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Primary Research Paper

Spatial and temporal patterns of water quality along the estuarine salinity gradient of the Scheldt estuary (Belgium and The Netherlands): results of an integrated monitoring approach

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Abstract

This paper presents the results of 7 years of integrated monitoring along the Scheldt estuary. The combination of two datasets resulted in a full description of the estuaries water quality parameters from the mouth to the upper boundary, including an extended fresh water tidal part. A synthesis of the monitoring results and all relevant ecological knowledge on the Scheldt allowed to identify opportunities to optimize its management. The results show that the effect of discharge on salinity has a distinct maximum in the polyhaline to mesohaline transition area. Oxygen conditions, nitrogen removal and phytoplankton regulation can be enhanced and improved through management measures within the estuary. To lower carbon and phosphorous loads however measures should be taken within the catchment. To restore most of its ecological functions the estuary needs more space. Optimal locations to address specific functions can be derived from the monitoring results.

Introduction

Estuaries are cited among the most productive biomes of the world (Costanza et al., 1993). They support important biogeochemical processes that are central to the planet's functioning, e.g. nutrient cycling (Billen et al., 1991; Costanza et al., 1997). Estuaries are the interface between terrestrial and coastal waters. They often are characterized by steep chemical gradients and complex dynamics, and these can result in major transformations in the amount, chemical nature and timing of the flux of material along these river–sea transition zones. As estuaries concentrate waters from very large land surfaces into relatively small water bodies (Heip et al., 1995), the biogeochemical processes and trophic interactions within estuaries can play an important role in the management of water quality problems. This ecological functioning is considered to be of major concern, as estuaries offer the last opportunity to manage water quality problems before they become uncontrollable in the coastal waters.

However, this huge potential of ecological functions is far often repressed by human impact (Suchanek, 1994; Gray, 1997). The Scheldt estuary is not an exception. It is characterized by a notorious history of pollution and eutrophication (Wollast, 1988; Boderie et al., 1993), and the estuarine intertidal habitats have suffered from important area reduction and quality degradation (Meire et al., 2005). The Schelde estuary nevertheless has some high ecological values, being internationally important for several bird species (Van den Bergh et al., this volume), and with large parts of the estuary being designated under the Ramsar Convention and European Birds and Habitat Directive. Although the risk of a further deterioration and habitat loss is still present, there is a growing awareness that conservation and restoration of the estuarine ecological functions is needed. In the past only some local measures have been undertaken, but it is only recently recognized that a restoration of ecological estuarine functioning requires an integrated, whole system approach. One of the fundamental steps in establishing such an integrated water management approach, is the development of an integrated monitoring programme. For the Scheldt estuary, ecological data in general and water quality monitoring data in particular are amply available, but they are scattered over many sources and ecological research focussed until recently mainly on the marine and brackish part of the estuary (Van Damme et al., 1995). The freshwater part received less attention (e.g. De Pauw, 1975; Hummel et al., 1988), although it covers more than one third of the total length of the estuary. A serious attempt to integrate ecological estuarine research at the scale of the whole Scheldt estuary, including the freshwater part, was initiated through the OMES program (Meire et al., this volume). This program was set up to fill in knowledge gaps in order to allow fundamental ecological management of the whole estuarine system through the development of an ecosystem model. Within the frame of this research program a long-term water quality monitoring program was set up in 1995 to monitor all essential parameters from the mouth till the upper boundary, including the freshwater part. The monitoring program sufficient spatial sampling to allow assessments of variability at the scale of the ecosystem. To achieve this programme, an ongoing monitoring program from the Netherlands Institute for Ecology – Centre for Marine and Estuarine Research (CEME), covering the marine and brackish zone of the estuary (e.g. Kromkamp et al., 1995), was extended with an OMES monitoring program covering the Belgian part of the estuary. The combination of these data allowed for the first time the presentation of an actualised full description of the basic water quality of the whole Scheldt estuary. In this study, spatial and temporal variability in susnpended particles, nutrients and chlorophyll *a* concentrations is described for the period 1995–2002. The major underlying processes and mechanism are discussed and some opportunities to optimise ecological water management are proposed.

Material and methods

Study area

The Scheldt estuary is located in Northern Belgium (Flanders) and the Southwest Netherlands (Fig. 1). It extends from the mouth at Vlissingen (0 km) till Gent (158 km); there tidal movement is stopped through a complex of sluices. The lower and middle estuary, the Westerschelde (55 km long), is a well mixed region characterized by a complex morphology with flood and ebb channels surrounding several large intertidal mud and sand flats. The surface area of the Westerschelde is 310 km^2, with the intertidal area accounting for 35% of the area. The average channel depth is approximately 15–20 m. Near the Dutch/Belgian border the estuary narrows and becomes characterized by a single tidal channel and is called Sea Scheldt (105 km long). The surface of the Sea Scheldt amounts to only 44 km^2. The Sea Scheldt is further divided into the Lower Sea Scheldt, stretching from the Dutch Belgian border to Antwerpen, and the Upper Sea Scheldt, stretching from Antwerpen to the upstream boundary at Gent. The major tributaries of the estuary are the Rupel (tidal), the Durme (tidal) and the Dender (non-tidal). The total length of the river, including both estuary and upper river, is 355 km. The catchment area of the Scheldt is 20 331 km^2. In this area about 10.4 million people are living, forming a dense population of an average more than 5 ind./ha. Large efforts for industrial and municipal waste water treatment has been undertaken during the last decade in Flanders, but still untreated municipal waste water is being discharged into the estuary, with the city of Brussels as the most prominent example. The untreated municipal wastewater of Brussels reaches the estuary

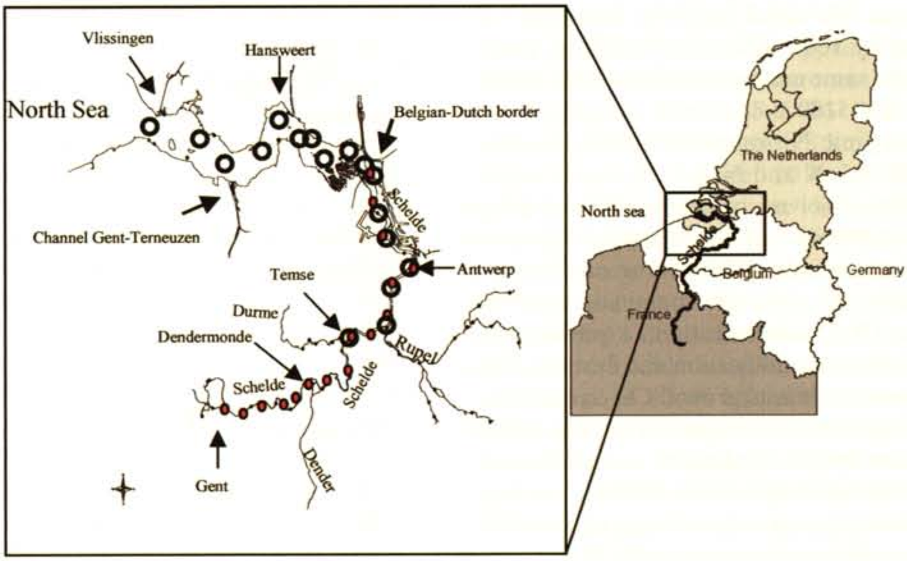

Figure 1. Map of the Scheldt estuary.

through the Rupel tributary near Schelle (90 km). Agriculture in the catchment area is intensive, and is responsible for a considerable part of the nutrient inputs to the estuary (Baeyens et al., 1998). Some large industrial areas are concentrated beside the estuary: near Gent, Antwerpen and Vlissingen.

Sampling

During both the CEME and OMES monitoring cruises surface water samples were taken monthly in the middle of the river from a ship. In the period March–May, during spring phytoplankton bloom, one additional CEME cruise was organized per month. The results presented here deal with the period December 1995–2001, as the OMES campaign started in December 1995. The CEME monitoring was divided over two subsequent days: a first cruise running from Vlissingen to Hansweert and a second cruise from Hansweert to Temse. The second CEME campaign cruise matched the first campaign cruise of the OMES program, running from the Dutch/Belgian border to Dendermonde. The next day the stations between Dendermonde and Gent were monitored, so that the estuary was each month fully covered within a time span of three days. The CEME campaign covered 17 stations between Vlissingen and Temse; the OMES campaign started at the Dutch Belgian border, and covered 16 stations along the longitudinal gradient of the Sea Scheldt (Fig. 1). In this way the zone between the Dutch Belgian border and Temse was overlapping in the two campaigns.

The Administration of Waterways and Sea (AWZ) continuously measures discharge of the Bovenschelde (the Schelde just upstream Gent, where tidal influence is stopped by sluices), the Dender and the Rupel, and we calculated daily average discharges using that data set.

Analysis

Analytical methods were as much as possible conducted in a similar way for both monitoring programmes, but small differences existed. During the CEME cruises, temperature, oxygen saturation, salinity and pH were measured *in situ* using a Water Quality Multiprobe Hydrolab H20. During the OMES cruises, temperature and oxygen were measured *in situ* with a 'WTW OXI 91' oxygenmeter, salinity was measured with a 'WTW LF 91' conductivity-meter, and pH was measured with a WTW pH 330 pH-meter.

Samples were stored at 4 °C and were analyzed within 24 h after sampling. CEME partner analysed NO_3^-–N, NO_2^-–N, NH_4^+–N, orthophosphate (DRP), dissolved silica (DSi) and total phosphorous (totP) (after destruction in H_2SO_4 and $K_2S_2O_8$) colorimetrically using a SKALAR SA 4000 segmented flow analyzer, while SO_4^{2-}–S was analysed using a SKALAR SA 2000. Excepted for DSi, analyzed by ICP–OES (Iris®), OMES partners applied the same methods as CEME but using a SKALAR SA 5100 colorimeter instead. Total Dissolved Inorganic Nitrogen (TDIN) is the sum of NO_3^-–N, NO_2^-–N and NH_4^+–N.

Samples for dissolved organic carbon (DOC) were filtered on Whatman GF/C glassfiber filters of 1.0 μm nominal porosity (CEME), or on Gelman glassfiber filters of 0.45 μm nominal porosity (OMES). For DOC determination, a preliminary treatment with H_2SO_4 acidification and flushing with nitrogen gas to remove background CO_2 concentration was performed at CEME and in OMES. DOC was then set free by UV-irradiation, and analysis of CH_4 on a SKALAR coupled FID (CEME), or further oxidation to CO_2 and analysis using a SKALAR (phenolphtalein 550 nm detection; OMES).

Suspended matter (SPM) was determined gravimetrically after filtration on pre-combusted Whatman GF/F filters. Particulate organic carbon (POC) was determined on the same filters using a Carlo Erba element analyzer after Cr_2O_3 and $AgCo_3O_4$ catalysed oxidation and segregation on a Haysep-Q-column (CEME & OMES).

Samples for analysis of chlorofyl *a* (Chl a) were first filtered on pre-combusted Schleicher Schuell number 6 filters (CEME) or 45 μ Sartorius filters (OMES). Chl a was by extracted in 90% acetone and analysed using reversed phase HPLC (Waters Fluorescence detector 474, excitation at 430 nm, emission at 650 nm; CEME); or set free after addition of N,N-dimethylformamide, and analysed colorimetrically at 647 and 664 nm (OMES).

Because sometimes a different methodology was used by CEME and OMES, a Wilcoxon Rank–Sum test was performed on results for a common sampling station at Antwerp, to test the significance of the differences between CEME and OMES data (Table 1). The comparison showed concordance for most parameters. Only for SPM, total P and pH did the datasets not match. In the present paper we used the OMES data set for SPM, total P and pH for the overlapping zone.

The monitoring results are presented graphically using the surface mapping system software 'SURFER', Version 5.01. Interpolation was performed using linear ordinary kriging, radius anisotropy 1/100. Statistics were performed using S-Plus 2000. The data for sampling points common to CEME and OMES cruises were aggregated and averaged per station before they were included in the dataset.

Table 1. Results of a Wilcoxon Rank-Sum test to test the significance of the differences between CEME and OMES data of a common sampling station at Antwerp

Parameter	*n*	Z	*p*
t°	85	0.32	0.75
pH	81	−4.99	0.00
Salinity	83	−0.87	0.39
O_2 (%)	81	0.09	0.93
NO_3^-	85	0.20	0.84
NH_4^+	85	0.43	0.67
NO_2^-	85	−0.59	0.55
DRP	85	0.28	0.78
TotP	85	3.10	0.00
DSi	79	0.46	0.64
SPM	65	−3.01	0.00
POC	65	1.02	0.31
Chl a	65	1.69	0.09
DOC	59	−1.39	0.16

The p-values of significant differences are in bold (t° = temperature, O_2 (%) = oxygen saturation, DRP = dissolved reactive phosphate = orthophosphate, tot P = total phosphorous, DSi = dissolved silica, SPM = suspended matter, POC = particulate organic mater, Chl a = chlorofyl a, DOC = dissolved organic

Results

Hydrology

The average yearly discharge at Schelle (103 km) varied from a minimum of 78 $m^3\ s^{-1}$ in 1996 to a maximum of 191 $m^3\ s^{-1}$ in 2001 (Fig. 2). Over this

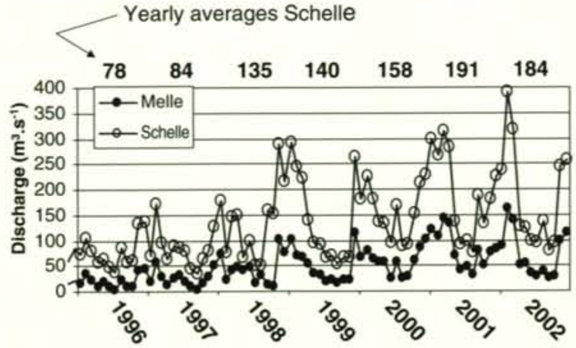

Figure 2. Discharge in the Scheldt estuary at the upward boundary (Melle) and at the mouth of the Rupel tributary (Schelle).

period the discharge increased year by year; in 2002 a similar discharge was observed as in 2001. There was a clear seasonal variation with maxima in winter and minima in summer (Struyf et al., 2004). Average monthly discharge was strongly related to total monthly rainfall in winter, but this relation was less pronounced in summer (Struyf et al., 2004). At the upward boundary (Melle, 158 km) the same seasonal pattern of discharge was noted as for Schelle.

Parameter patterns

Temperature ranged between 0 °C (in January 1996 with pack ice drifting in the estuary) and 24 °C in August 1997 and showed an obvious seasonal pattern (Fig. 3). Applying the Venice System (1958) the polyhaline zone (18–30 PSU) was observed to range from the mouth of the estuary to the zone between approximately 20 km (winter 2001) and 55 km (summer 1996) (Fig. 4). An oligohaline zone (0.5–5 PSU) was situated roughly upward Antwerp (78 km), with downstream extensions of over 20 km during peaking discharge. Sporadically 'true' limnetic conditions (0–0.5 PSU) appeared. The oligohaline and limnetic zones together will further be referred to as the 'freshwater part' of the estuary. The mesohaline zone (5–18 PSU), further called brackish part, was highly variable in space and time, stretching over a range of 20–40 km, and shifting according the changes in the oligohaline and the polyhaline zones. Between 1996 and 2001 isosalinity lines generally shifted I downstream direction over more than 20 km. For all stations, the slopes of the linear trends of salinity vs. discharge (data for the

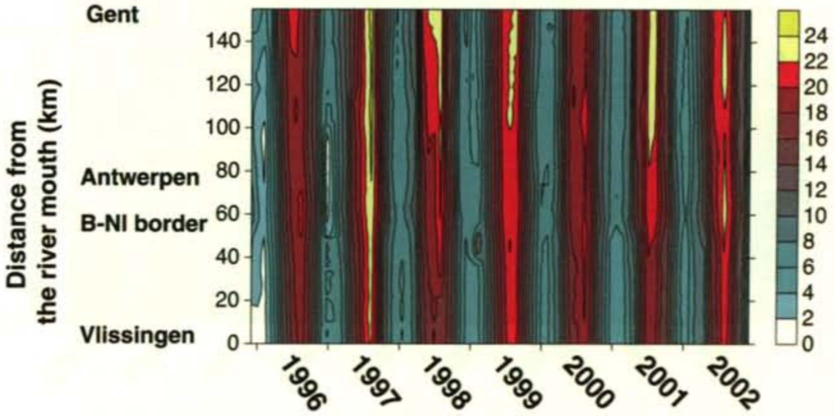

Figure 3. Temperature along the estuarine axis of the Scheldt, unit: °C.

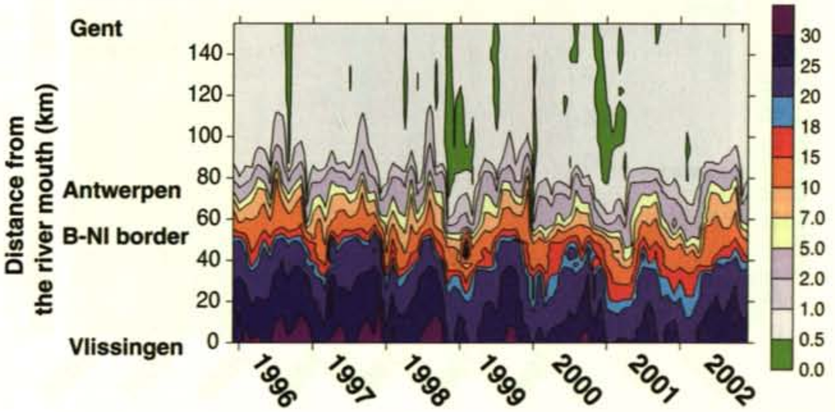

Figure 4. Salinity along the estuarine axis of the Scheldt, unit: psu.

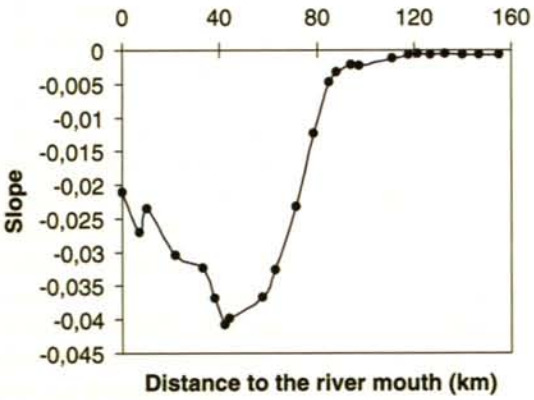

Figure 5. Slopes of the linear trend lines of salinity vs. discharge plots over the longitudinal gradient. Salinity – discharge plots were made for every sampling station and included all data of the studied period.

whole sampling period) were plotted vs. station position along the river (Fig. 5). The linear trends were all negative and significant (data not shown). The results show that the effect of discharge on salinity is in general most pronounced around 42 km (most negative slope), i.e. at the transition of the polyhaline to the mesohaline zone.

Values of pH were confined in the alkaline range (Fig. 6). Lowest values were measured predominantly in the zone between 70 and 100 km, i.e. in the downstream part of the oligohaline zone.

Oxygen concentrations are generally low in the Sea Scheldt and increase quickly near the Dutch Belgian border, downstream 70 km (Fig. 7). During winter O_2 concentrations increased in the Sea Scheldt, whereas in the Westerschelde supersaturation was noted during spring and summer,

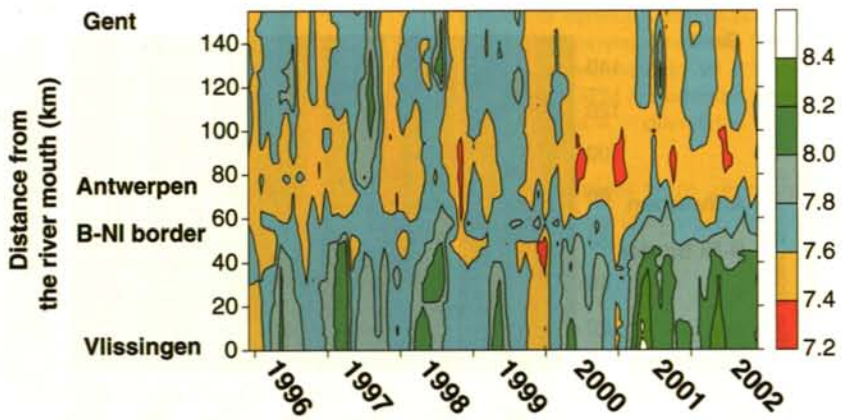

Figure 6. pH along the estuarine axis of the Scheldt.

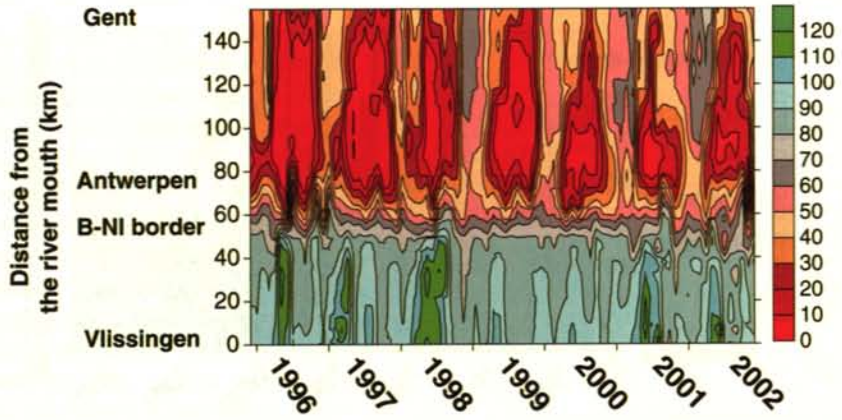

Figure 7. Oxygen saturation along the estuarine axis of the Scheldt, unit: %.

accompanied by a rise of pH (Fig. 6). In the freshwater zone average summer oxygen concentrations improved considerably from 1996 to 2001, and this increase related positively with discharge (Struyf et al., 2004.). The worst O_2 conditions persisted around the mouth of the Rupel tributary (92 km), carrying mostly untreated domestic waste from the Brussels region.

In the Westerschelde NH_4^+–N concentrations dropped almost to zero, a phenomenon that was extended in summer upstream until Antwerp (78 km) (Fig. 8). In winter ammonium concentrations were much more elevated in the Sea Scheldt, but a significant decrease was observed from 1996–1997 to 2001–2002. This difference was more pronounced for winter compared to summer (Struyf et al., 2004). Year round nitrate concentrations were significant even downstream the Dutch Belgian border (58 km) (Fig. 9). Yearly maxima, reached in winter time, slightly dropped during the study period, but seasonal differences became smaller and in 2001 and 2002 nitrate concentrations remained high throughout the year; in 2001–2002 nitrate concentrations at the river mouth were higher than in 1996–1997. Nitrite peaked during summer in the freshwater part, while it did during winter in the saline part, although in the latter concentrations were very low (Fig. 10). Extreme nitrite values were recorded in the summer of 2001, coinciding with extreme Chl a (Fig. 11) contents. In contrast to O_2, the TDIN pattern did not reflect a direct influence of the Rupel tributary (Fig. 12). TDIN exhibited a clear longitudinal gradient, with maximal values in the

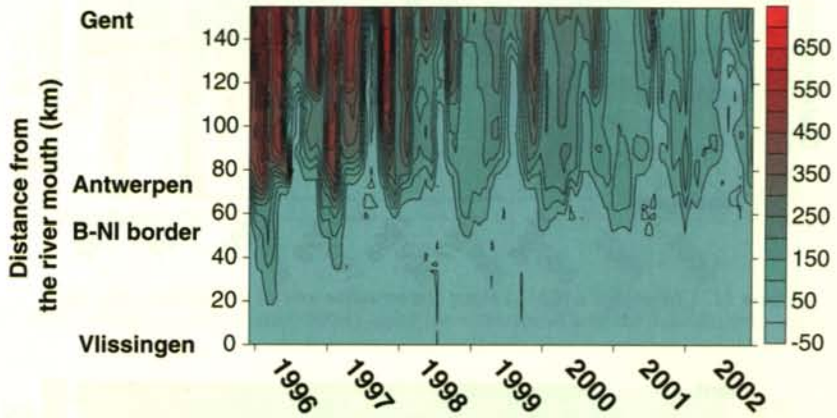

Figure 8. Ammonium along the estuarine axis of the Scheldt, unit: μmol N.L^{-1}.

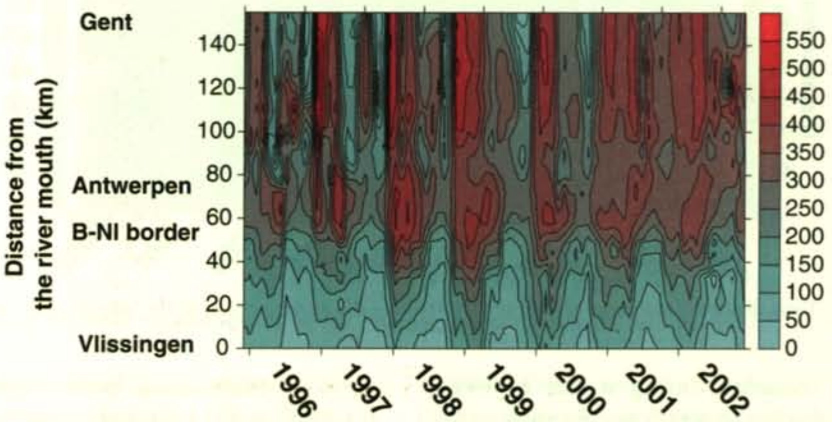

Figure 9. Nitrate along the estuarine axis of the Scheldt, unit: μmol N.L^{-1}.

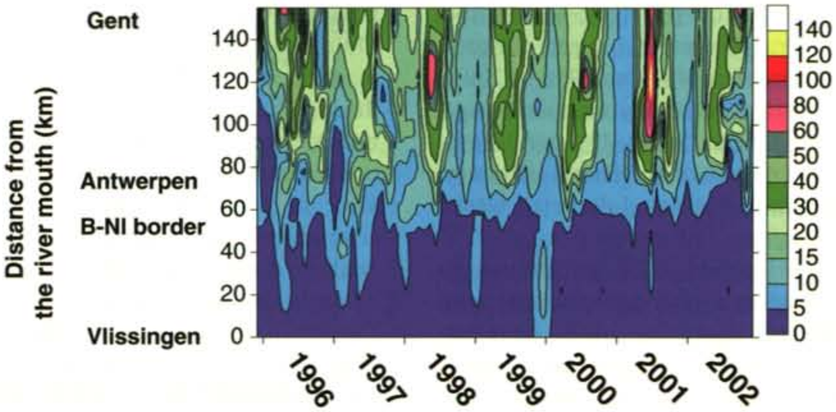

Figure 10. Nitrite along the estuarine axis of the Scheldt, unit: μmol N.L^{-1}.

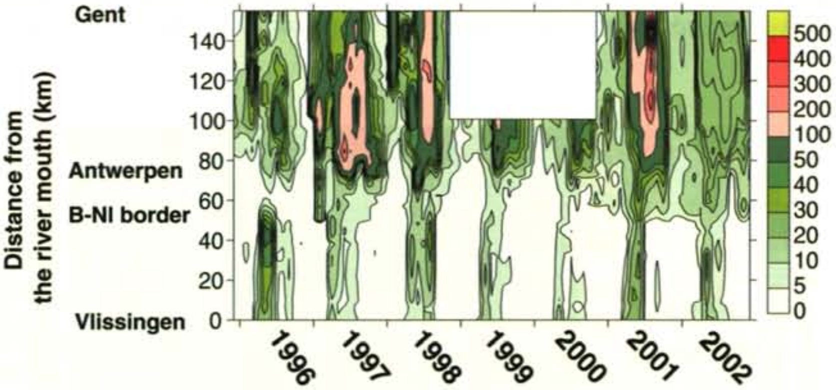

Figure 11. Chlorophyl a (Chl a) along the estuarine axis of the Scheldt, unit: μg.L^{-1}.

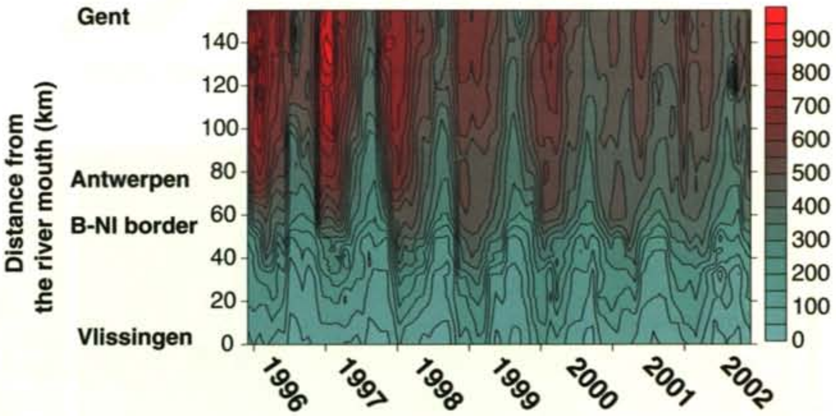

Figure 12. Total dissolved inorganic nitrogen (TDIN) along the estuarine axis of the Scheldt, unit: μmol N.L^{-1}.

freshwater part recorded during winter of 1996 and 1997 when discharge was low. No such variability was observed near the river mouth.

A distinct zone with high DRP concentrations was apparent upstream 90–100 km (Fig. 13). DRP and total P concentrations

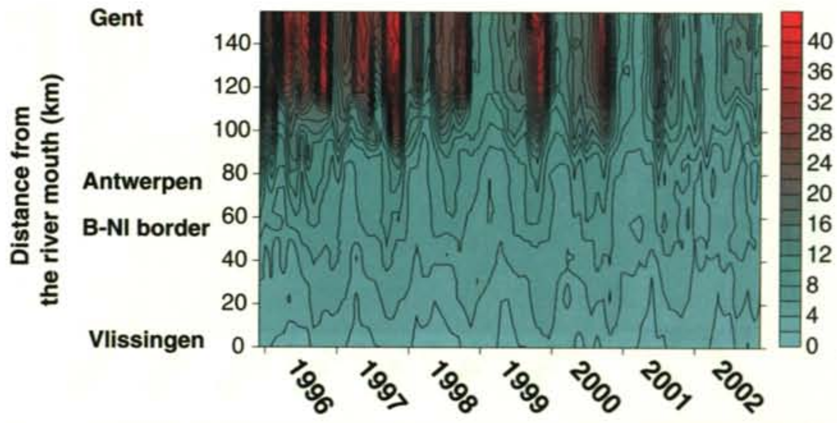

Figure 13. Orthophosphate (DRP) along the estuarine axis of the Scheldt, unit: μmol P L^{-1}.

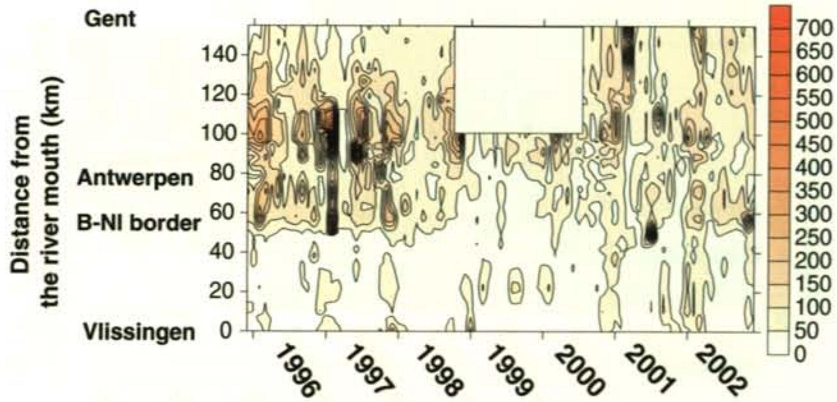

Figure 14. Suspended particulate matter (SPM) along the estuarine axis of the Scheldt, unit: mg L^{-1}.

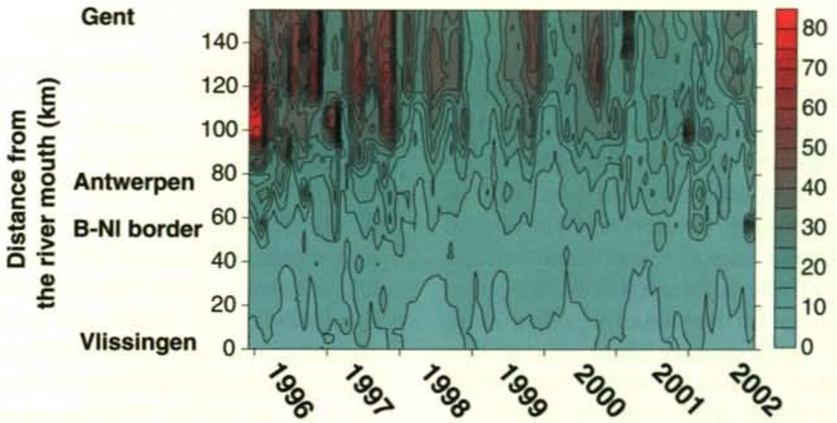

Figure 15. Total phosphorous along the estuarine axis of the Scheldt, unit: μmol P L^{-1}.

dropped significantly in the region where higher SPM concentrations prevailed (100–120 km) (Figs 14 and 15). In the outer reaches of the estuary DRP minima coincided with phytoplankton blooms during spring or summer (Fig. 11).

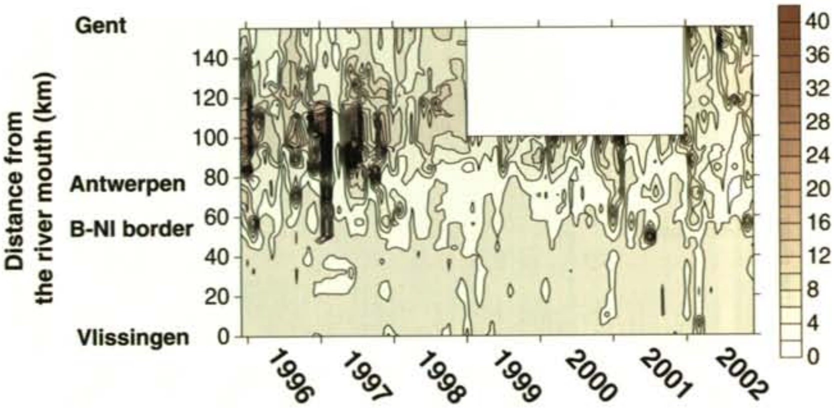

Figure 16. Particulate organic carbon (POC) along the estuarine axis of the Scheldt, unit: mg C L^{-1}.

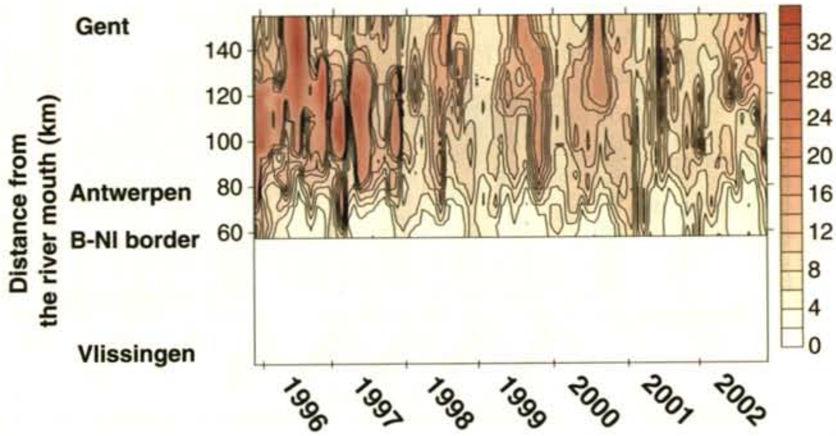

Figure 17. Biological oxygen demand (BOD) along the estuarine axis of the Scheldt, unit: mg L^{-1}.

The pattern of SPM was quite patchy (Fig. 14), showing in general much higher concentrations in the Sea Scheldt than in the Western Scheldt. The seasonal variations are due to a complex of factors, such as the river discharge (transport, shift of the turbidity maximum), temperature (biological activity, climatologic factors) and land erosion (terrestrial input of fine sediments). The pattern of SPM is discussed in further detail in Chen et al. (2004). For the whole estuary there was a close linear relationship between SPM and POC (Fig. 16) (POC = 0.052 SPM + 0.190, $R^2 = 0.71$, d.f. = 1636, $F = 3946$, $p = 0$). Biological oxygen demand (BOD) was not measured in the Westerschelde proper, but from Figure 17 it is clear that BOD decreased with increasing salinity, as was also observed for DOC (Fig. 18). However, there was no significant relationship between BOD and DOC. High BOD values were observed in 1996–1997, while DOC concentrations remained rather constant over the years.

Chl a concentrations were highest in the freshwater part, decreased in the brackish part, and increased again in the marine part (Fig. 11). Chl a showed peaks during spring and summer, especially in the freshwater part. No clear trend was observed during the study period, but in 2002 the Chl-a concentrations in the freshwater part were relatively low. The pattern of DSi showed a distinct decrease during summer compared to winter (Fig. 19). Winter DSi concentrations remained rather constant over the years, while summer concentrations greatly increased over the study period (Struyf et al., 2004).

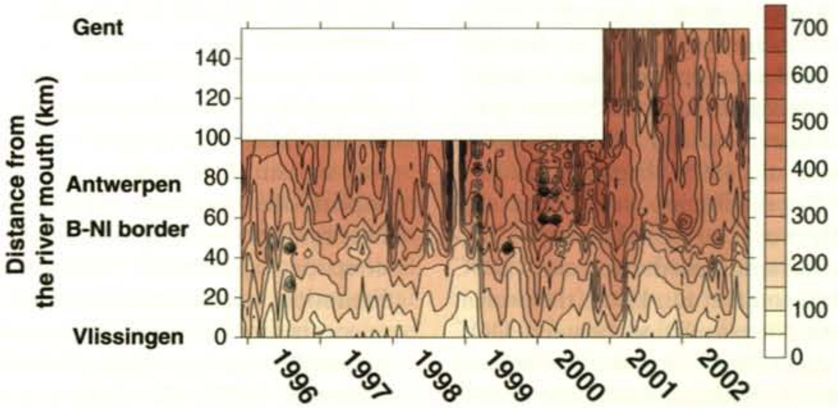

Figure 18. Dissolved organic carbon (DOC) along the estuarine axis of the Scheldt, unit: μmol C L^{-1}.

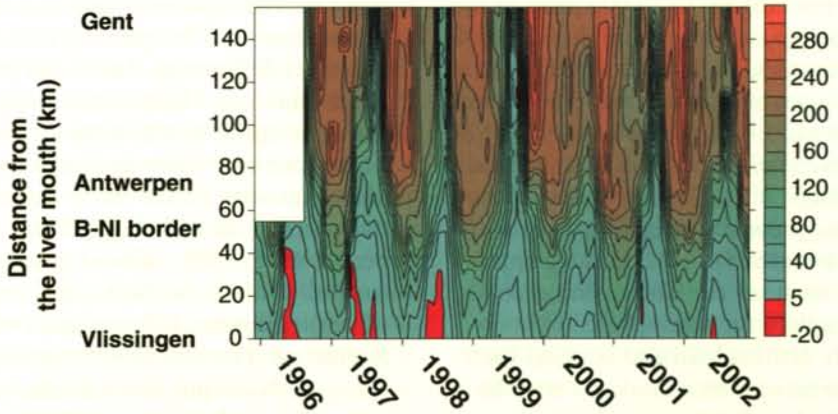

Figure 19. Dissolved silica (DSi) along the estuarine axis of the Scheldt, unit: μmol Si L^{-1}. Concentration contour fills lower than 5 μM Si are highlighted in red, indicating possible limitation for diatoms (Van Spaendonk et al., 1993).

Discussion

The high spatial resolution and the monthly measurement frequency in this study allowed for a detailed description of the spatial and temporal variability of the water quality along the whole salinity gradient of the Scheldt estuary. For most parameters, spatial gradients were obvious, reflecting the downstream increasing influence of the seawater entering the estuary from the North Sea during each tide. Seasonal and inter-annual variability was large, mainly reflecting climatic conditions such as fresh water run off and temperature variations.

Carbon and oxygen

The phytoplankton contribution to POC estimated from stable carbon isotope data of SPOM and DIC, was estimate to vary between 17 and 65%, during periods of algal bloom (Hellings et al., 1999). Slightly lower values, from 10% to 30%, are reported by Muylaert et al. (personal communication), as deduced from phytoplankton productivity studies (Muylaert et al., Submitted). Since the contribution of terrestrial vegetation to POC was estimated at 2–10% (Tackx et al., 1999), the anthropogenic fraction of the POC in the freshwater part can therefore be estimated at around 45% during summer and 80% during winter (Hellings et al., 1999).

Several carbon balances have been published for the Scheldt estuary in the past (Wollast, 1978; Soetaert & Herman, 1995a; Frankignoulle et al., 1996). They all show that the major part of the carbon input in the estuary does not reach the sea and that most of the carbon load is processed within the estuary itself. Only the most refractory fraction reaches the coastal waters (Soetaert & Herman, 1995a). Bacterial production in the Scheldt estuary, especially in the brackish and freshwater part, showed highest values amongst other European estuaries (Goosen et al., 1995). Especially around the mouth of the Rupel tributary strong heterotrophy was noted (Heip et al., 1995). Although a correlation between bacterial production and DOC hs been reported (Goosen et al., 1995), the present study did not reveal a correlation between DOC and BOD. The oxygen depletion is most prominent at the Rupel mouth (90 km), where the mostly untreated waste water of Brussels, reaches the estuary. Bacterial respiration is high resulting in CO_2 partial pressure, reaching up to 10 000 μatm in the water column, corresponding to more than 2 500% oversaturation (Frankignoulle et al., 1996). These were the highest values of several European estuaries (Frankignoulle et al., 1998), the freshwater part of the Scheldt not even being included. For the freshwater part, Hellings et al. (2001) calculated even higher CO_2 partial pressures exceeding those in the brackish zone by about about 50%. Hence it appears that the fate of most of the carbon input to the estuary is outgassing to the atmosphere.

The enormous detritus load and coupled bacterial production seems to form a dead end in the food chain. So far, neither detritus nor bacteria were proven to constitute an important nutrient source for higher trophic levels, considering such facts as the importance of phytoplankton for macrobenthic suspension feeders and of microphytobenthos for deposit feeders (Herman et al., 2000), the feeding selectivity of zooplankton (Tackx et al., 2003), the lesser importance of grazing on bactivorous ciliates than on herbivorous ciliates (Hamels et al., 1998), or the relationship between system averaged macrozoobenthic biomass and system averaged phytoplankton productivity (Herman et al., 1999). Thus, it would appear that the anthropogenic carbon load does not benefit to overall estuarine ecological functioning as it does not really lead to enhanced productivity of the higher trophic levels.

Nitrogen

In the fresh water part the relative contribution of ammonium to the total N pool represented more than 50% during winter (January–February). At the Dutch Belgian border (brackish part) nitrate contributed always, except in the low discharge winter periods of 1996 and 1997, more than 70% of total N. At the transition from the freshwater to the brackish zone a peak of nitrification activity was measured (De Bie et al., 2002) which could explain the observed pH minima in this zone. Nitrification activity declined in downstream direction, probably because of ammonium limitation, while in the upstream part low oxygen concentrations appear to control nitrification, as deduced from modeling (Billen, 1975; Billen & Somville, 1982). As oxygen concentrations increased since the second half of the seventies (trend at the Dutch Belgian border), the nitrification front moved upstream and intensified to become the process with the greatest impact on the N-load (Soetaert & Herman, 1995b; Regnier et al., 1997). More than one third of the oxygen consumption was then due to nitrification (Ouboter et al., 1998).

Nitrate is not only produced in the estuary, it is also removed. In the early eighties, a first mass balance for nitrogen (N), based on data for the period 1975–1983, showed a reduction of the N-load of 40–50% in the estuary, mainly as a result of denitrification (Billen et al., 1985). Modeling by Soetaert & Herman (1995b) showed that 10 years later the N-output towards the sea had doubled. and confirmed that denitrification decreased due to the improved oxygen conditions. This phenomenon became locally known as the paradox of the Scheldt estuary.

Denitrification in the Scheldt was found to be most important in the pelagic compartment (Soetaert & Herman, 1995b). Pelagic denitrification however is subject to changes in residence time of the water in the estuary, itself largely controlled by discharge. The nitrogen fraction that was denitrified could be predicted from the freshwater residence time for several estuaries (Dettmann, 2001). Soetaert & Herman (1995c) modelled that the difference between a typical winter and summer discharge caused a difference of residence time in their most upstream studied compartment (i.e. around Temse; 100–108 km): In summer the residence time was about 30% less than in winter. Further upstream the impact of discharge is bigger, so the effect on residence time must be greater. The average summer discharge (June–August) in 2000 (118 $m^3\ s^{-1}$) was double of that in 1996 (56 $m^3\ s^{-1}$). According

Soetaert & Herman (*op. cit.*) this implies that in the fresh water part the nitrogen turnover due to pelagic denitrification had to be at least 20% lower in 2000 than in 1996. This is not reflected in the profile of TDIN, but the effects of discharge on N-nutrients are multiple (Struyf et al., 2004). A discharge dependent increase of the nitrate and TDIN load in the fresh water part indicated increased input in the estuary proper, buth this effect was partly masked by dilution. So far an increased output to the coastal zone is not clear. Apart from the pelagic volume aspect of denitrification there is also a surface effect. For a tidal mudflat in the brackish zone near the Dutch Belgian border Middelburg et al. (1995) calculated that 55% of the total N flux from water column to sediments was denitrified. Comparison with Soetaert & Herman (1995b) leaded to the conclusion that tidal areas between the Vlissingen and Temse (110 km) accounted for 14% of the total N removal in that part of the estuary. In the fresh water zone more intense denitrification values were found (Van Damme et al., in preparation), which is in accordance with Rysgaard et al. (1999) who revealed a clear negative relationship between denitrification and salinity. In tidal marshes the flux of nitrate from the water into the sediment appears to be the limiting factor for denitrification, while it is consistently enhanced by bioturbation (e.g. Chatarpaul et al., 1980; Pelegri & Blackburn, 1995). Extreme densities of Oligochaetes are reported for the the Scheldt estuary (Seys et al., 1999) and it is therefore likely that these will have a significant effect on dentitrification. In contrast, recent results for a whole ecosystem ^{15}N-ammonium labeling experiment in a fresh water marsh at Tielrode (104 km) indicate that about half of the retained ammonium label was nitrified, while denitrification was of minor importance compared with nitrification (Gribsholt et al., in preparation). Denitrification in the root zone of marsh plants proved to be much less than was indicated by the concept of Reddy et al. (1989) and was only enhanced at the end of the growth season (Starink et al., in preparation). Based on these different observations the hypo- thesis can be put forward that denitrification is relatively more important in mudflats than in marshes, while in the latter nitrification largely exceeds denitrification. Nitrite concentrations are well above 0.64 μM, the Flemmish standard for fish water. While the toxic action of nitrite on fish is incompletely known, long exposure to sublethal concentrations of nitrites was reported not to cause much damage to fish (Svobodova et al., 1993). The LC50 can be influenced by various factors of which salinity is an important one. Minimal Cl/NO_2-N ratios of 8–17 (expressed as mg/mg) have been recommended for fresh water and these conditions were met in the Scheldt throughout. Although a major fish kill was spotted during the September 2002 cruise, this coincided with relatively low nitrite concentrations. In contrast, during summer 2001 when nitrite concentrations peaked, no fish kill was observed.

Phosphorus

Phosphorus (P) is a reactive element and its chemistry in aquatic systems is complex (Corell, 1998). In the Scheldt estuary P received less attention than N, probably because in coastal waters, particularly in the Dutch coastal zone, N and not P have been reported as limiting nutrient for phytoplankton (Peeters & Peperzak, 1990; Billen, 1993). Within the Scheldt estuary phytoplankton N:P ratios generally exceeded 16, indicating a surplus of N on P, although both nutrients were amply available (Van Spaendonk et al., 1993; Kromkamp et al., 1995). Nevertheless, elsewhere in this special issue, Billen et al. submitted discuss indications that nutrient limitation might have shifted from N to P.

In the Western Scheldt, particle-bound P was dominated by organic P (Zwolsman, 1994). Up till Temse in upstream direction, , Fe-bound P was the major particulate P carrier (detailed data on P speciation further upstream are lacking). The Elbe estuary showed an similar distribution of P species (Van Beusekom & Brockmann, 1998). In the freshwater part, the more complex seasonal pattern of P suggests interaction between a physicochemical buffering mechanism, biological processes and factors controlling input from the river basin (Wollast, 1982; Boderie et al., 1993; Zwolsman, 1994).

Orthophosphate and total P are decreasing since the peak concentrations in the seventies (Van Damme et al., 1995), although concentrations today are still one to two orders of magnitude larger than values expected for pristine conditions (Froelich, 1988). The decline in phosphate is attributed to the banning of phosphate based detergents (Zwolsman, 1994), improved agricultural practice and progressive industrial and municipal water treatment. Furthermore, from 1996 to 2001 increasing discharge diluted the P-load in the freshwater part (Struyf et al., 2004), but this effect was not observed further downstream. At Antwerp the total P-load was even smaller than the one in Rupel and Upper Scheldt. This is a confirmation that P must be retained within the estuary. Thus, the seasonal pattern and trend indicate that in the downstream part of the estuary P is mainly removed by phytoplankton, while more upstream physicochemical processes are the dominant processes.

Phytoplankton

Light is the predominant limiting factor for phytoplankton growth in practically the entire Scheldt estuary, nutrient limitation being almost non-existent (Van Spaendonk et al., 1993; Soetaert et al., 1994; Kromkamp et al., 1995; Cloern, 1999; Muylaert et al., 2000a). Despite lower concentrations of SPM in the Westerschelde, Chl a concentrations are lower than in the freshwater part, probably due to differences in mixing depth – photic depth ratios. Despite the light limitation, high anthropogenic inputs of N and P can eventually induce silica limitation and a subsequent dominance of non-diatom phytoplankton over diatoms, especially.in coastal zones (Schelske et al., 1983; Smayda, 1990; Smayda, 1997). At the mouth of the estuary limitation of DSi for diatoms was evidencec from several approaches: (1) summer, N:Si nutrient ratios <1 (Van Spaendonk et al., 1993; Kromkamp et al., 1995); (2) modeling indicates a DSi limitation of maximum 15% (Soetaert et al., 1994); (3) low spring DSi concentrations compared to a half-saturation constant, estimated at 1–5 μM (Van Spaendonk et al., 1993). While large amounts of DSi are biologically removed during summer, this element behaves conservatively in wintertime (Boderie et al., 1993; Zwolsman, 1994). Silica loads at the mouth of the estuary were related with discharge (Struyf et al., 2004). As DSi predominantly originates from biogeochemical reactions which set free dissolved silica from alkali and aluminosilicate minerals (Correll et al., 2000), increasing discharge resulted in increased Si input in the estuary. Furthermore, peaking discharge had a negative influence on DSi uptake by diatoms since estuarine phytoplankton communities were washed away. This, however, is not reflected in the chlorophyll concentration pattern since the estuarine phytoplankton community was replaced with one of riverine origin, not adapted to conditions prevailing in the brackish zone (Muylaert et al., 2001).

Phytoplankton primary production in estuaries plays an essential role in element cycling, water quality, and food supply to heterotrophs. As phytoplankton in the Scheldt is highly important for the food web (Herman et al., 1999), it is crucial to know how it is transferred to higher trophic levels. It was estimated that phytoplankton in the Westerschelde might be controlled by grazing (Soetaert et al., 1994), mainly by copepods, dominating the zooplankton community (Soetaert & Van Rijswijk, 1993; Soetaert & Herman, 1994). Tackx et al. (2003) showed for the dominant copepod *Eurythemora affinis*, that 80% of the food required to achieve optimal physiological condition could be obtained via grazing if 3% of the POC load consists of phytoplankton carbon (Phyto-C), conditions which might not be fulfilled during winter and when turbidity peaked. In the freshwater part Rotifers are the dominant zooplankton species, and these can withstand low oxygen conditions (Soetaert & Van Rijswijk, 1993; Muylaert et al., 2000b). Because of their feeding characteristics it is unlikely they seriously reduce phytoplankton stocks (Muylaert et al., 2000b). Such as for phytoplankton, the salinity gradient also turns out to be lethal for both saline and freshwater zooplankton communities.

Implications for estuarine management

Although a serious attempt has been made to integrate two international monitoring programmes, it is clear that further efforts are required, e.g. through a further standardization of the methods and the parameters measured (e.g. adding BOD measurements to the CEME monitoring). A 7-year study period is of course rather limited, and we will need continuous monitoring data that span periods of decades, to be able to untangle the multiple variability mechanisms and to separate anthropogenic influences from natural variability in order to understand the effects of our current use of water resources. Despite this relatively short study period some implications for management are obvious.

The Scheldt estuary is heavily burdened with organic carbon and nutrient inputs. Concerning carbon efforts focus should be on the reduction of the anthropogenic immission into the estuary. In the freshwater part a complete reduction of anthropogenic carbon immission would lead to about a 10% decrease of the SPM load during phytoplankton bloom, stimulating in that way estuarine primary production. Bacterial respiration consumes a major amount of the available oxygen. As the main source of oxygenation in the estuary is aeration (Soetaert & Herman, 1995a), creation of areas with high surface/depth ratios is advised in the zone where oxygen concentrations are minimal.

The waste water treatment plants of Brussels are nowadays under construction. It is believed that once the waste load of this major source of pollution is treated, the water quality of the Scheldt will improve consistently. However, a risk of generating a new carbon load exists. Indeed, since the water coming from Brussels via the Zenne has about 30 km to flow before it reaches the Scheldt, diffuse input of nitrate from surrounding agricultural terrain to the rehabilitated tributary would favor phytoplankton growth and induce again excessive carbon load.

Management can influence removal of nitrogen in the estuary. Expansion of intertidal or flooding areas contributes both to the reactive denitrification surface and to an increase of storage capacity and hence would increase N_2-efflux to the atmosphere. It is expected that the reactive surface will gain importance on the pelagic aspect as restoring oxygen conditions will subdue pelagic denitrification even further as it has already done.

Billen & Garnier (1997) stated that phosphate removal through water treatment is essential to prevent coastal eutrophication once the N-problem is under control. As water treatment is still expanding throughout the drainage basin, it is expected that the decreasing temporal trend for P will persist.

It is important to enhance phytoplankton growth provided that primary production is channelled to higher trophic levels. For phytoplankton, a combination of measures towards attenuation of dynamics, both of tidal dynamics and wash outs, and measures to reduce nutrients and enhance silica cycling is the best option. Silica cycling can be enhanced by creation of tidal wetlands.

Before all else higher trophic levels require improved oxygenation degree, especially in the Sea Scheldt. Filter feeders (zooplankton and macrobenthic filterfeeders) deserve priority because they pass on the primary production towards higher levels in the food web such as fish and birds.

The quantification of the relative importance of the different quality goals mentioned for restoration of ecological estuarine functioning remains a scientific challenge.

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