Deltares

Light and primary production in the Western Scheldt. Primary production model scenarios

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Samenvatting

De Vlaams-Nederlandse Schelde Commissie (VNSC) wil meer inzicht in de effecten van menselijke ingrepen in het Schelde-estuarium. Een belangrijk onderwerp in het onderzoeksprogramma van de VNSC is troebelheid. De troebelheid is grotendeels gerelateerd aan de slibconcentratie, die stuurt de hoeveelheid licht die in het water doordringt. In de Schelde blijkt de hoeveelheid licht zeer bepalend voor de groei van algen in de waterkolom (het fytoplankton). Deze groei vormt het grootste deel van de primaire productie (wat de basis van de voedselketen is).

Het is bekend dat veranderingen in troebelheid (zoals verschuivingen bij veranderde rivierafvoer of extra aanbod van slib door baggeren en storten) effect heeft op het hele voedselweb. Om dit effect te onderzoeken en te kunnen kwantificeren heeft Deltares voor de VNSC een 3D-model voor de primaire productie in de Westerschelde ontwikkeld. Met dit model zijn gevoeligheidsanalyses en drie berekeningen van ingrepen/scenarios gedaan. Hierover wordt gerapporteerd in dit rapport.

De gevoeligheidsanalyse laat zien dat het effect van veranderingen in troebelheid afhangt van de plaats in de Schelde. Dit komt door de verschillen in gemiddelde verblijftijd tussen gebieden in het estuarium en door het diepteprofiel. Bij grotere diepte is er lokaal, gemiddeld, nog nauwelijks licht beschikbaar. Minder troebelheid brengt in die dieptegemiddelde lichtdoordringing weinig verandering. In gemiddeld ondiepere delen met relatief hoge troebelheid kan een verheldering wel veel effect hebben.

De gevoeligheidsanalyses tonen ook aan dat diatomeeën sterker reageren op een verandering in troebelheid (en dus licht) dan niet-diatomeeën. Dit is een belangrijke observatie, want diatomeeën zijn een hoogwaardiger voedsel in het voedselweb dan andere algen in de waterkolom. De resultaten laten zien dat minder slib (op de juiste plaatsen) niet alleen de primaire productie verhoogt, maar dat zoöplankton ook meer gaat grazen. Het netto effect is een hogere primaire productie en consumptie door zoöplankton, wat ten goede kan komen aan de rest van het ecosysteem.

De ingreep die is doorgerekend betreft de aanleg van Deurganckdok. Dit heeft geleid tot een aanslibbing van sediment. Deze sedimentatie vermindert het slibgehalte, dus de troebelheid, in de Westerschelde. Dit doet de primaire productie toenemen. Deze toename wordt niet teniet gedaan door het lokaal baggeren en elders storten van dit slib. Het wordt namelijk wel weer in de Schelde teruggebracht, maar stroomopwaarts van Antwerpen. Op die plaats is het water al erg troebel en daardoor vindt er bijna geen primaire productie meer plaats in de waterkolom. Het model laat zien dat de aanleg van het Deurganckdok, in combinatie met de wijze van onderhoud, een netto positief resultaat op de helderheid en de primaire productie heeft gehad.

De tweede berekening betreft een scenario van een relatief droog jaar (hiervoor is 2011 gekozen). Een lagere rivierafvoer in de winter had een lagere gemiddelde troebelheid tot gevolg. Hierop nam de primaire productie in de Westerschelde met ongeveer 20 % toe. Een deel van de toename werd ook weer gelijk geconsumeerd door zoöplankton, waarvan de rest van het voedselweb kan profiteren. Bij de berekening van fytoplanktonproductie in dit scenario is niet meegenomen dat ook de verblijftijd toeneemt bij lagere rivierafvoer. In werkelijkheid kan de productie van fytoplankton en zoöplankton daardoor nog groter zijn in een relatief droog jaar.

De derde berekening betreft een scenario van geplande stortactiviteiten in verband met de herbouw van het sluizencomplex bij Terneuzen. Onderzocht is de verandering in troebelheid in een scenario waarin het slib in het begin van het jaar, net voor het groeiseizoen, gestort wordt. Dit had slechts een matig negatief effect op primaire productie(max 10 %) in vergelijking met niet storten. Het effect zou bij storten in het groeiseizoen veel groter zijn geweest. In de licht-gelimiteerde Westerschelde heeft elke verandering van troebelheid direct gevolg voor primaire productie door fytoplankton. De effecten zijn het minst in de wintermaanden, wanneer nauwelijks productie plaatsvindt.

Effecten zijn ook minimaal in de overgang met de Zeeschelde, waar door de diepte en het hoge slibgehalte al niet veel productie voorkomt. De sterkste respons is overwegend in het middendeel van de Westerschelde, waar voldoende ondiep water is voor primaire productie, en toch licht gemiddeld het gehele jaar limiterend is.
Kanttekening bij het overnemen van de modelresultaten is dat in het model zoöplankton sterk reageert op de verhoogde primaire productie. Of dit in werkelijkheid ook zo is, is lastig te voorspellen, omdat de precieze reactie sterk af zal hangen van de hoeveelheid en soorten zoöplankton die op een bepaald moment aanwezig zijn. De informatie om deze reactie goed in het model in te bouwen is niet beschikbaar. Meer informatie over groei van en graas door zooplankton in de Schelde is nodig.

Inhoud

1 Introduction

The Schelde estuary has been recovering from severe nutrient pollution and oxygen deficiency (Cox et al. 2009; Meire et al. 2005). Still, it is heavily influenced by human activities, directed to safety, transport, and natural resources. This causes pressures on the natural habitats in the Schelde and its shores. Understanding the natural processes and how they are influenced by human activities is essential for management of the Schelde estuary.
Primary production by pelagic and benthic algae is the energetic basis of the natural habitats in any marine ecosystem. In the Schelde, pelagic primary production is strongly limited by the amount of available light (Spaendonk 1993; Kromkamp and Peene 1995) due to the high concentrations of suspended sediments. Benthic production is probably limited by the surface area that is available, in practice, this is the surface area of tidal flats. Therefore, benthic production is less sensitive for light extinction by suspended sediment in the water column.
Pelagic primary production results in phytoplankton biomass, but the ambient concentration of phytoplankton depends heavily on the residence time of a water parcel in favorable conditions (Muylaert, Sabbe, and Vyverman 2009), as well as loss terms such as grazing by zooplankton and benthic filter feeders. It is therefore difficult to estimate primary production from standing stock phytoplankton biomass or, even more difficult, from chlorophyll-a. Still, these types of biomass measurements form the basis for many assessments and management decisions. Production measurements are done but are usually more limited in time and space. A model gives a possibility to test our knowledge on processes relating suspended sediment, light, primary production and phytoplankton standing stock.
More insight is needed in the precise effect of changes in turbidity (suspended sediment) and primary production in the Schelde. This report describes a sensitivity analysis, and presents results of a set of the construction of Deurganckdok, and two realistic scenarios in order to get a better insight in these effects.

1.1 Westerschelde primary production model

In 2019, an update water quality model, simulating the year 2014, was delivered. (Stolte and Schueder 2019). The model schematization and hydrodynamic model covers both the Schelde and Westerschelde (Figure 1.1). Water quality, phytoplankton biomass (chlfa) and primary production also cover the whole estuary but have only been validated for the Dutch part (Westerschelde).

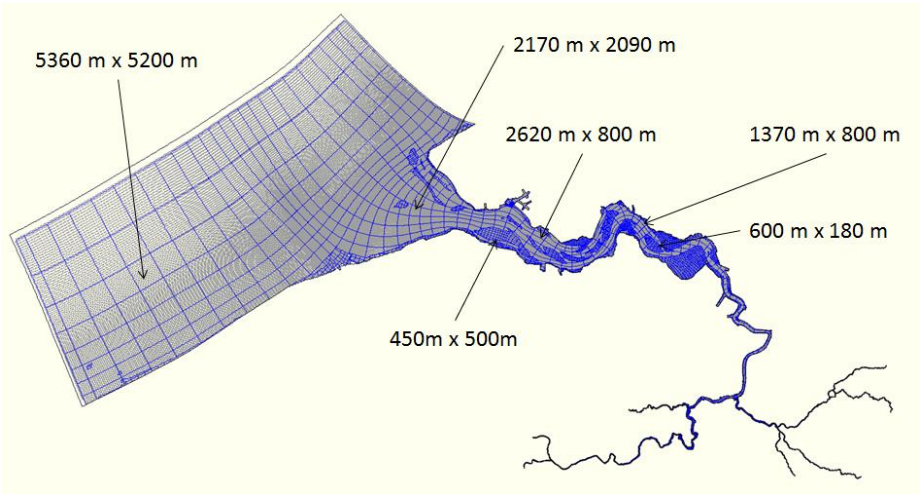

Figure 1.1.Grid and domain of the primary production and water quality model for the Schelde. The dimension of the aggregated grid cells is indicated. At shallow areas (< 5 m below mean sea level) the grid has a higher resolution.

In short, the model calculates primary production, and biomass of diatoms and non-diatoms using the DYNAMO subroutine (Monod growth kinetics). Suspended matter (SPM) is forced by using the output of separate sediment model runs. Grazing by zooplankton is simulated using measured maximum zooplankton biomass. The light extinction E was calculated using concentrations of SPM, DOC, chlorophyll-a and POC. Calibration was done by changing the mortality rates of phytoplankton (validation report). Phytoplankton biomass and production was reasonably well reproduced by the model (Stolte and Schueder 2019). The validation of the extinction coefficient (E) of K_d) remained however unsatisfactory.
More accurate relationship between SPM and available light is necessary. A first attempt is included in this report.

1.2 Objective

In this study we employ the Westerschelde model to:

- To test and improve our knowledge on the combined effects of hydrodynamics and sediment dynamics on primary production and phytoplankton biomass in the Schelde estuary, especially the Westerschelde.
- To evaluate the sensitivity of primary production and phytoplankton biomass to changes in environmental variables, such as concentration of nutrients and suspended sediment.
- To evaluate the effects of realistic measures or scenarios , such as variation in river discharge, or human activities, such as dredging, on primary production and biomass of phytoplankton

the report describes the results of two types of model runs:

1. Sensitivity analyses. Sensitivity of PP for variations in SPM (light) is important to predict changes due to human activities or measures that influence SPM concentrations.
2. Measures/scenarios. The validated 3D water quality model can be used to explore the effects of measures and/or scenarios on varying spatial and temporal scales. Three scenarios have been analysed.

2 Methods

As a baseline, the calibrated and validated model from 2019 was used. Some changes were made to the model formulations leading to a 2020 version of the model. This was done, because at careful inspection, it appeared that some formulations were not correct. The following changes were made:

- Denitrification in the water column was defined twice in two different processes. One of these processes has been switched off. This had no impact since denitrification did not occur in the water column.
- Nitrogen and phosphorus from dying phytoplankton was not allocated properly, and part of it disappeared from the system. This was fixed so that it now became part of the detritus pool. This led to higher concentrations of detritus in the 2020 model.
- A process converting the most labile form of detritus to a more stable form was switched on but was lacking the receiving state variable. This caused a mass balance failure and an underestimation of the detritus concentration. The process was switched off in the revised model. After fixing, this also led to higher concentrations of detritus.

To summarize, the 2020 model showed higher detritus concentrations. With the settings of the validated model, this led to a higher concentration of zooplankton grazers, which in its turn led to more grazing on phytoplankton, and subsequently lower phytoplankton biomass and production. After some test runs, the preference of zooplankton for detritus was set to 25% of the preference for phytoplankton (this was 100 % in the validated model, but detritus concentrations were much lower there), which led to very similar results as in the original validated model (Figure 2.1)

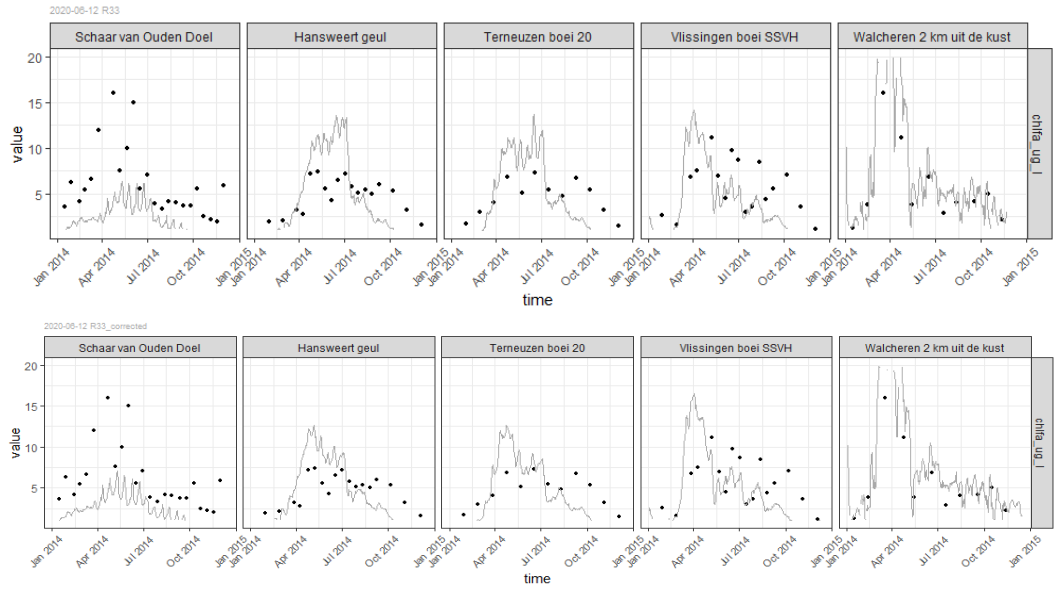

Figure 2.1 Model-data comparison for chlorophyll-A between the 2019 (top) and 2020 (bottom) reference runs.

2.1 Sensitivity analysis

The time and space varying SPM concentrations are applied to the model as a forcing function, generated by a specialized model setup (Deltares 2018). The results of this model are directly used in the water quality and primary production model.
Sensitivity analyses were done by applying decreased (-25 and -50 %) and increased (25 and 50 %) forcing functions of SPM to the model. For different areas in the model, the effect of these

changes on total net primary production was calculated and expressed as percent change as compared to the reference.

2.1.1 Measures/scenarios

The different scenarios that were considered are realistic changes as a result of human activities in the area. Scenarios that were chosen were:

1. Effect of siltation, dredging and dumping at Deurganckdok on primary production in the Westerschelde. The reference situation was a situation without any siltation, dredging, or dumping.
2. Effect of sediment concentrations from 2011, a year with a lower discharge over the year as compared to the reference (2014), and therefore also different concentrations of suspended sediment. Only the sediment concentrations from 2011 were used, hydrodynamics were used from 2014.
3. Effect of dredging at Terneuzen (reference). The suspended sediment in this scenario includes a proposed dredging and dumping scheme for planned activities. This situation was compared with the model simulation where Deurganckdock siltation, dredging and dumping is included.

*Table 2.1Overview of runs. *SPM source model codes refer to the codes used in*

Code	SPM source model	Characteristics of the SPM model	Objective
S03	A0*	baseline in DGD (Deurganckdok) model, equal to validated water quality model	serve as baseline for Deurganckdock and residence time scenario
S04	A5*	S03/A0 but with dredging at DGD	effect of siltation and subsequent dredging at DGD as compared to a situation without DGD
S05	B6*	Merge of S03 with hydrodynamics of B6 (2011).	Effect of 2011 TIM (caused by very different flow as compared to 2014) on primary production was evaluated.
S06		S03 with planned dredging at Terneuzen	Effect of planned dredging activities at Terneuzen

The evaluation of the results are given as time series at the main monitoring stations in the Western Scheldt or as yearly integrated rates per OMES area (Figure 2.2).

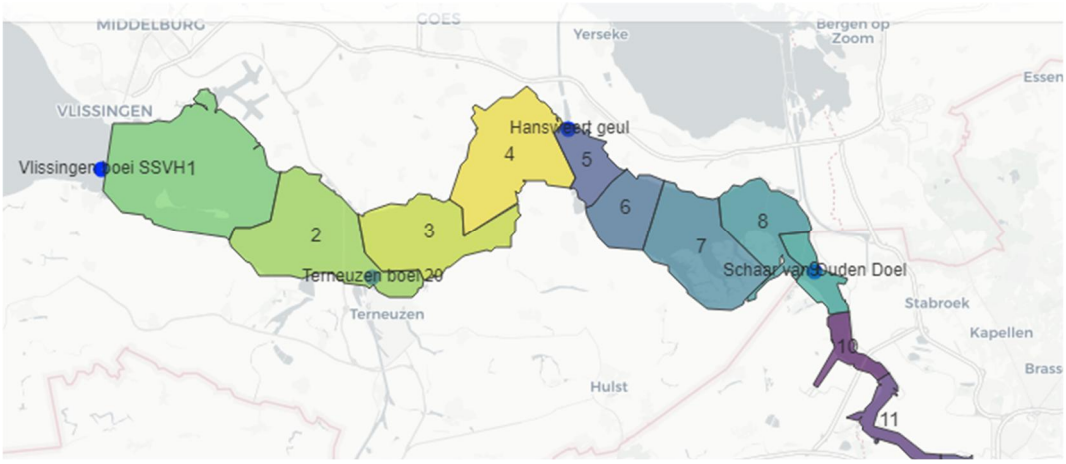

Figure 2.2 OMES areas of the mesohaline and polyhaline part of the estuary.

The results are presented as deviations from the baseline. Phytoplankton and zooplankton biomass changes are evaluated over space and time. The relative change in primary production per area (Figure 2.2) and the relative change in grazing loss per area is evaluated. The grazing module that is used in the model setup does not allow for calculating directly the secondary production. It can be assumed that changes in secondary production will be lower than changes in phytoplankton grazing losses due to respiration and maintenance costs for zooplankton.

3 Results

3.1 Sensitivity analysis

Varying the SPM concentration, model time series output at the main MWTL monitoring locations shows that a reduction of SPM leads to an earlier and higher spring bloom peak expressed as chlorophyll-A as compared to the reference situation. This increased spring bloom was seen at all four stations as expected (Figure 3.1). However, after the spring bloom, chlorophyll-A concentrations were lower using reduced SPM concentrations as compared to the reference Apparently, due to the higher spring bloom peak, zooplankton grazers could develop to a higher biomass (Figure 3.1, lower panel), and therefore a higher grazing pressure on phytoplankton. This caused a lower phytoplankton biomass after the spring bloom in simulations with reduced SPM as compared to the reference. The opposite was also observed. Higher SPM concentrations led to a lower spring bloom, but higher chlorophyll-A concentrations, due to a poorly developed zooplankton grazer population. The response of zooplankton to enhanced phytoplankton, and the effect of grazing on phytoplankton biomass after the spring bloom was strongest at the intermediate stations Hansweert Geul and Terneuzen.

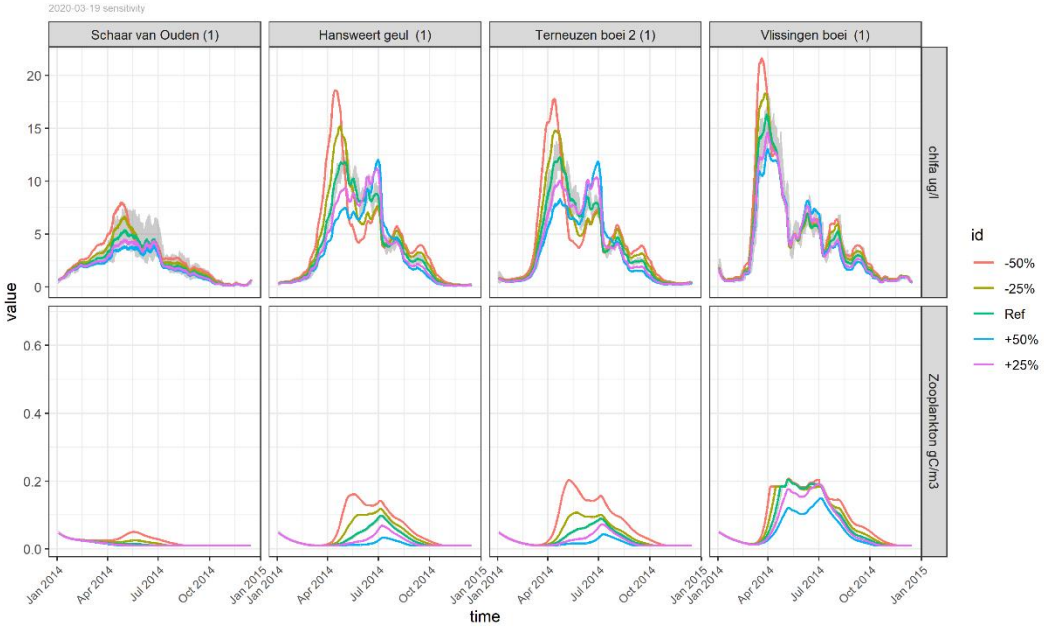

Figure 3.1 Modelled time series of chlorophyll-A in ug/l (top) and zooplankton biomass in mg C/l (bottom) for the year 2014 in the top layer at the main MWTL monitoring stations. The grey area are hourly outputs of the reference model. Results describe a situation where SPM is either reduced or increased with 25 and 50 % over the whole grid and year.

Daily variation of chlorophyll-A is mostly due to tidal driven transport and mixing in the estuary and is likely highest at locations with strong gradients. At Schaar van Ouden Doel, the daily variation of chlorophyll-A was highest (grey area in Figure 3.1). This indicates the strong gradient between relatively high biomass in the Western Scheldt, and low biomass in the very turbid and deep water of the lower Zeeschelde near Antwerpen.

Of the two phytoplankton groups modelled, diatoms reacted stronger to a change in SPM than non-diatoms. Since diatoms are superior competitors for light, providing silicate is available, this is in line with expectations. Diatoms also react strongest to the increase in zooplankton grazing because they are the preferred food source of zooplankton grazers in the model. In general, therefore, diatoms respond more to changes in the balance between light availability and grazer activity.

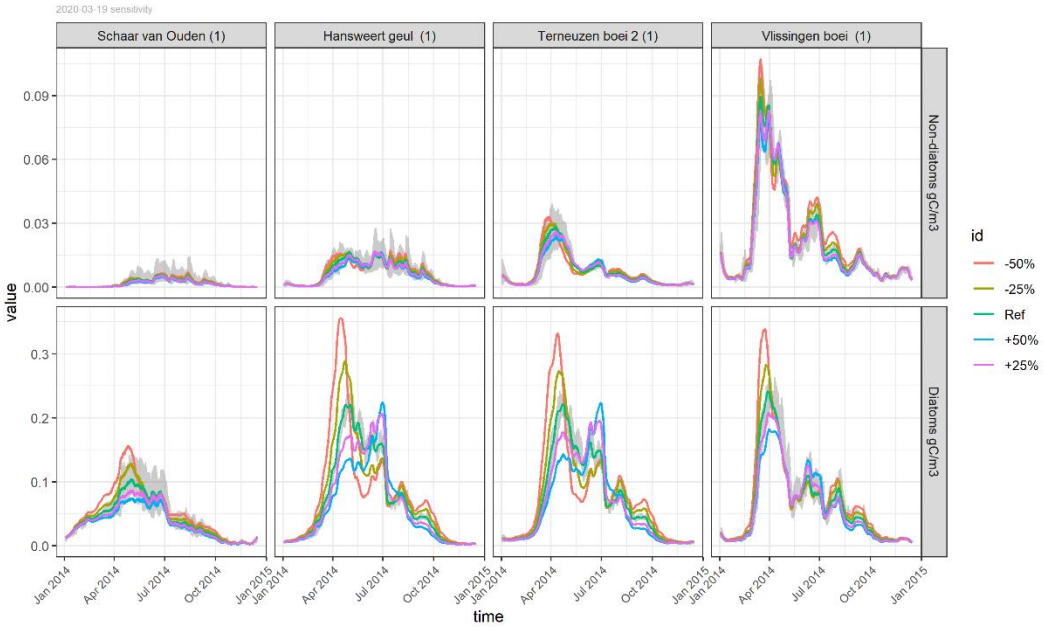

Figure 3.2. Modelled time series of non-diatoms (top) and diatoms (bottom) in gC/m^3 for the year 2014 in the top layer at the main MWTL monitoring stations. The grey area are hourly outputs of the reference model. Results describe a situation where SPM is either reduced or increased with 25 and 50 % over the whole grid and year.

Yearly integrated production per OMES area

The effect of increase and decrease of suspended particulate matter (SPM) was also expressed as a relative change in yearly integrated net primary production. Only results for OMES areas in the polyhaline and mesohaline part of the estuary are presented, because the model is validated in this part, and not further upstream. A lowered SPM resulted in an enhanced production, and vice versa (Figure 3.3). Qualitatively, this result is as expected, because light is the main limiting factor for primary production (Kromkamp and Peene 1995). However, quantitatively, the response is less strong than may be expected. Net primary production does not increase proportionally with the percentual increase or decrease of SPM concentration in most areas.

The relative change in phytoplankton grazing loss were calculated to be higher than the relative change in net production by the model. This indicates that an improved light climate due to reduced SPM concentration benefits zooplankton grazers more than proportional. It should be noticed that this result is likely dependent on the exact model formulations and parameterization of the zooplankton grazing process.

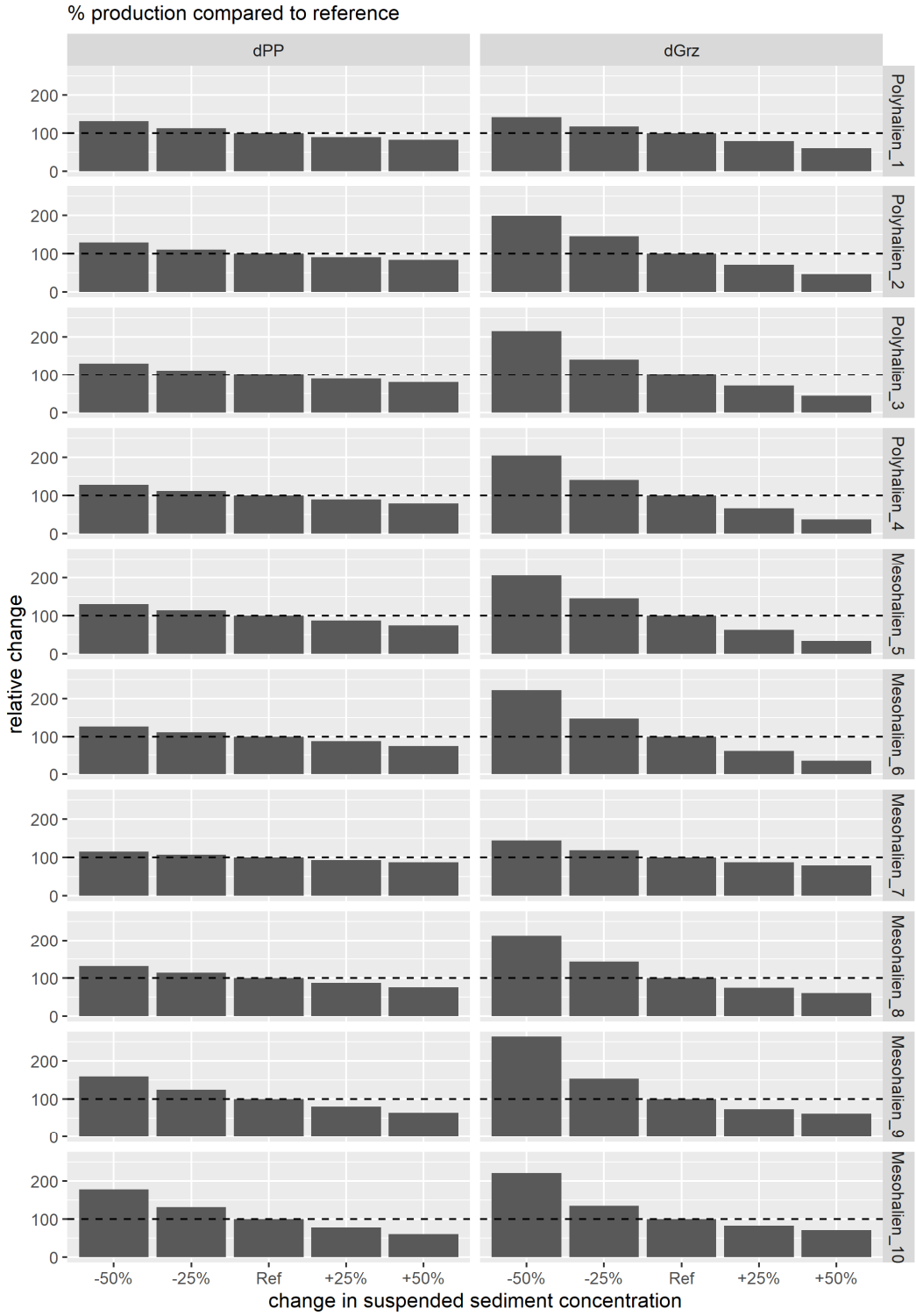

Figure 3.3 Percentual change of integrated net primary production (nPP, left panel) and lost due to grazing (dGrz, right panel) after an overall change in concentration of suspended particulate matter in the Schelde. Areas indicate OMES areas (see Figure 2.2)

3.2 Results from measures/scenarios

The scenarios presented here were chosen from existing sediment model runs . Thus, suspended sediment concentration results of the corresponding sediment models were directly used in the primary production model. The results are presented as deviations from their respective baseline,

in terms of biomass, primary production and grazing losses, for the different OMES areas in the mesohaline and polyhaline part of the Scheldt estuary.

3.2.1 Effect of dredging at Deurganckdok on primary production in the Westerschelde

In this calculation, the effect of siltation, dredging and dumping of sediment at Deurganckdok is compared to a situation where there is no dredging and, because of mass conservation, also no siltation at Dearganckdok (Lankriet and Cronin 2018).

The effect of dredging at Deurganckdok on suspended sediment has been modelled in a previous study using a dedicated sediment model (Lankriet and Cronin 2018). It appeared that, due to the sediment trapped in DGD, the suspended sediment concentration in the Westerschelde part of the estuary was lower than in a situation without siltation at DGD. The biggest change occurred in the mesohaline areas, where suspended sediment was reduced with concentrations up to 50 g/m3. This is explained by a combination of sediment trapping by DGD and tidal excursion of the water. The fact that suspended sediment is not reduced upstream of DGD is explained by the dumping of dredged materials from DGD upstream. Contrarily, and in line with expectations based on mass conservation, suspended sediment concentration upstream of DGD is higher compared to the reference as a result of the dredging and dumping.

3.2.2 Spatial variation

To illustrate the spatial variation of the effect, spatial model results are shown at one occasion (20th April, start of the spring bloom).

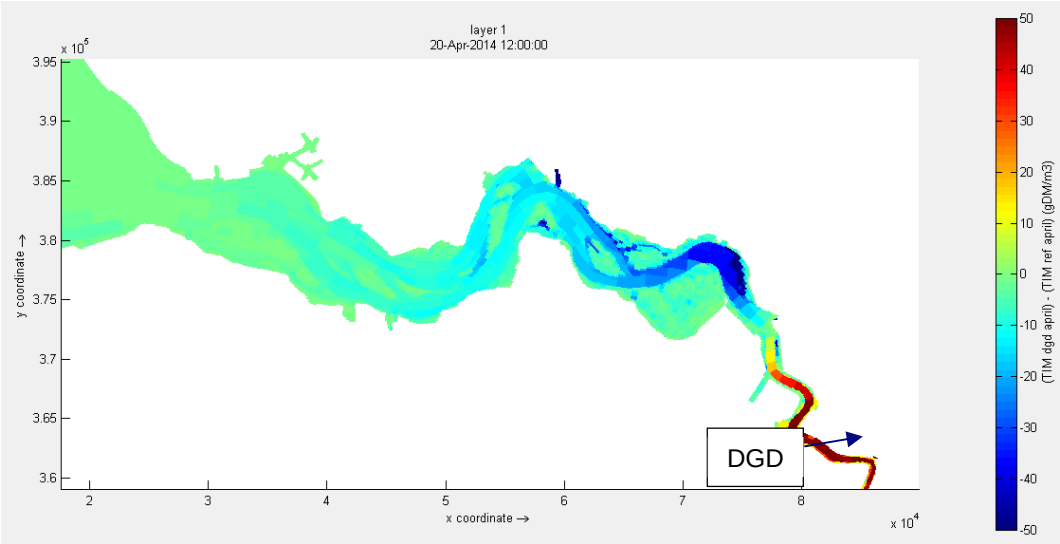

Figure 3.4 Change of suspended sediment concentration due to siltation, dredging and dumping at Deurganckdok as compared to a situation where there is no siltation (and therefore no dredging and dumping) at Deurganckdok. For details on the sediment model , see (Lankriet and Cronin 2018).

The effect of siltation, dredging and dumping at DGD on the spatial distribution of phytoplankton and primary production is shown in Figure 3.5. In April, at the onset of the spring bloom, an increase of 2 – 3 ug chlfa/l, as a proxy for phytoplankton biomass, is seen in the fairways of the mesohaline part of the estuary, stretching out to the polyhaline part. Near DGD, no change is observed probably due to the very dark conditions in that part of the estuary. Remarkable is the increase at Land van Saeftinghe. Although this increase looks impressive, the area over which is spread is a very shallow area, and the contribution to total biomass and production is therefore limited.

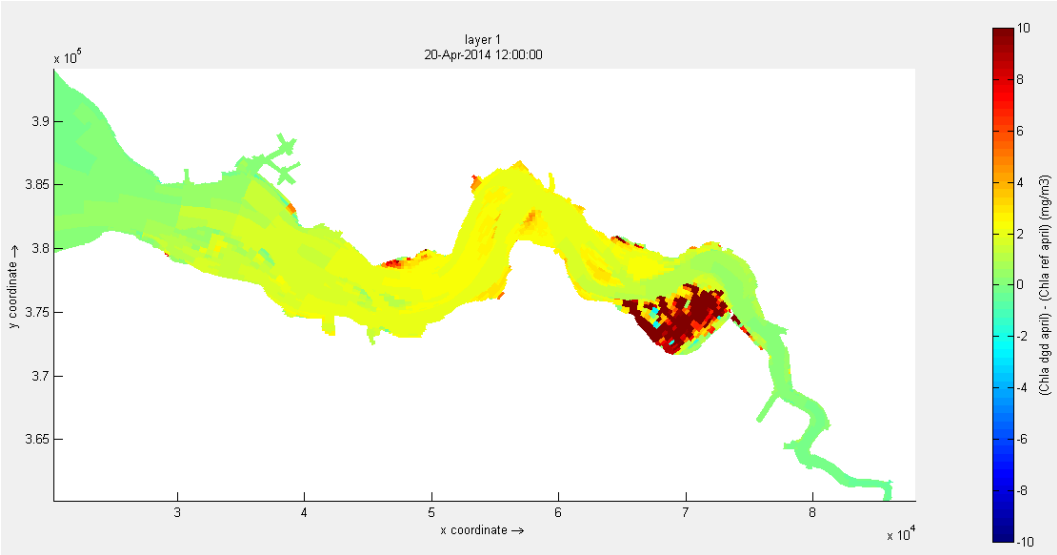

Figure 3.5Change of phytoplankton biomass, expressed as concentration of chlorophyll-A in ug/l in a situation with siltation, dredging and dumping at Deurganckdok relative? to a situation where there is no siltation (and therefore no dredging and dumping) at Deurganckdok. For details on the sediment model , see (Lankriet and Cronin 2018).

3.3 Temporal variation

The decrease of suspended sediment due to Deurganckdok (DGD) is not equal over time (Lankriet and Cronin 2018). In short, the differences were (Figure 3.6):

- The decrease is stronger during winter and early spring, when also concentrations of suspended sediment are highest
- The effect of DGD is most pronounced at Schaar van Ouden Doel. This is the location closest to DGD, but also the station with highest suspended sediment concentrations.
- The effect of DGD on suspended sediment concentration is almost not noticeable at Vlissingen, located at the mouth of the Westerschelde.

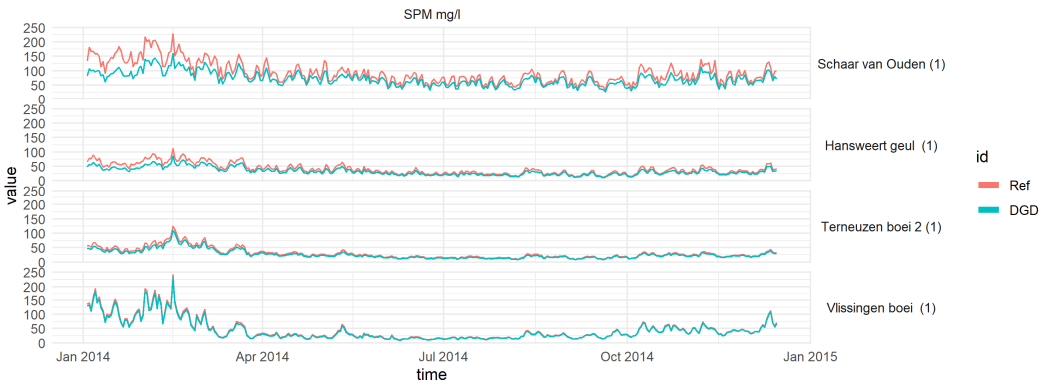

Figure 3.6 Suspended sediments in mg/l, modelled for the year 2014 in a model schematization allowing for siltation, dredging and dumping at Deurganckdok (DGD, blue), as compared to a reference without siltation, dredging and dumping (Ref, red).

The temporal effect of DGD on concentrations of chlorophyll-A also differs per station (Figure 3.7). In early spring, when the largest decrease of suspended sediment was observed in the situation with DGD in place, concentrations of chlorophyll-A are still very low, due to low light availability. Consequently, no effect is observed. During the rest of the year, hardly any effect is seen at Schaar van Ouden Doel, presumably due to the very dark conditions and therefore low phytoplankton biomass at that location. At location Vlissingen, the effect is also very small,

presumably because this location is too far away (see also Figure 3.6 for effect on suspended sediment). At the locations in between, Hansweert and Terneuzen, the concentrations of chlorophyll-A increased during the period end of March to end of June. From end of June, the concentration of chlorophyll-A decreased as compared to the reference. The decrease as compared to the reference cannot be explained as a direct effect of changes in suspended sediment concentrations.

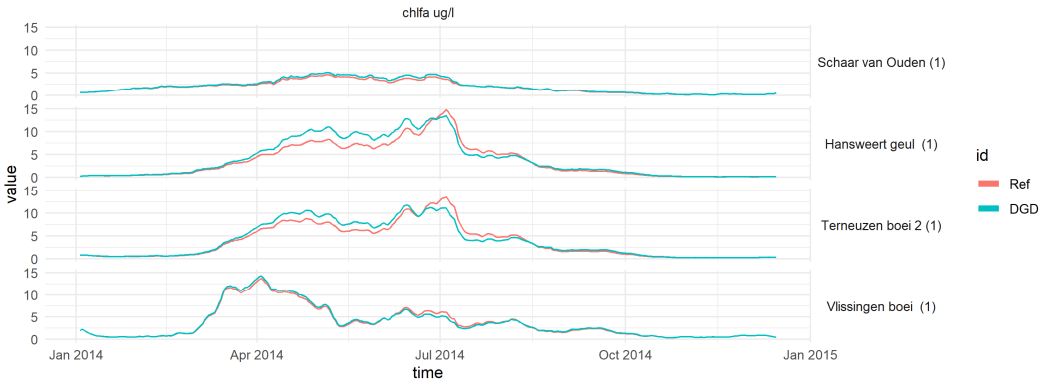

Figure 3.7 Chlorophyll-A, as a proxy for phytoplankton biomass in ug/l, modelled for the year 2014 in a model schematization allowing for siltation, dredging and dumping at Deurganckdok (DGD, blue), as compared to a reference without siltation, dredging and dumping (Ref, red).

Presumably as a result of increased phytoplankton biomass, zooplankton concentrations increased during spring/early summer at Hansweert and Terneuzen as compared to the reference (Figure 3.8). This increase was substantial and could well account for the decrease of phytoplankton (chlorophyll-A) during summer as compared to the reference run because of a higher grazing pressure.

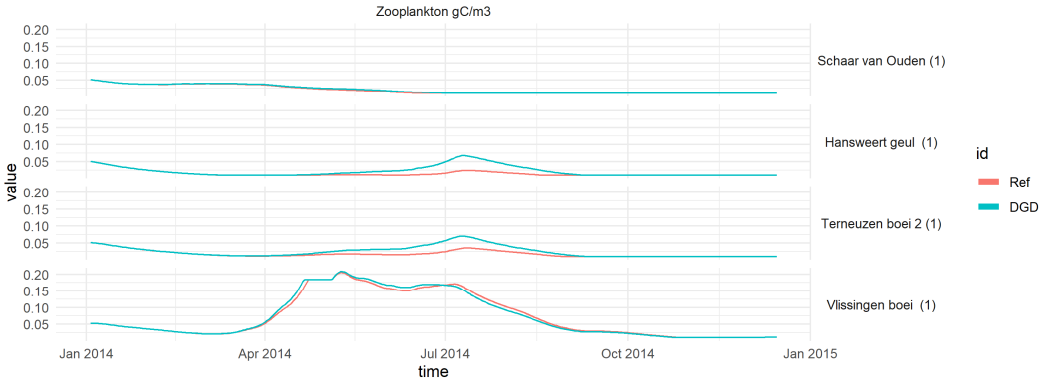

Figure 3.8 Zooplankton grazer biomass, mgC/l, modelled for the year 2014 in a model schematization allowing for siltation, dredging and dumping at Deurganckdok (DGD, blue), as compared to a reference without siltation, dredging and dumping (Ref, red). Locations compare to the top layer of segments where the respective MWTL monitoring stations are located.

3.4 Change in yearly integrated production

Integrated over the whole year, net primary production increased in the run with DGD in place, about 10 – 20 %, and up to 25 % at mesohaline 9, as compared to the reference. Similar to the increase in chlorophyll-A concentrations, the primary production increase was highest close to DGD, except at “mesohaline 10” where production is strongly light limited due to extremely high turbidity.

As expected, the relative increase of biomass concentration at Land van Saeftinghe Figure 3.5 in area “mesohaline 7” did not result in an exceptional increase of the yearly production.
The relative increase in production was largest in the areas immediately downstream Antwerp (mesohaline areas 9-10). Suspended sediment concentration mostly decreased in that area due to siltation in DGD, mixing and downstream transport. However, still at the mouth of the estuary (polyhaline 1), a slight relative increase of production was observed. Indeed, reduced suspended sediment concentrations in the situation with DGD stretch to this area. The relative low increase in the intermediate areas (e.g. mesohaline 7 and 8) are perhaps explained by the morphology and hydrodynamics. This area combines relatively deep fairways with small areas of tidal flats as compared to the more downstream areas.
Yearly integrated zooplankton grazing increased even more (10-75 %) than the net phytoplankton production rates in some areas. In the current model, grazers have a relatively high maximum biomass capacity, which makes them react quickly to any change in phytoplankton biomass. Also, the pattern of increase is different than the increase in primary production. The relative increase in grazing was found at the centrally located areas of the Westerschelde.

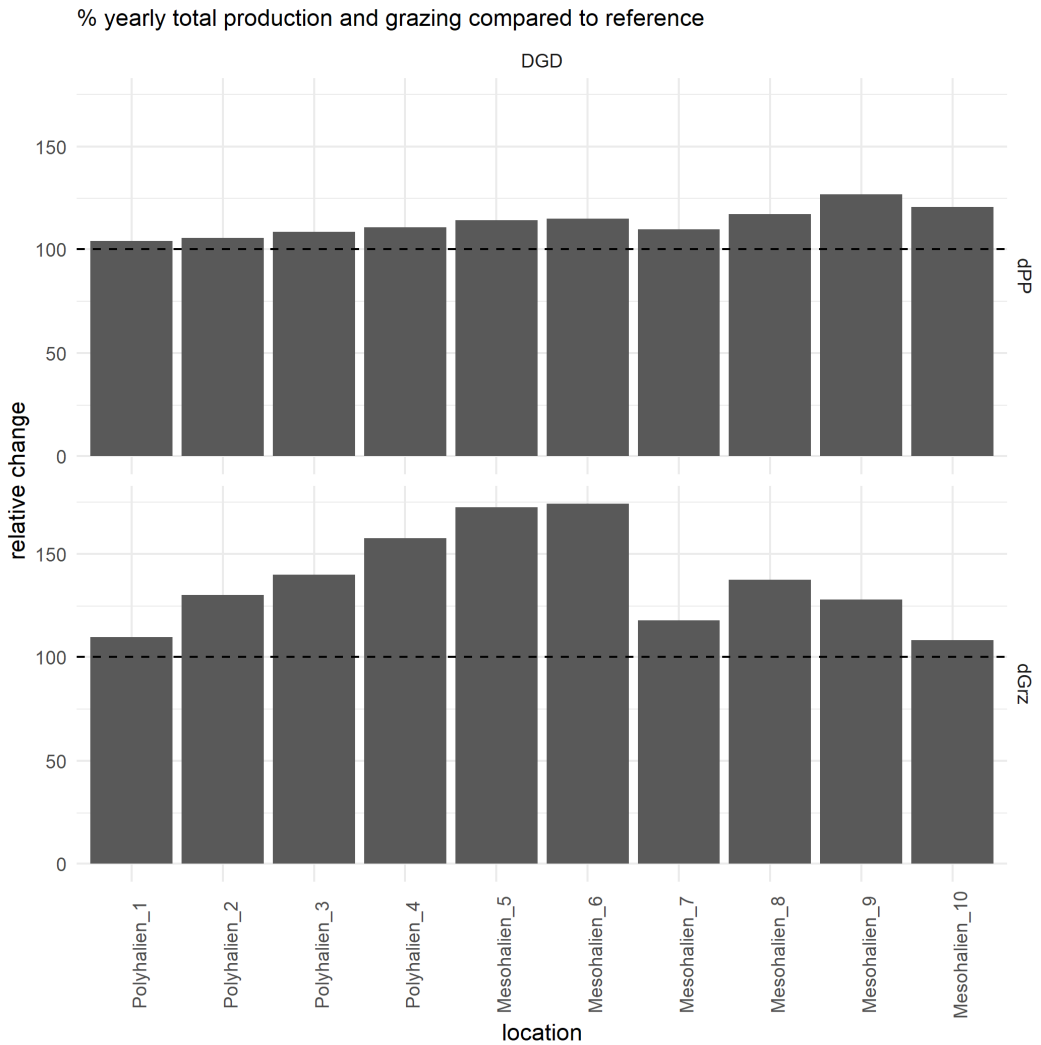

Figure 3.9 Yearly integrated relative change in primary production (dPP) and losses through grazing (dGrz) in different OMES areas. Primary production and grazing are expressed as % change in the DGD runas compared to the value of the reference model run. For location of the areas, see Figure 2.2.

3.5 Effect of suspended sediment concentrations in a low discharge year (2011)

This scenario is using the suspended sediment concentrations that were calculated based on a sediment model simulation using the hydrodynamics, meteorology and boundary conditions of the year 2011. In the water quality and primary production model, only the suspended sediment concentrations were used, while hydrodynamics, meteo and boundary conditions of 2014 were applied.

3.6 Spatial variation

The year 2011 differed from 2014 because discharge in 2011 was substantially lower during the winter. In the summer the discharge in 2011 was higher than in 2014. This resulted in an on average lower SPM concentration throughout the year. Around mid-April, SPM concentrations were reduced at most with around 50 g/m3 just downstream the Belgian-Dutch border. Upstream and downstream, the effect was much less, but still resulting in a decrease of around 10-15 g/m3 in most of the Westerschelde.

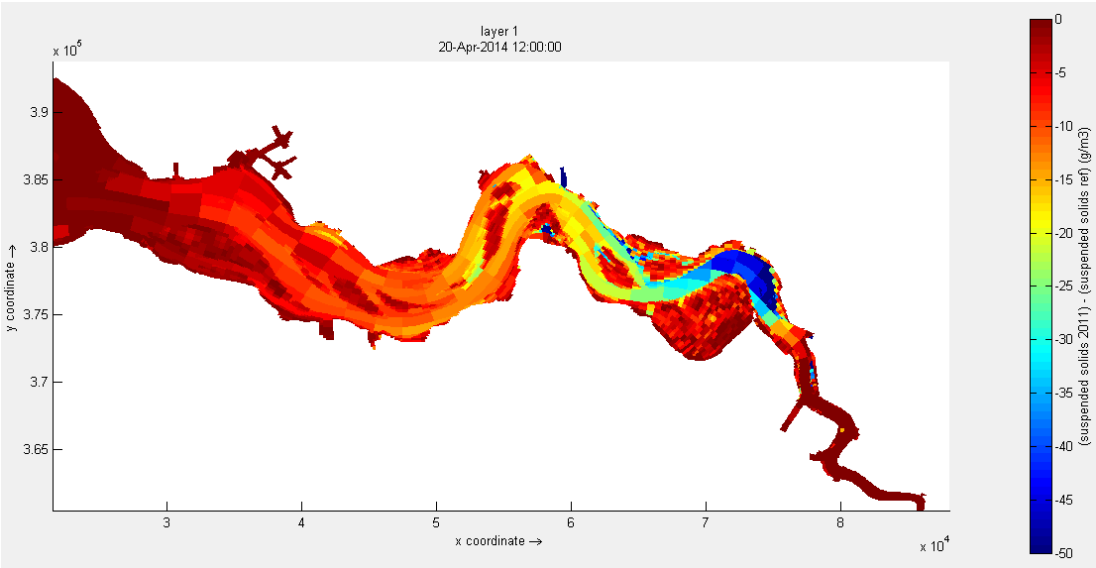

Figure 3.10 Change of suspended sediment concentration in case 2011 suspended particulate matter concentrations are applied as compared to the 2014 situation. For details on the model scenario, see (Lankriet and Cronin 2018).

The reduced SPM concentration resulted in an increase of chlorophyll-A. The increase was most pronounced in the middle of the Westerschelde, close to the MWTL stations Hansweert and Terneuzen. As in the previous scenario, the large increase of chlorophyll-A at Land van Saeftinghe has very little effect on the total production in the area, because this area is very shallow, and often dry.

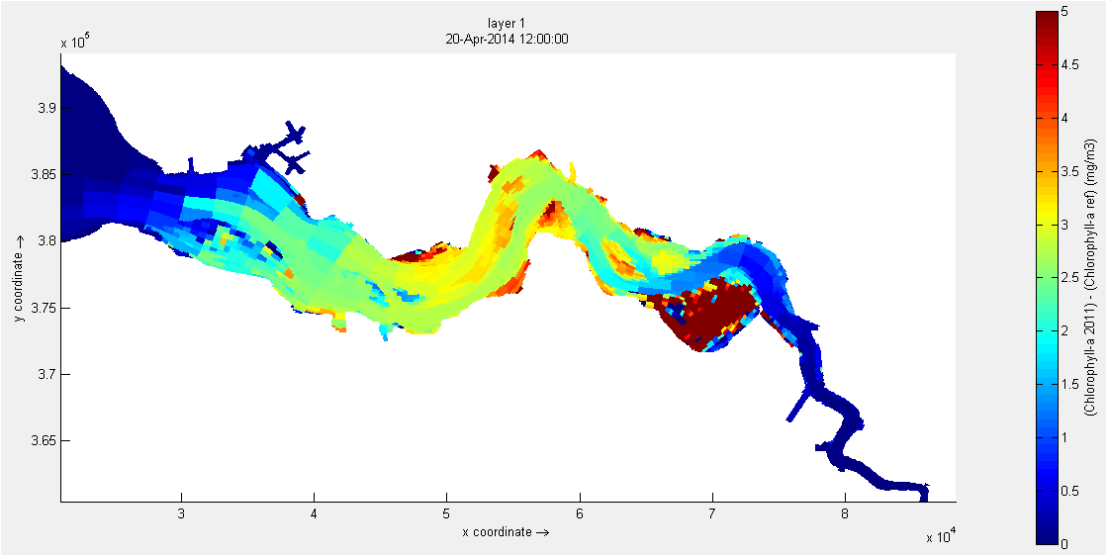

Figure 3.11 Change of chlorophyll-A concentration in case 2011 suspended particulate matter concentrations are applied as compared to the 2014 situation. For details on the model scenario, see (Lankriet and Cronin 2018).

3.7 Temporal variation

Over the whole year, SPM concentrations in the 2011 scenario were lower than the reference, and the difference was highest in the upstream location Schaar van Ouden Doel (Figure 3.12).

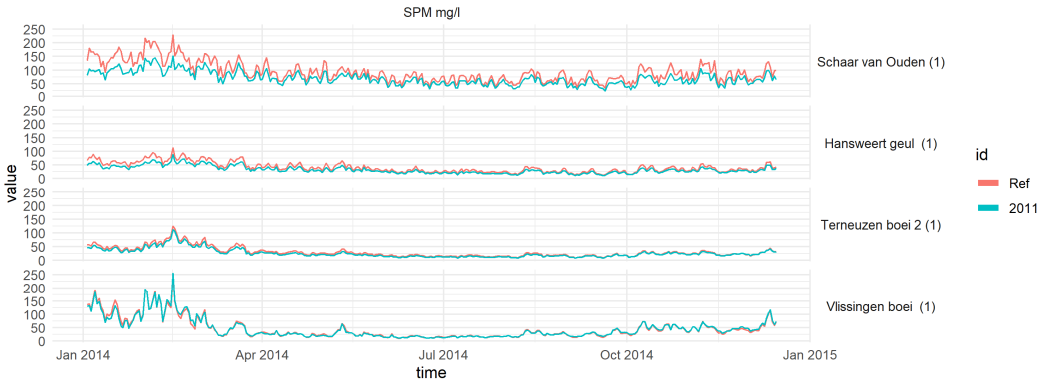

Figure 3.12 Suspended sediments in mg/l, modelled for the hydrodynamic year 2014 in a model schematization using SPM concentration from 2011 (2011, blue), as compared to a reference (Ref, red).

The concentration of chlorophyll-A was higher in the 2011 scenario than in the reference. This can be explained by the lower SPM concentrations. Although the SPM was most influenced at Schaar van Ouden Doel, it did not result in higher chlorophyll-A concentrations at that location. This is since it is too dark and too deep at this station and directly upstream of it. Largest effects are therefore seen at the intermediate stations Hansweert and Terneuzen (Figure 3.13).

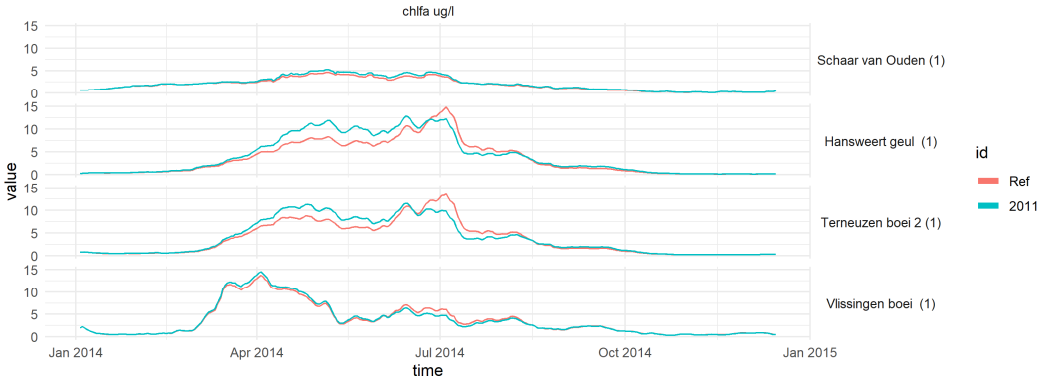

Figure 3.13 Chlorophyll-A in ug/l, modelled for the hydrodynamic year 2014 in a model schematization using SPM concentration from 2011 (2011, blue), as compared to a reference (Ref, red).

Due to the higher biomass of phytoplankton in spring, zooplankton grazers could grow to higher biomass in that time (Figure 3.14), which caused an increased grazing during summer, and therefore lower phytoplankton biomass (Figure 3.13 and Figure 3.14).

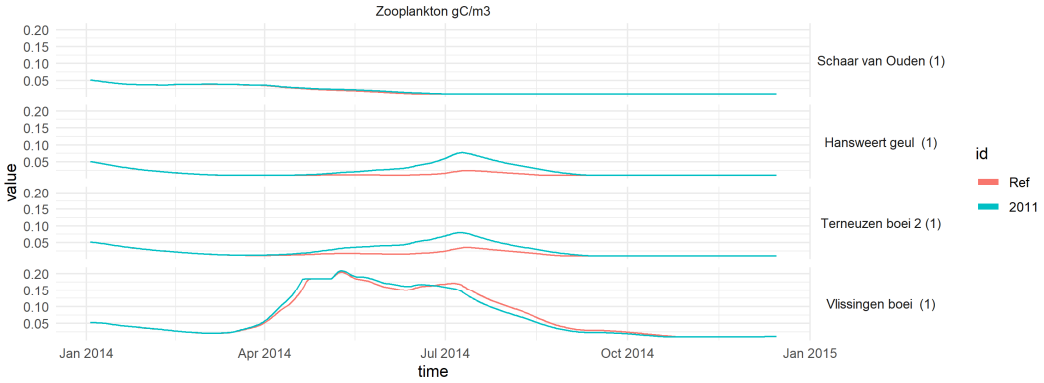

Figure 3.14 Zooplankton biomass in gC/m³, modelled for the hydrodynamic year 2014 in a model schematization using SPM concentration from 2011 (2011, blue), as compared to a reference (Ref, red).

The total yearly primary production of the 2011 scenario as compared to the reference was enhanced by up to 40 % depending on the area. Losses due to grazing were in some areas increased by 100 %. In this simulation, therefore, the trophic transfer of energy from primary production was almost doubled in some areas, especially in mesohaline 5 and 6, which is in the middle part of the Westerschelde. Close to the Belgian border, in mesohaline 9 and 10, a smaller increase of grazing was seen. At the mouth, the relative increase of grazing was again higher. Overall, the pattern of relative change of production and grazing as a result of low discharge closely resembles the previous scenario, the effect of siltation at Deurganckdok, but is stronger.

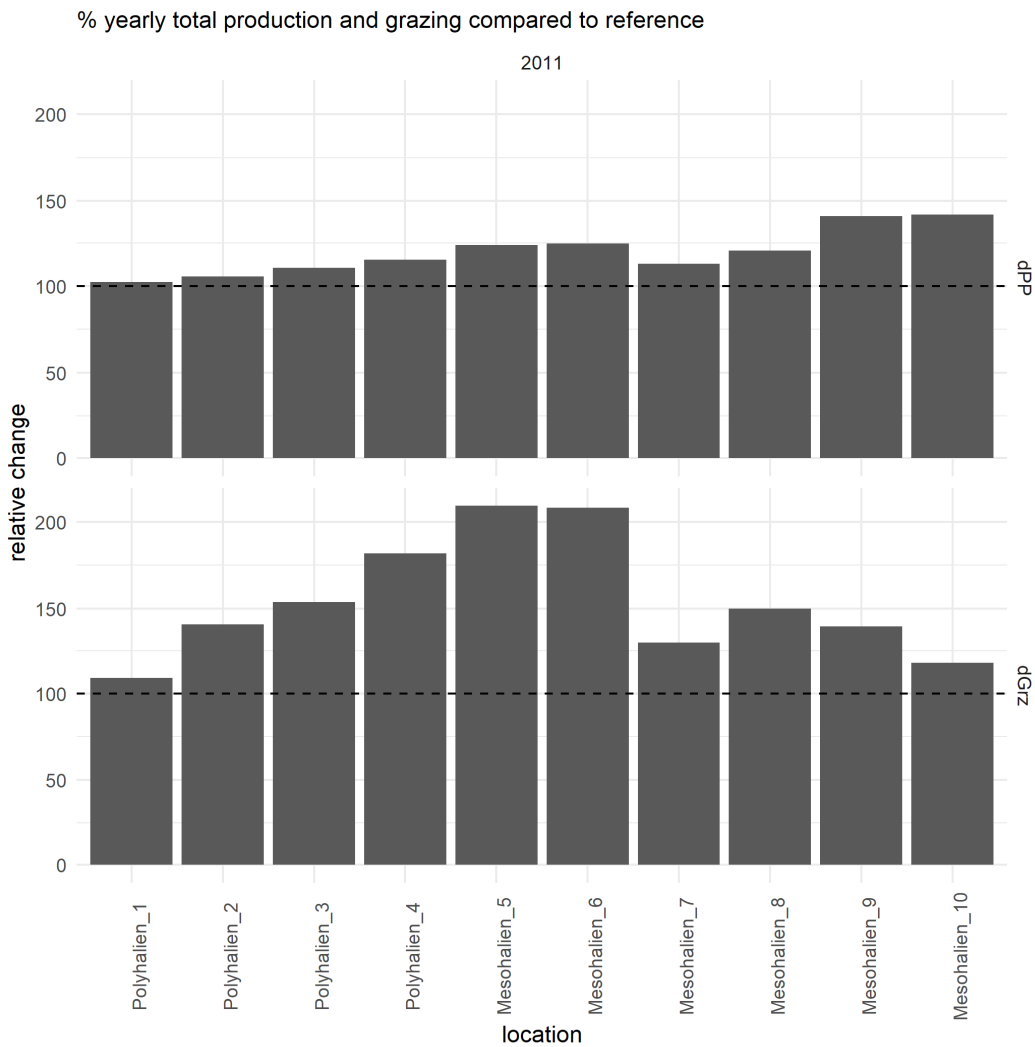

Figure 3.15 Yearly integrated relative change in primary production (dPP) and losses through grazing (dGrz) for diatoms (red) and non-diatoms (blue) in different OMES areas. Primary production and grazing are expressed as % change in the 2011 scenario as compared to the value of the reference model run (2014). For location of the areas, see Figure 2.2

3.8 Effect of dredging and dumping at Terneuzen on primary production in the Westerschelde

The basis of this scenario uses suspended sediment concentrations that were calculated using the hydrodynamics, meteorology and boundary conditions for 2014, including the siltation, dredging and dumping at Deurganckdock (the actual situation). The scenario includes extra dredging and dumping as proposed in the construction works for the new sluices at Terneuzen (planned for 2021). In this model scenario, the dredged material is dumped in the Westerschelde near Terneuzen during the winter months, as this was expected to have minimal impact on primary production.

Indeed, the increase of SPM was only seen in the months until April (Figure 3.16). Overall, the effect of dredging on the total concentration of suspended matter was small. As expected, the difference is strongest at Terneuzen.

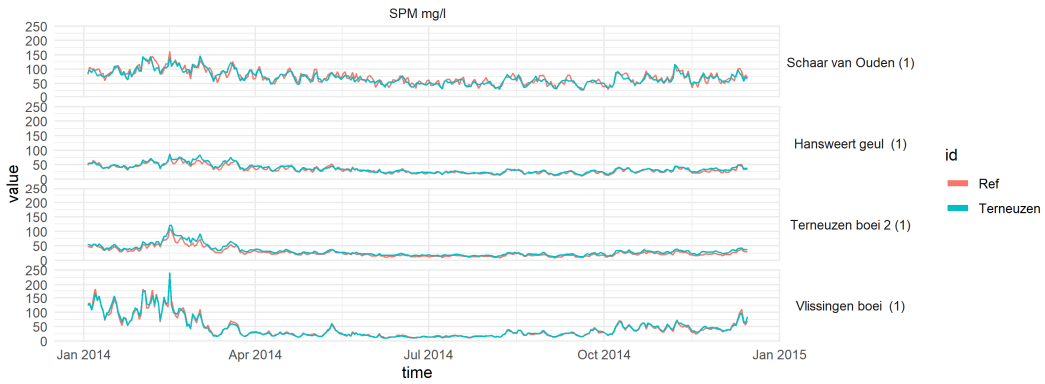

Figure 3.16 Suspended sediments in mg/l, modelled for the hydrodynamic year 2014 in a model schematization using SPM concentration from a scenario with extra dredging and dumping at Terneuzen (terneuzen, blue), as compared to a reference (Ref, red).

Although the effect of SPM was very limited, still some effect of the dredging and dumping is seen on the phytoplankton biomass as chlorophyll-A in the middle part of the Westerschelde. In spring, chlorophyll-A concentration was slightly reduced as compared to the reference at the locations Hansweert and Terneuzen. At the other locations, no difference is noticeable by visual inspection. In summer, phytoplankton biomass is slightly elevated as compared to the reference. This is caused by a lower grazing pressure by zooplankton (Figure 3.18).

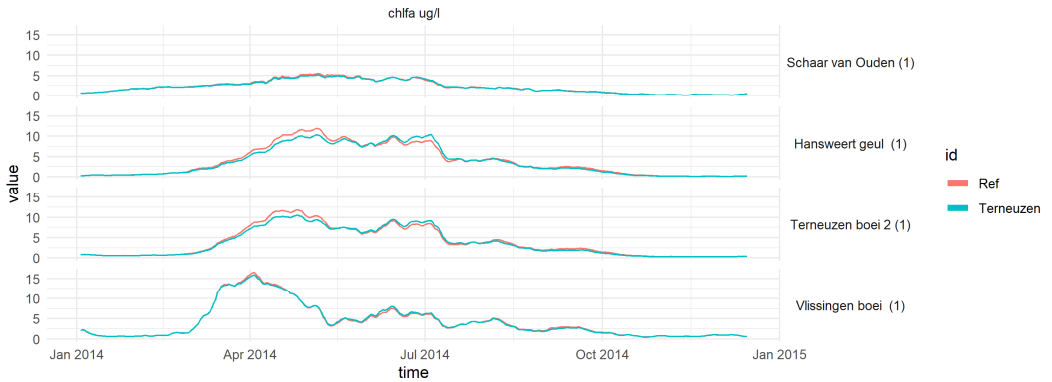

Figure 3.17 Chlorophyll-A in ug/l, modelled for the hydrodynamic year 2014 in a model schematization using SPM concentration from a scenario with extra dredging and dumping at Terneuzen (terneuzen, blue), as compared to a reference (Ref, red).

Due to reduced phytoplankton during spring as compared to the reference, zooplankton biomass is also reduced slightly due to food limitation. At Vlissingen, at the mouth of the estuary, a slightly increased zooplankton biomass is visible during summer. This is probably caused by enhanced phytoplankton biomass during summer in the estuary. At Hansweert and Terneuzen, this is not observed. Possibly this is because zooplankton biomass was reduced more at those stations than in Vlissingen, and for a longer period of time.

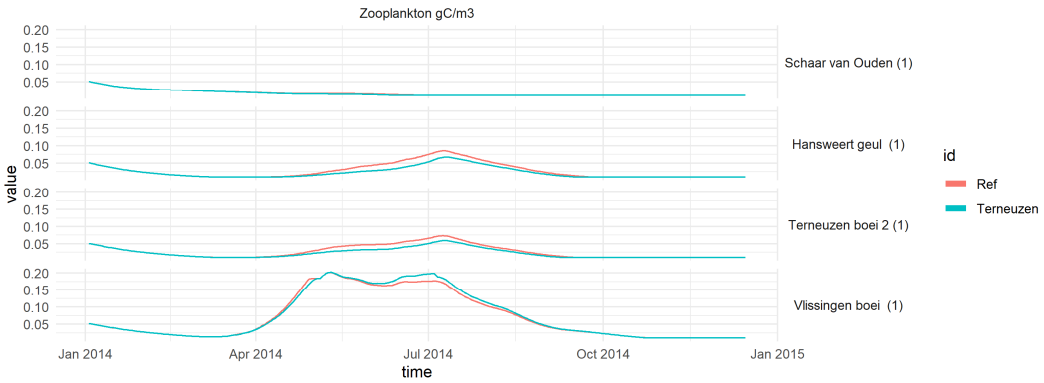

Figure 3.18 Zooplankton biomass in mgC/l, modelled for the hydrodynamic year 2014 in a model schematization using SPM concentration from a scenario with extra dredging and dumping at Terneuzen (terneuzen, blue), as compared to a reference (Ref, red).

The relative change of yearly production by diatoms was slightly decreased as compared to the reference in areas 1,2,3 and 4 by up to 10 %. More upstream, the pattern is less clear and very little effect is observed. Grazing losses for diatoms were reduced up to 20 %, and also extended to a wider area. This is due to dispersion of the zooplankton grazer population. Both production and grazing of non-diatoms (greens) were reduced more, up to 50% and 60% respectively.

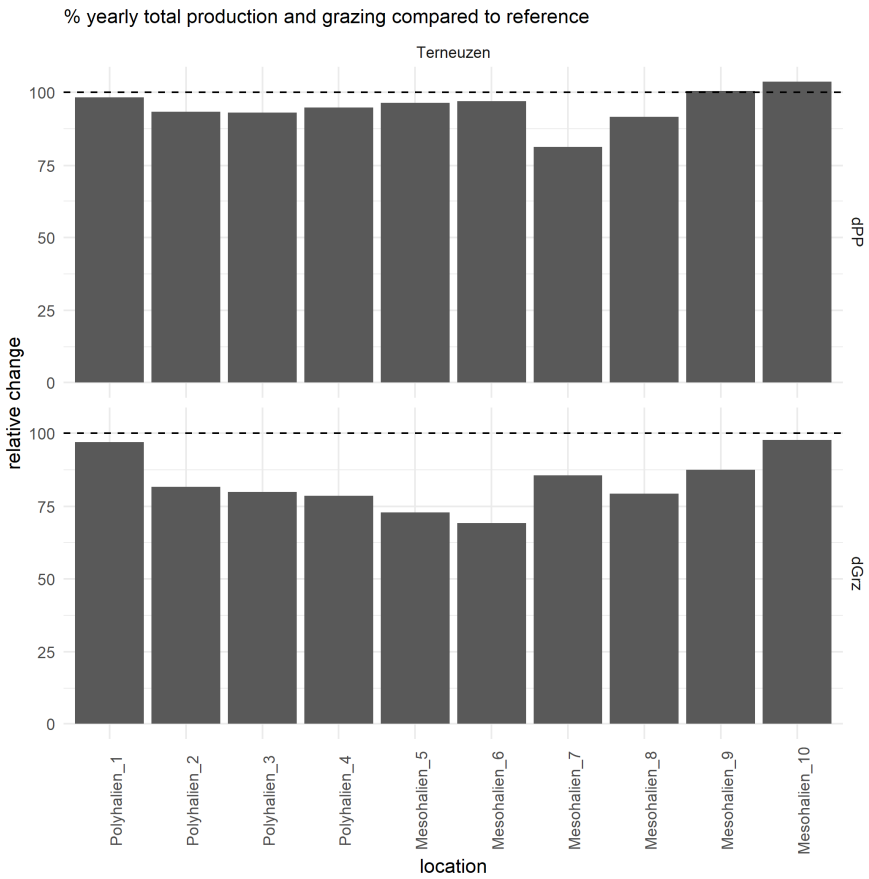

Figure 3.19 Yearly integrated relative change in primary production (dPP) and losses through grazing (dGrz) OMES areas. Primary production and grazing are expressed as % change in the scenario with included dredging and dumping at Terneuzen, as compared to the value of the reference model run. For location of the areas, see Figure 2.2

4 Discussion

All scenarios in this study are based on changes in suspended particulate matter (SPM). This choice has been made because:

- SPM has been identified as the main contributor to the extinction of light available for photosynthesis in the Schelde and Westerschelde (Kromkamp & Peene 1995, Cloern 1999).
- Light has been identified as the main limiting factor for primary production
- Primary production is likely to be the limiting factor for production of higher trophic levels in the Westerschelde ecosystem.

A summary scheme of the main processes regarding to the scenarios is depicted in Figure 3.13

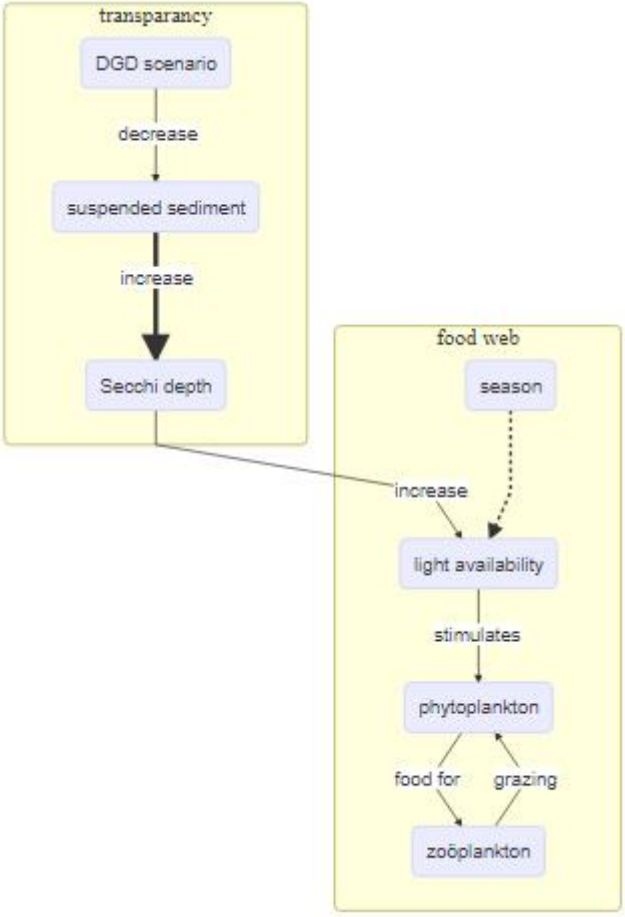

Figure 4.1Schematic representation of main processes depicting a decrease of suspended sediment concentration, as is the case in the Westerschelde part of the estuary in the DGD scenario.

4.1 Sensitivity of primary production to suspended particulate matter

The sensitivity of phytoplankton net production to changes in SPM are non-proportional. A 50 % reduction does not result in a doubling of net production. Reasons for this are amongst others:

- Depth-integrated production depends highly on the depth of the euphotic depth as compared to the mixing depth. If this ratio is low, reduction of SPM will only result in a small increase of primary production e.g. in mesohaline 8 and 9.
- SPM is not the only substance responsible for light extinction in the water column. DOC, and to a lesser extent chlorophyll-A also plays a role. DOC decreases proportional with salinity. The contribution of SPM varies therefore per area.
- An increase in production will also lead to an increase in grazing. The immediate (grazing) and lagged (growth, mortality) response of grazers to variations in available phytoplankton is called a trophic cascade (Maar et al. 2018; Carpenter et al. 2008). In this case, the availability of phytoplankton food is a regulating factor for grazing pressure, thereby damping the effect of changes in light availability on primary production
- The Schelde is strongly mixed by tidal currents. The observed effects are combinations of changes in growth, loss factors and transport/mixing processes. Therefore, the observed changes in one area are (partly) also the result of changes in processes elsewhere.

4.2 Suitability of the model for scenario calculations

The 3D model schematization of water quality processes and primary production in the Scheldt used in this study calculates the biomass and production of phytoplankton as a function of available light, nutrients and potential zooplankton grazer capacity. As concluded earlier (Stolte and Schueder 2019), nutrients are almost never limiting for phytoplankton growth and production in the Westerschelde. In the current situation, primary production and the biomass of phytoplankton is completely dependent on available light, which is regulated by the ratio of photic depth and mixing depth (total depth). The variation in photic depth in the estuary is strongly influenced by suspended particulate matter. The model takes all this into account, by combining the essential processes, including light attenuation, photosynthesis, grazing, and remineralization, with advective transport and mixing from a full scale hydrodynamic model using a fine gridded bathymetry (Stolte and Schueder 2019).
Although the schematization also covers the Zeeschelde and rivers, the model is currently only validated for the Westerschelde part of the estuary. Therefore, the model is less suitable for scenarios that involve water quality or primary production located upstream of Antwerpen.

4.2.1 Role of zooplankton grazing

In the scenarios, zooplankton grazing increases more than primary production and algal biomass. In other words, in the model, phytoplankton is controlled by zooplankton grazing, at least in some areas. This is consistent with findings in earlier studies (Maris, Oosterlee, and Meire 2010).
How much of the increased production is actually transferred to zooplankton and higher trophic levels depends on many more factors than can be taken into account in the current model. One underlying assumption in the current model is that the potential maximum grazer biomass is defined as a time function based on monitoring data, but spatially constant. The actually modelled zooplankton biomass is determined by the availability of food (phytoplankton), but will not exceed the maximum grazer biomass as specified in the input. In the runs presented in this report, this maximum grazer biomass was not reached, and therefore, grazers could easily profit from an increase in phytoplankton production.
The strong response of the grazing population in the model is an indication of the transfer of energy and carbon to the rest of the food web. A relatively small decrease of SPM, and a corresponding relatively small increase of phytoplankton biomass can thus have a large relative effect of energy transfer in the food web. This has been suggested earlier by (Maris, Oosterlee, and Meire 2010). In the Westerschelde and especially close to the North Sea, zooplankton drifted into the Westerschelde from coastal waters where they were dying, rather than grazing/producing (Soetaert and Herman 1994). However, conditions for both phyto- and zooplankton in the

upstream part of the Scheldt have improved since then, and grazing in the Westerschelde may not be dependent anymore on grazers drifting in from the coastal zone.
In the current model, the maximum grazing biomass is defined as a time series based on monitoring data (process CONSBL, Deltares 2016). Grazing rates were applied so that the phytoplankton biomass that developed in the model did not saturate grazing capacity and grazer biomass. This way, there remained potential for the grazer biomass to respond to increased production and biomass, which makes it possible for zooplankton to control phytoplankton. A dynamic zooplankton is also available within the Delft3D package, namely Dynamic Energy Budgets (DEB) zooplankton, but it would require more effort to parameterize. Currently it is unclear whether an alternative would provide a more realistic estimation of zooplankton biomass and grazing rates. But, because the phytoplankton biomass seems to be controlled very strongly by grazing in the model, further research on the implementation of phytoplankton grazing is desirable.

4.3 Conclusions and recommendations

The conclusions of the current research are:

1. Changes in suspended material have most effect in the central part of the Westerschelde.
2. The effect on primary production is transferred effectively to zooplankton production. According to the current assumptions, zooplankton production reacts even stronger on changes in suspended matter than phytoplankton production.
3. Dumping of dredged material in winter/early spring has very low impact on primary production by phytoplankton.

Dumping of dredged material in the turbidity maximum (close to and upstream Antwerpen) has low impact on the primary productivity in that zone. However, the turbidity maximum may shift in place depending on tidal excursion and mostly river discharge.

4.4 Proposal for further research I: Better quantification of water column light attenuation for improved modelling of primary production.

A crucial role in the prediction of primary production as related to observed or projected changes in suspended material is the estimation of its effect on light attenuation in the water column. The currently calculated extinction coefficient underestimates at low values of E (low concentrations of DOC and SPM) and overestimates E at very high values of E (Stolte and Schueder, 2019). There is room for improvement.
In the current model, an empirical model is used taking into account extinction by SPM, DOC, Chlorophyll-A and detritus. The measured extinction coefficient (E, which would be a K in optics notation) was related to other regularly measured water quality parameters for Westerschelde and an estimate of the coefficients was made by multiple regression:
$E = a + b*SPM + c*DOC + d*CHLFA$
The values for the used constants were determined using multiple regression on Westerschelde data (Stolte and Schueder, 2019), which were transferred into the model.
Improvements may be found when considering extinction from a marine optics point of view. In the appendix A, it is illustrated how marine optics can contribute to improve the estimation of underwater light availability, which results in following recommendations.
Recommendation 1: Investigate the possibility to deploy this equipment, e.g., acs and BB3 or LISST-VSF in a true monitoring setup.
Recommendation 2: Take measurements using both Kd protocols and compare results. Good optical measurements (of e.g. Irradiance, Ruddick et al., 2019) are scarce and needed to calculate or validate Kd. Save and store also the original measurements.
Recommendation 3: The Gons et al. algorithm can be tested for Westerschelde, or reconstructed when optical measurements are available, the QAA algorithm can be tested on in situ spectra.

Recommendation 4: Test the approach as outlined above and consider redoing the RT modelling chain if e.g. implementation in the biogeochemical-ecological modelling is the ambition. Note that KdPAR (and probably also E_0) can then be derived from HydroLight.

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A Radiative transfer, the physics behind marine optics

The propagation of radiation (light) through water is affected by absorption and scattering processes. Even in the simplest case of horizontally homogeneous water and time independence, the radiance distribution is a function of four variables: depth (z, in m), zenith angle (θ, in deg), azimuthal angle (ϕ in deg), and wavelength (λ in nm). The equation of radiative transfer (RT) describes these interactions mathematically and constitutes the physical and mathematical framework for hydrologic optics. Analytic solutions to the radiative transfer equation (RTE) exist for simple cases, but for more realistic media numerical methods are required.
When a photon interacts with matter one of two things can happen. The photon can disappear, with its energy being converted to another form such as heat or the energy contained in a chemical bond. This process is called absorption. The photon can also change its direction and/or energy. Either of these processes is called scattering. The absorption and scattering properties of a medium such as seawater do not depend on the ambient light field, they are *Inherent Optical Properties (IOPs)*. That is, a volume of water has well defined absorption and scattering properties whether or not there is any light there to be absorbed or scattered. This means that absorption and scattering properties can be measured in the laboratory on a water sample, as well as in situ in the ocean.
The absorption coefficient describes how a medium absorbs light. The volume scattering function likewise describes how the medium scatters light. If you know these two, then you know how the medium interacts with unpolarized light. Surprisingly, measurement of these properties is not yet standard in autonomous, or ship-based monitoring, though researchers have used them on stations and buoys, bio-optical profiling floats and gliders, and in cruises.
Recommendation: Investigate the possibility to deploy this equipment, e.g., acs and BB3 or LISST-VSF in a true monitoring setup and communicate this with RWS.

A.1 The attenuation of light, Kd(λ), KdPAR, and observations of extinction by MWTL program.

Kd, the vertical diffuse attenuation function is a useful *Apparent Optical Property (AOP)*. These AOPs have properties that (1) depend both on the medium (the IOPs) and on the geometric (directional) structure of the radiance distribution, and that (2) display enough regular features and stability to be useful descriptors of a water body.
Following Lambert-Beer's law, downward irradiance (E_d in W m^{-2}) diminishes in an approximately exponential manner with depth (z). Under conditions, for which the incident lighting is provided by the sun and sky, the various radiances and irradiances all decrease approximately exponentially with depth in homogeneous water, when far enough below the surface (and far enough above the bottom, in optically shallow water) to be free of boundary effects. It is therefore convenient to write the depth dependence of $E_d(z,\lambda)$, for example, as

$$E_d(z,\lambda) \equiv E_d(0,\lambda) \exp\left[-\int_0^z K_d(z',\lambda)\, dz'\right] , \qquad (1)$$

where $K_d(z,\lambda)$ in m^{-1} is the diffuse attenuation function for spectral downwelling plane irradiance.
The diffuse attenuation function is the normalized or logarithmic depth derivative of E_d,

$$K_d(z,\lambda) = -\frac{d\ln E_d(z,\lambda)}{dz} = -\frac{1}{E_d(z,\lambda)}\frac{dE_d(z,\lambda)}{dz} \quad (\mathrm{m}^{-1}), \tag{2}$$

Change in magnitude of E_d will cancel out when it is present in both E_d(z1) and E_d(z2), leaving the value of K_d unchanged. K_d thus satisfies the stability requirement for an AOP. K_d also depends on the IOPs because changing them will change the irradiance changes with depth (Mobley, 1994). $\overline{K}_d$ is the average value of the vertical attenuation coefficient of downwelling irradiance over a depth interval 0 to z. For simplification, the average over depth symbol (line above K) and wavelength dependence (λ) are often not shown. Kd of 0.5 m^{-1} indicates that a photon entering the water column has a chance of $e^{-0.5}$ of being attenuated in the first meter (using: $x = \ln a; e^x = a$,note in this mathematical notation a does not indicate absorption).

For primary production we're interested in euphotic depth (and in-water light climate) for photons over **PAR**, the entire optical part of the spectrum. Irradiance (E) is expressed in W m^{-2} nm^{-1}, and (consequently, matching the left- and right-hand side) Mobley (1994) defines PAR coursing through a point $\vec{x}$ (in photons s^{-1} m^{-2}) as

$$PAR(\vec{x}) \equiv \int_{350nm}^{700nm} \frac{\lambda}{hc} E_0(\vec{x};\lambda)d\lambda \tag{3}$$

(E_0 is scalar irradiance, and PAR is often defined only for the visible wavelengths 400-700 nm because the 350-400 nm are absorbed near the water surface.) E_{dPAR} at depth z, using Eq 1, and introducing Avogadro's number gives:

$$E_{dPAR}(z) = \int_{350}^{700} E_0(\lambda)e^{-K(\lambda)z} \cdot \frac{\lambda \cdot 10^9}{c \cdot h \cdot 6.023 \cdot 10^{23}} \cdot d\lambda \tag{4}$$

K_{DPAR} can then be defined from E_{dPAR}(z) assuming that the spectrally integrated irradiance is attenuated completely in analogy with Eqs. 1 and 2

$$K_{DPAR} = \frac{1}{z_{euPAR} - z_0} \ln \frac{E_{PAR}(z_0)}{E_{PAR}(z_{euPAR})} \tag{5}$$

and equally so optical depth (see Eq 2)

$$\zeta = K_{DPAR} \cdot \left(z_{euPAR} - z_0\right) = \ln \frac{E_{PAR}(z_0)}{E_{PAR}(z_{euPAR})} \tag{6}$$

Optical depths of particular interest for primary production are those corresponding to attenuation of downward irradiance to 10% and 1% of the subsurface values: these are ζ= 2.3, fr $E_d(z)/E_d(0)$ = 0.10 and ζ= 4.6 for $E_d(z)/E_d(0)$ = 0.01. These optical depths correspond approximately to the mid-point and lower limit of the euphotic zone (z_{eu}=4.6/K_d).

In the Westerschelde, extinction coefficient was determined using 2 protocols (RWS,2003): the marine protocol (for all but Schaar van Ouden Doel, SvOD), and the fresh water protocol (for SvOD only), Stolte, oral comm. In turbid inland waters, the extinction can only be accurately measured if the distance between the two sensors is less than 0.5 m, both sensors measure under water. In salt waters, the extinction measurement is performed by varying the distance between the two light sensors, with one sensor on deck (RWS, 2003). This on deck sensor can measure $E_{dd}(\lambda)$ for direct solar irradiance, $E_{dsr}(\lambda)$ for diffuse molecular-scattered irradiance (Rayleigh component), and $E_{dsa}(\lambda)$ for diffuse aerosol-scattered irradiance, which is different from E measured in water.

The two methods are similar again when all salt water subsurface measurements are normalised to surface irradiance:

Eq. 3 $K_d(z_{1-2}) = ln\ [E_d'(z_1) – E_d'(z_2)] / (z_2 - z_1)$

where $E_d'(z)$ is normalized to the surface irradiance:

Eq. 4 $E_d'(z,\lambda) = E_d(z) / E_d(0+)$

Recommendations: Take measurements using both Kd protocols and compare results. Good optical measurements (of e.g. Irradiance, Ruddick et al., 2019) are scarce and needed to calculate or validate Kd. Save and store also the original measurements.

A.2 Which Kd algorithms were recently tested? An overview

Reflectance derived from in situ measurements or atmospherically corrected satellite data can (indirectly) provide information about the extinction of light in water. Reflectance is upwelling irradiance (Eu) normalised by downwelling irradiance (Ed), and Eu and Ed interact with optically active substances, the water molecules, chlorophyll, suspended matter and CDOM through their absorption and / or scattering properties. Both measurements are AOPs and hence they also depend on the geometric (directional) structure of the radiance distribution.

Eleveld et al. (in prep) found promising first results for turbid Lake Markermeer from a (semi-)empirical algorithm (Gons et al., 1997) that related ratios of reflectances and backscattering at 776 nm to LI-COR extinction measurements. With optical modelling, b_b776 can be related to reflectances at 776, these reflectances can be measured with in situ spectrometers (Vabson et al., 2019) and by imaging spectrometers on satellites. The latter measurements still need to be atmospherically corrected.)

For a second algorithm, Verhoog (2018), Eleveld et al. (2018) and Villars et al. (2019) have shown that using reflectances from of atmospherically corrected Landsat-8 OLI and Sentinel-2 MSI imagery in the Quasi-Analytical Algorithm (QAA, Lee et al., 2005 and updates) does not always work to calculate KdPAR for turbid waters such as Westerschelde and Markermeer. Results might be impacted by the atmospheric correction, several were tested in the reports mentioned above. It could also be that the underlying assumptions for the optical model in QAA are not valid for these water types and should be recalculated using locally measured absorption and scattering properties of optically active substances in the water. In the QAA model, first absorption (a) and backscattering coëfficiënts (b_b) are estimated based on reflectances. Subsequently, Kd-values are calculated for individual bands based on absorption- (a) and backscatter coëfficiënts (b_b) and a solar zenith angle (θ set at 45 deg).

Recommendations: The Gons et al. algorithm can be tested for Westerschelde, or reconstructed when optical measurements are available, the QAA algorithm can be tested on in situ spectra.

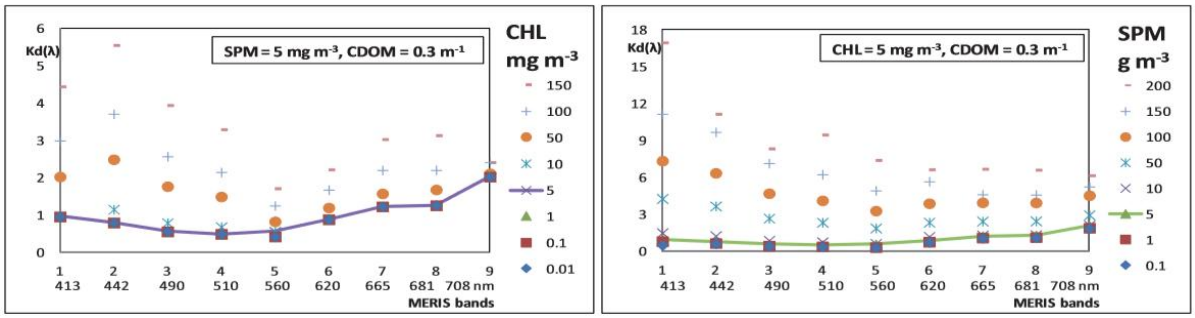
The optimal measurement setup would be to combine measurements from an in situ spectrometer with LI-COR measurements, and preferably also some simultaneous measurements of absorption and scattering property scoops (IOPs) and concentrations, to further calibrate and validate the algorithms, though semi-empirical and semi-analytic solutions always have limited applicability.

The use of (output from) a numerical radiative transfer model, HydroLight (Mobley and Sundman, 2001) is proposed in the next section. So that we work directly from first principles, a physical approach, instead of returning to empirical work using indirect measurements of IOPS and approximations. (In physics and other sciences, working from first principles means that you use established science and do not make assumptions such as empirical modeling and parameter adjustment). Such a physically-based robust approach seems to optimally complement the regression model described in section 1.

A.3 Proposal to use numerical RT results for Westerschelde

For comparison with the regression model, now the RT-relation between Kd(λ) at 1m depth and other regularly measured water quality parameters is proposed for Westerschelde. HydroLight was created to compute radiance distributions and related quantities (irradiances, reflectances, diffuse attenuation functions) from bio-optical input (IOPs).
After correct parameterisation of Hydrolight, Kd can be calculated as a function of total absorption (a), total scattering (b), depth -1 m), zenith angle (θ), azimuthal angle (ϕ), and wavelength (λ), Pasterkamp et al., 2005.
Next, a simple specific IOP based bio-optical model for the Westerschelde (Restwes99Oroma02, Van der Wal et al., 2010; Eleveld et al, 2014) can be used to distinguish the contribution of individual optically active substances to total a and total b. (Such a simple bio-optical model is e.g. $a_k(\lambda) = a_k^*(\lambda) \cdot conc_k$ and similar for b or b_b k is the optically active constituent.

Consequently, Kd attribution for variable concentrations and IOPs can be computed and plotted, as tested previously by Eleveld et al. (2013) for the North Sea. See Fig. 1. for simulations with variable concentrations for the North Sea.



FigureApx A.1Example showing a tests of extreme scenarios for the North Sea (Eleveld et al, 2013). Note also the spectral dependency, which will be overlooked when examining Kd over PAR. This is important because photons of different wavelengths are not equally likely to be absorbed.

As a first estimate, the following settings are proposed for such a sensitivity study, based on analysis of RWS monitoring data (Stolte and Schueder 2019), and four Scheldt estuary transects (in 1999, 2002, winter 2005 and summer 2006, Eleveld et al., 2014; Astoreca et al., 2006):
CHL varies from 0.01 – 35 µg l^{-1}(Stolte and Schueder 2019, Figure 2.3)
SPM from 0.1 – 200 mg l^{-1} (Stolte and Schueder 2019, Figure 2.5; Eleveld et al., 2014)
aCDOM from 0.01– 2 m^{-1} (Eleveld et al., 2014; (Astoreca et al., 2009)
The benefits are that with this setup key controls on the bottom-up modelling of primary production in optically complex waters can already be explored.
Recommendation: Test the approach as outlined above and consider redoing the RT modelling chain if e.g. implementation in the biogeochemical-ecological modelling is the ambition. Note that KdPAR (and probably also E_0) can then be derived from HydroLight.
More research on biogeochemical feedbacks, and dependencies between light climate, concentrations and sIOPs is still needed, including to link this back to empirical modelling constants b =0.022, c = 0.393 (Section 1). The feedback with pelagic primary production in light limited systems is interesting. Also for calculating gross primary production, various ocean colour/marine optics approaches have been developed and these are likely complementary to the model approach.

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