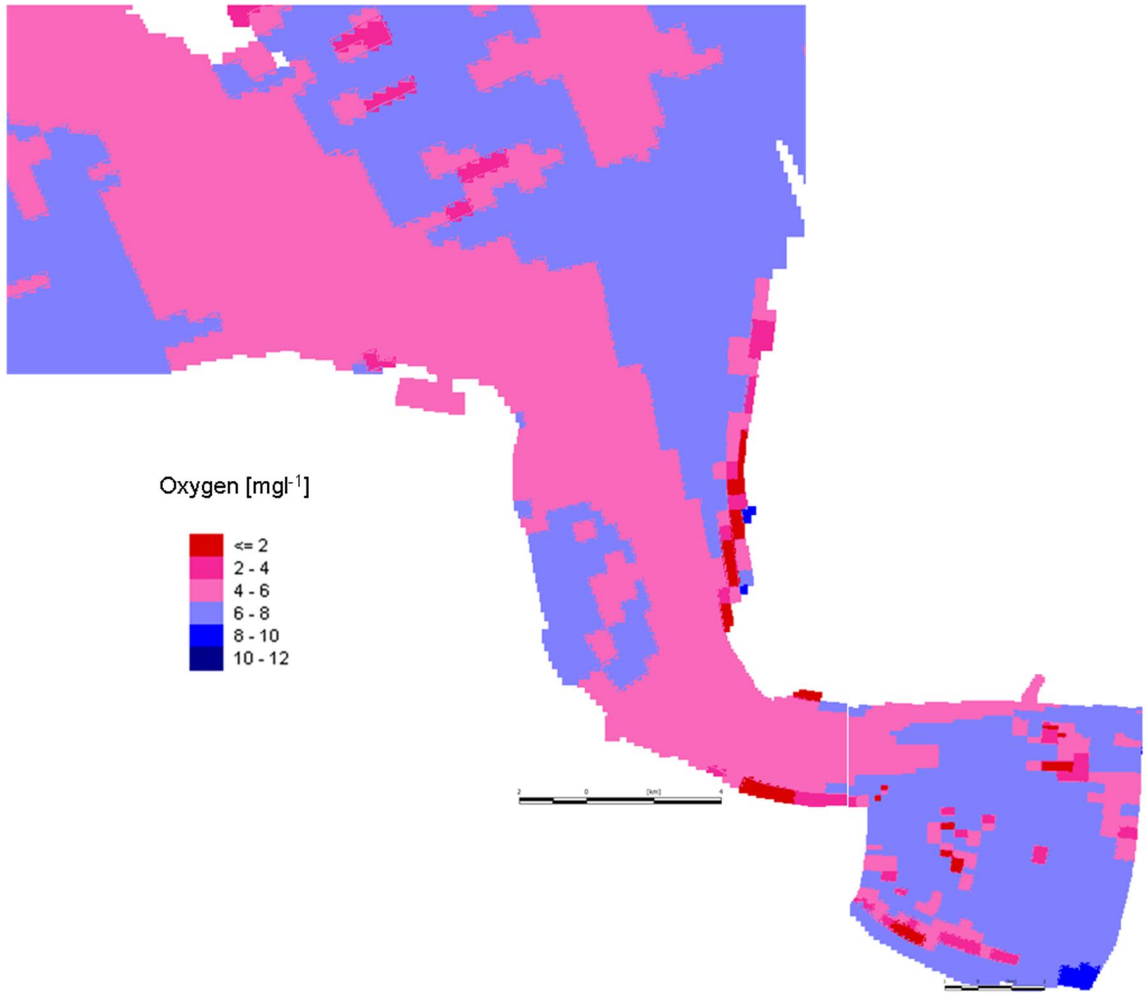

Figure B.23 2nd Percentile oxygen concentration in 2001.

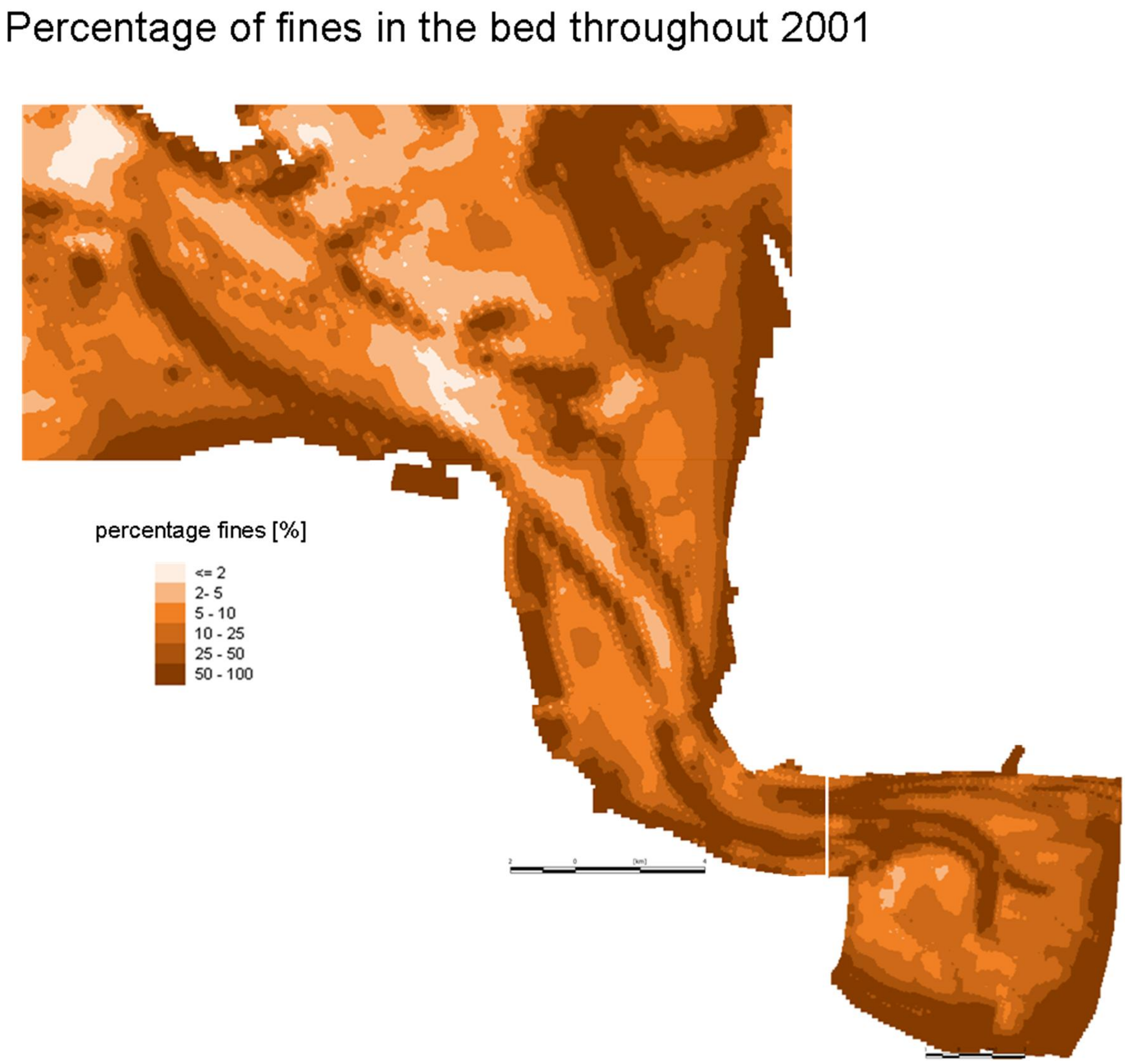

Figure B.24 Percentage of fine (<63µm) material in the bed in 2001. Values were very constant throughout the year.

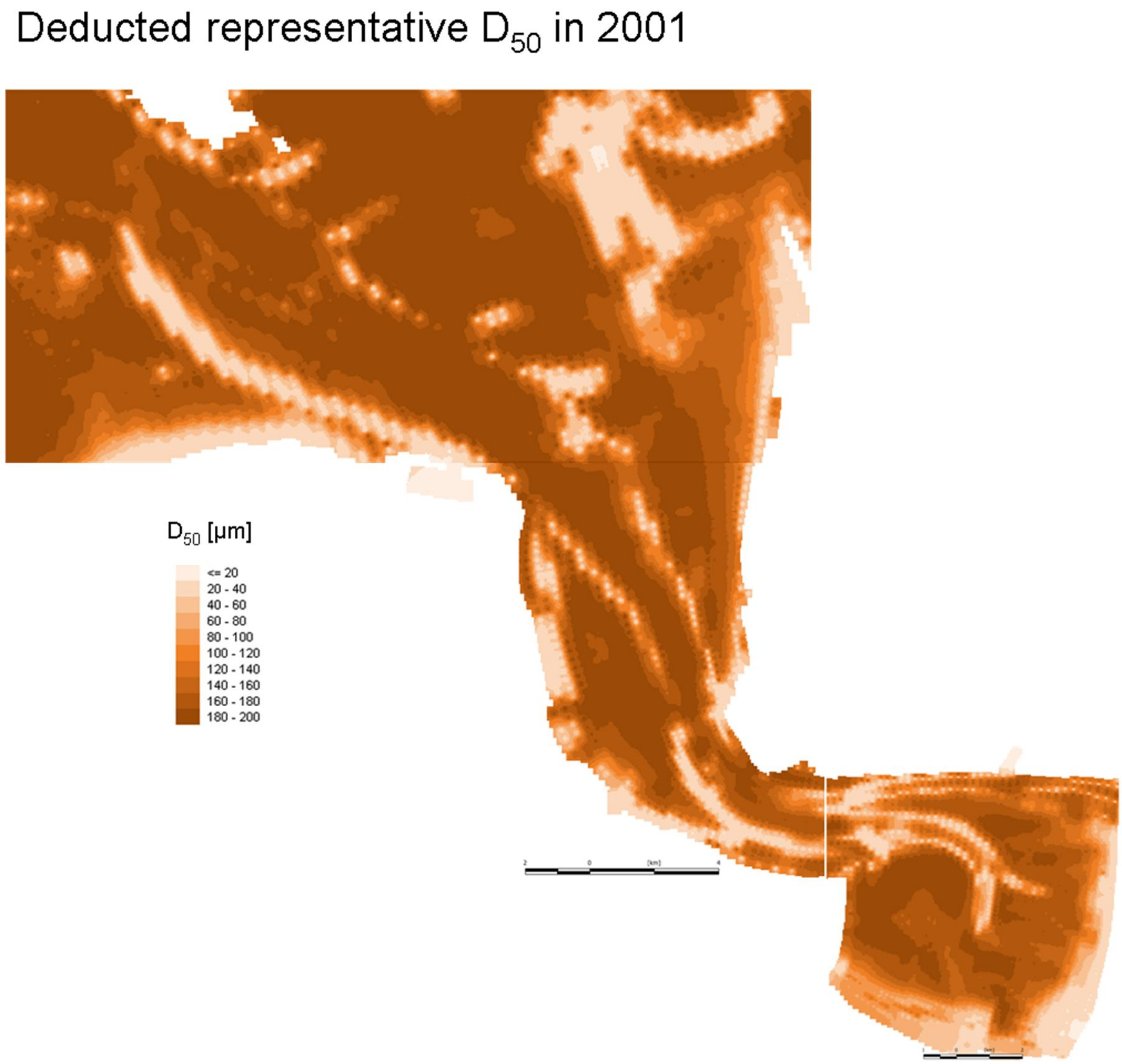

Figure B.25 D_{50} in 2001, as deducted from the percentage fines. As this map is based on simple estimations of constant grain sizes for the fractions of fine and coarse material, the bed material will actually be coarser near the North Sea and finer in the Dollard.

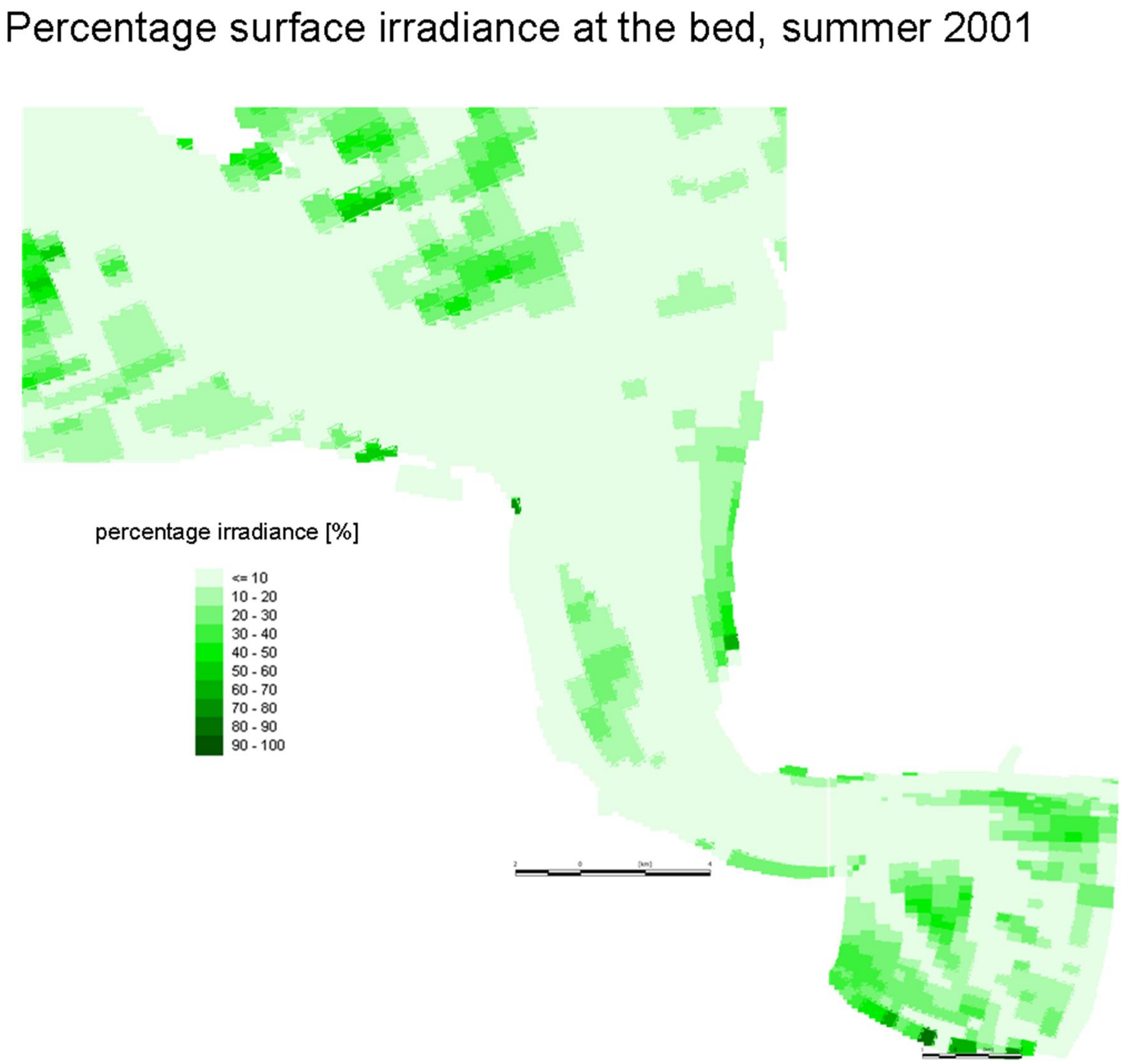

Figure B.26 Percentage surface irradiance that reached the seabed in summer 2001.