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Substrate origin and morphology differentially determine oxygen dynamics in two major European estuaries, the Elbe and the Schelde

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Abstract

The expansion of oxygen minimum zones (OMZ's) in estuaries can be harmful for ecology and economy, prompting the demand for expensive measures. Here we look at the oxygen dynamics in two northern temperate European estuaries, the Schelde (The Netherlands/Belgium) and the Elbe (Germany) and analysed data from the period of 2004-2009. The Schelde is characterized by two zones of increased oxygen consumption; the Elbe shows one zone of increased oxygen consumption. Despite reduction in biochemical oxygen demand in both estuaries, oxygen conditions improved in the Schelde estuary, while the oxygen minimum zone persisted in the Elbe estuary. To understand these different oxygen dynamics, we applied a one-dimensional reactive transport model to both estuaries. In the Schelde we found low oxygen concentrations to be related to organic matter and ammonium input from the major tributaries. In the Elbe, additionally to a high input of organic matter from upstream, oxygen dynamics were influenced by abrupt changes in estuarine morphology. Next, the origin of the organic matter substrate differs between the two estuaries. In the Elbe, the organic matter imported is mostly composed of algal die-off produced in the Elbe River upstream. In the Schelde the organic matter and ammonium input is mostly related to sewage input of anthropogenic origin. This implies that waste water treatment will be more effective to remediate hypoxia related problems in the Schelde than in the Elbe.

Keywords: Estuaries – Dissolved oxygen – Biochemical oxygen demand – Oxygen minimum zone (OMZ)
– Morphology – Residence time

1 Introduction

Decreasing oxygen concentrations are considered the most serious threat related to eutrophication (Carpenter *et al.* 1998, Diaz and Rosenberg 2008, Kemp *et al.* 2009, Rabalais *et al.* 2010). The threshold value for hypoxia is usually set at 2 mg O₂ L⁻¹ (63 O₂ mmol m⁻³) (Diaz 2001, Diaz and Rosenberg 2008, Conley *et al.* 2009), although there is evidence for a broader range of oxygen concentration thresholds at which ecological stress is caused (Vaquer-Sunyer and Duarte 2008). In this study a threshold value of 5 mg L⁻¹ (156 O₂ mmol m⁻³) is adopted, corresponding to the level below which fish may experience sub-lethal effects (Best *et al.* 2007, Vaquer-Sunyer and Duarte 2008). When low oxygen concentrations are reached, invertebrate and fish communities can collapse (Gazeau *et al.* 2005, Maes *et al.* 2008, Vaquer-Sunyer and Duarte 2008, Mialet *et al.* 2010) and biogeochemical changes in water column and sediment are induced (Gunnars and Blomqvist 1997). Furthermore, ecological damage is accompanied by economical loss of fishery resources (Helly and Levin 2004, Diaz and Rosenberg 2008, Vaquer-Sunyer and Duarte 2008). Although hypoxia existed through geological time, its occurrence in marine, coastal and estuarine areas increased significantly (Diaz 2001), doubling every ten years since 1960 (Diaz and Rosenberg 2008). Currently hypoxia is a worldwide phenomenon (Conley *et al.* 2009), particularly severe in many North-American and European countries (Howarth *et al.* 2011).

Estuaries are naturally eutrophic and productive systems as they concentrate water from large drainage areas in relatively small water bodies (Van Damme *et al.* 2005). However, excess in nutrients and organic matter input, following from increased human activity (industry, agriculture, sewage disposal), on top of the naturally eutrophic conditions, can lead to severe oxygen deficiencies (Rabalais *et al.* 2010). Examples of oxygen deficiencies in estuaries in Europe can be found amongst others in the Seine (France; Garnier *et al.* (2001)), Schelde (Belgium – the Netherlands; Soetaert *et al.* 2006, Van Damme *et al.* 2005), Elbe (Germany; Bergemann *et al.* 1996, Schroeder 1997, Kerner 2007, Amann *et al.* 2012), Gironde (France;

Lanoux *et al.* 2013), the estuary of Bilbao (Spain; Villate *et al.* 2013), the Ems estuary (the Netherlands-Germany; Talke *et al.* 2009), and the Tamar and Tweed estuary (UK; Uncles *et al.* 2002).

Hypoxia or oxygen deficiency develops when aerobic respiration and (bio)chemical oxidation of reduced substances cannot be compensated by oxygen supply (primary production, air-water exchange, transport) (Schroeder 1997, Rabalais *et al.* 2010). There are many biological, chemical and physical regulating factors that influence the strength, timing, frequency and location of low oxygen zones (Rabalais *et al.* 2010). Temperature and salinity affect oxygen concentrations directly and indirectly, i.e. by their control on solubility, and for temperature also by the rate at which biochemically mediated oxygen demanding and producing processes proceed (Testa and Kemp 2011). The duration and intensity of oxygen deficiencies depends on the availability and degradability of the dissolved or particulate organic carbon (Abril *et al.* 2002), on the presence of primary producers and on the availability of ammonium which may create a large oxygen demand when being nitrified (Kemp *et al.* 2009). In the 90s in the Schelde for instance, nitrification contributed to more than half of the oxygen consumption (e.g. in Soetaert *et al.* 2006). Residence time of the water is another crucial factor that determines the degree of mineralization (Rabalais *et al.* 2010) and hence oxygen consumption. Furthermore, the relationship between the euphotic zone and the total water depth is of importance, as it will influence the amount of oxygen supplied due to primary production (Wolfstein and Kies 1999).

Since the implementation of the European Water Framework Directive in national legislation, nutrient and organic load reduction have been widely applied to reverse eutrophication and associated hypoxia events (Kemp *et al.* 2009). Despite waste water treatment in both the Schelde and Elbe estuary, oxygen concentrations increased in the Schelde estuary (Van Damme *et al.* 2005, Soetaert *et al.* 2006), while oxygen deficiency situations persist in the Elbe estuary (Amann *et al.* 2012). The reasons why these estuaries respond differently to measures of nutrient reduction are not known. In the Elbe the sudden increase in depth, in and downstream of the Hamburg port area has been associated with the oxygen minimum zone observed here, as the increase in depth reduces the euphotic depth/mixing depth ratio

(Bergemann *et al.* 1996, Wolfstein and Kies 1999, Kerner 2000). Estuaries experience high pressure due to the harbour large populations and economic importance, resulting in anthropogenic changes in morphology (embankments, rectifications, channel deepening and widening), biogeochemistry (eutrophication, inputs of contaminants) and food webs (Meire *et al.* 2005). Therefore, it is important to elucidate the underlying mechanisms that may lead to oxygen depletion or hypoxia. Only then the effectiveness of current management strategies can be assessed and adjusted if necessary.

We hypothesize that in contrast to the Schelde, in the Elbe not only water quality, but also the current morphological situation, e.g. the sudden increase of average depth in the port area, causes the recurrence of oxygen deficiency situations in summer. To elucidate how water quality improvement and morphological changes differentially affect the oxygen dynamics in the Schelde and Elbe estuaries we use a one-dimensional reactive transport model and perform two types of scenarios: a reduction by half of the input of oxygen consuming substances at the boundary and tributaries, and a change in estuarine morphology, by adapting depth or width.

2 Material and Methods

2.1 Study area

The Schelde estuary (Fig.1D, The Netherlands/Belgium) has a length of approximately 160 km from the weir and sluices near Gent (51.00214° N; 3.722781 E – 0 km) to the mouth near Vlissingen (51.412110 N; 3.566041 E – 160 km) (Meire *et al.* 2005). The border is located at 102 km from the weir. The Elbe estuary (Fig.1E, Germany) has a length of about 150 km reaching from the weir at Geesthacht (53.423279 °N; 10.3347464 °E – 0 km) to the mouth at Cuxhaven (53.897798 °N; 8.6954743 °E – 150 km) (Amann *et al.* 2012). Both estuaries discharge in the North Sea, and both include a large port of international importance, the Port of Antwerp at 82 km, and the Hamburg Port area at 39 km from the weir. The river catchment area of the Schelde is about 21 500 km², while that of the Elbe is approximately 7 times larger (148 300 km²). About ten million people live in the river basin in the Schelde (465 inhabitants km⁻²), 24.5

million people live in the river basin of the Elbe (165 inhabitants km⁻²) (Meire *et al.* 2005, Amann *et al.* 2012).

2.2 Data sources

Schelde water quality data from the monitoring programs in Flanders (Maris and Meire 2016) and the Netherlands (Knuijt and Soetaert 2012) were combined. In total about 30 stations (Fig. 1D) were sampled monthly. Water quality data from the Elbe estuary were obtained from the FGG Elbe data portal (FGG 2015). In total about 25 stations (Fig. 1E) were sampled six times a year, among which monthly in summer and once every other season. Oxygen concentrations and biochemical oxygen demand data were taken for the study period extending from 2004 till 2009, for which a consistent data set was available collected in the framework of the European Interreg IVb project TIDE (Tidal River Development 2017). Unfortunately, biochemical oxygen demand was not measured in the Dutch part of the Schelde estuary. Biochemical oxygen demand defines the oxygen consumed over 5 days (Schelde) or 7 days (Elbe) for water incubated at 20 °C; it comprises oxygen needed for both mineralization and nitrification, since no inhibition for nitrification was applied prior to measurement. Spearman rank correlations of biochemical oxygen demand (BOD), particulate organic carbon (POC), dissolved organic carbon (DOC), chlorophyll *a* (chl *a*) and ammonium (NH₄); and POC/chl *a* ratios were calculated using the R-package PerformanceAnalytics to discuss the origin of the substrate in the Schelde and Elbe estuary.

Morphology data of the Schelde and Elbe, also collected in the framework of the European Interreg IVb project TIDE (Tidal River Development 2017), was obtained from the Flanders Hydraulics Research Centre and Hamburg Port Authority, respectively (Vandenbruwaene *et al.* 2012). Bathymetry for the Schelde and Elbe was based on data from the year 2009 and 2006, respectively. The Schelde grid has a resolution varying from 5 x 5 m to 20 x 20 m. The Elbe grid has a resolution of 10 x 10 m. Bathymetrical depth is expressed with respect to the local sea level reference (Fig. 1D-E), in the Schelde in m TAW (Tweede Algemene Waterpassing), in the Elbe in m NN (Normalnull). Geometry data of the Schelde

(width, depth and area) were calculated based on mean high and mean low water levels from the year 1991 to 2000. Geometry data of the Elbe were calculated based on data of the year 2006.

Freshwater discharge data for the Schelde was provided by Flanders Hydraulics Research Centre (Waterinfo 2017); freshwater discharge data for the Elbe was brought by the Wasser- und Schifffahrtsamt Lauenburg (Pegel Online 2017).

2.3 Morphology and hydrology.

The cross-sectional area and width were estimated as the mean value during mean low water level and mean high water level. Mean depth was deduced by dividing cross-sectional area by width.

The Elbe is much wider (Fig. 1A), ranging between 500 – 1 500 m, compared to 50 – 500 m in the Schelde. A prominent widening in the Elbe occurs between 50 and 100 km from the weir; the widening in the Schelde is more gradual. The mean depth in both estuaries varies between 3 and 20 m (Fig. 1B). The largest average depth is found near the mouth in the Schelde estuary and downstream the harbour of Hamburg in the Elbe estuary. The mean cross-sectional area in both estuaries varies between 250 and 7 000 m² (Fig. 1C) and follows roughly the same shape. However, a clear discrepancy can be found between 30 and 100 km, where the surface area in the Elbe is about three times larger than the Schelde. Note also that, in the Elbe, near 20 km the area drops from 2 000 m² to 1 000 m² (Fig. 1A).

The average freshwater discharge in the Elbe is about 7 times larger (700 m³ s⁻¹) than in the Schelde (100 m³ s⁻¹). In the Elbe freshwater discharge is usually highest in spring, due to snow melting at its source in the Czech Republic, reaching values as high as 3 000 m³ s⁻¹ (Weiss *et al.* 2013). In the Schelde discharges are generally higher in winter with peak values of 350 m³ s⁻¹ due to increased run-off (Meire *et al.* 2005). Proportionally, tributaries in the Schelde contribute much more to the freshwater input than in the Elbe. The Schelde tributaries Dender, Durme, Rupel (located at 39, 45 and 69 km from the weir) and lateral inflow, constitute about 70 % of the total freshwater discharge. In the Elbe the tributaries only add up to 11 % of the total freshwater input.

2.4 The biogeochemical oxygen model

A one-dimensional (1-D), tidally averaged advective-dispersive reaction model was developed and applied to the Schelde and Elbe estuary, using three state variables (Table 1).

Both estuaries are spatially compartmentalized in 100 boxes, 1.59 and 1.50 km in length per box for the Schelde and Elbe, respectively. The reactive-transport models have been implemented in R (R Development Core Team 2014) using the ReacTran package, and solved using the rootSolve and deSolve packages (Soetaert and Herman 2009, Soetaert and Meysman 2012).

2.4.1 Advective-dispersive transport

Since the vertical and lateral gradients in concentration along the length axis are negligible compared to gradients along the estuarine axis, a one-dimensional advection-dispersion equation can be used to describe transport (Soetaert and Herman 2009). This neglects settling of particulate matter, which can potentially be important for inorganic-organic aggregates, introducing vertical gradients and residual horizontal transport in biochemical oxygen demand. However, there is no observational evidence of vertical gradients in BOD in the Schelde and the Elbe (Bergemann *et al.* 1996, Maris *et al.* 2015), which justifies our assumption. It is assumed estuarine boxes are moving up- and downstream with the tide, so that only tide averaged, turbulent mixing and net freshwater flow needs to be explicitly modelled. This leads to the volumetric advective-dispersive transport for a tracer C , given by:

$$\frac{\partial C}{\partial t} = -\frac{1}{A} \cdot \frac{\partial(Q \cdot C)}{\partial x} + \frac{1}{A} \cdot \frac{\partial}{\partial x} \cdot \left(A \cdot E \frac{\partial C}{\partial x} \right)$$

Where Q is the volumetric flow rate ($m^3 s^{-1}$), A is the cross-sectional area (m^2), E the horizontal dispersion coefficient ($m^2 s^{-1}$), t is time and x is the position along the estuarine length axis.

A is assumed to be constant in time, but variable in space; Q is imposed to the model as daily values. E was obtained by fitting the chlorinity model to the observed data.

Using a 1-D model matches the data availability in both estuaries, which are both sampled along the length axis. The Schelde has been successfully modelled with a 1-D reaction-transport model (Soetaert

and Herman 1995a, b, c, Vanderborght *et al.* 2002, Hofmann *et al.* 2008). In the Elbe valuable insights have been obtained with a zero- and a one-dimensional model (Schroeder 1997, Quiel *et al.* 2011).

In the Schelde, transport and input from tributaries the Dender, Durme and Rupel (at 39, 45 and 69 km respectively) is accounted for in the model. Tributary input in the Elbe is assumed to be negligible. In the Scheldt, effluents of large waste water treatment plants (WWTP) and canals are either indirectly included via the tributaries (e.g. Brussels via the Rupel, Ghent via Upper Scheldt), or they enter the estuary far downstream the oxygen minimum zone (Antwerp). Smaller lateral inputs are negligible compared to the estuarine load (Deneudt *et al.* 2003). Due to the large discharge of the Elbe, WWTP-inputs have small impact and are neglected (Hamburg Wasser 2017). Upper boundary concentrations in the Schelde model are set to concentrations observed at a monitoring station located 7 km upstream of the weir in Gent. In the Elbe model, the upper boundary concentrations are set to concentrations observed at the Geesthacht monitoring station at the weir (0 km). Seaward boundary concentrations in the Schelde model are set to observed concentrations at the monitoring station at Vlissingen (160 km). In the Elbe the seaward boundary concentrations are set to observations at Cuxhaven (151 km). In the Schelde there were no estimates of biochemical oxygen demand concentrations available at the seaward end; a concentration of biochemical oxygen demanding units of $2 \text{ mg O}_2 \text{ L}^{-1}$ ($63 \text{ O}_2 \text{ mmol m}^{-3}$) was assumed based on historical observations at Vlissingen (de Pauw 1971).

2.4.2 Oxygen dynamics

The oxygen model simulates the coupled dynamics of two state variables: biochemical oxygen demand units (BODU) and dissolved oxygen (O_2) (Soetaert and Herman 2009). Biochemical oxygen demand units represent any substance that can easily be oxidized and is expressed by the amount of oxygen needed for breakdown ($\text{mmol O}_2 \text{ m}^{-3}$). Reaeration brings the oxygen concentration back to saturation. BODU decay is described as first-order to BODU, hyperbolic in O_2 (Monod constant of oxygen limitation ks_{O_2}).

The coupled dynamics that define the oxygen concentration over time ($\frac{\partial O_2}{\partial t}$) are defined by oxygen consumption ($\frac{\partial BODU}{\partial t}$) and oxygen production (*Reaeration*), and are described as follows:

$$\frac{\partial BODU}{\partial t} = -r \cdot BODU \cdot TempRate \cdot \frac{O_2}{O_2 + K_{SO_2}}$$

$$\frac{\partial O_2}{\partial t} = -r \cdot BODU \cdot TempRate \cdot \frac{O_2}{O_2 + K_{SO_2}} + Reaeration$$

Where r = first-order decay rate of biochemical oxygen demand units (d^{-1}) at 20 °C

Temperature was imposed to the model as monthly values. To account for a temperature dependent modulation of the decay rate a Q10 formulation is used, assuming a doubling in rate for each increase in temperature with 10 °C (Soetaert and Herman 2009).

$$TempRate = \exp \left[\left(\frac{T-20}{10} \right) \log(Q10) \right]$$

Where $Q10 = 2$

Reaeration is modelled proportionally to the over- and under-saturation of oxygen. The piston velocity is parameter, sometimes further parameterized as function of wind speed and tidal currents, defining the gas transfer velocity across the air-water interface, in relation to over-or undersaturation of O_2 in the water (Ho et al.. 2011). Here, we use a single value for the piston velocity, representative for average wind and tidal currents.

$$Reaeration = \frac{k}{D} \cdot (O_2sat - O_2)$$

Where k : is the piston velocity ($m d^{-1}$), D the average depth (m), O_2sat : oxygen saturation concentration ($mmol m^{-3}$)

Although its description is principally used for air-water exchange, it also represents oxygen production via photosynthesis, since photosynthesis is also inversely related to water depth and the model data is fitted to the observed data. The dissolved oxygen saturation concentrations are estimated as a function of temperature and salinity using the R-package marelac (Soetaert *et al.* 2016). The BOD measurements measure the oxygen consumption of a sample, for a period $t = 5$ or 7 days (in the Schelde and the Elbe respectively) at 20 °C (BOD_t^{20}). In order to compare the modelled BODUs with the observed BOD values,

and to use the BOD data as boundary conditions for the modelled quantities, we need to calculate one from the other. Thus, the biochemical oxygen demand is calculated by integrating BODU decay over the requested period at 20 °C, obtaining:

$$BOD_t^{20} = -BODU(exp^{-rt} - 1)$$

$$BODU = -\frac{BOD_t^{20}}{(exp^{-rt}-1)}$$

2.4.3 Model calibration

The dispersion coefficient E, the piston velocity k and the first-order decay rate r of BODU are model parameters that have been derived from model calibration, so that model output variables (O₂, BOD and CL) reproduce the observed variables for 2004 to 2009 as close as possible. The dispersion coefficient for the Schelde was split in two parameters, one for the upstream stretch that is oriented North-South, the other of the downward stretch oriented East-West; both separated a 111 km from the weir. This gave a better fit of the chlorinity model to the data. Calibration was performed in such a way, that both slope and regression coefficient of the model versus data comparison was as close to one as possible.

2.4.4 Model simulations

A reference run (**R**) and **two types of scenarios** were tested for the Schelde and Elbe estuaries:

The first scenario (**S1**) quantifies water quality improvement by reducing the riverine input of biochemical oxygen demand units with 50%.

The second scenario (**S2**) involves two types of change in morphology (Fig. 1A-C, Fig. 6A-B).

In **S2a**, the depth profile along the estuarine axis was adapted. The section with large average water depth in the Elbe near Hamburg was removed, and replaced by a constant shallow depth of 6 m from 30 to 50 km. Inversely a section with large average water depth (~15m) was imposed to the Schelde by replacing its depth profile with that of the Elbe between 30 and 50 km

In **S2b**, the width profile along the estuarine axis was adapted. An abrupt increase in width was imposed to the Schelde, resembling the situation in the Elbe. In the Elbe a more gradual increase was prescribed, making it similar to the Schelde. More specifically, widths in the Schelde from 40 to 80 km were doubled. In the Elbe a more gradual increase was established by recalculating width between 30 and 120 km using a smooth fit to the width at the extremities of the estuary (0 – 30 km, 120 – 151 km). Effects of these scenarios are described by changes in oxygen concentrations, oxygen consumption ($\frac{\partial BODU}{\partial t}$), oxygen production (*Reaeration*), morphologies and residence times in the estuary. Each change is expressed as relative or absolute change with respect to the reference scenario, per zone distinguished (zones are described in more detail in '3.1 Oxygen and biochemical oxygen demand in Schelde and Elbe'). We compared summer 2007 in the two estuaries. Summer, because oxygen deficiencies are more pronounced when temperatures are higher. Year 2007, because it was an average discharge year for the six-year period studied in both estuaries, and the year in which the two zones of increased oxygen consumption in the Schelde estuary were still visible before further diminishing in 2008 and 2009. The summer season refers to the average for June, July and August.

3 Results

3.1 Oxygen and biochemical oxygen demand in Schelde and Elbe

Observed oxygen concentrations vary between 19 and 453 mmol O₂ m⁻³ and between 28 and 559 mmol O₂ m⁻³ in the Schelde and Elbe respectively. In the Schelde highest oxygen concentrations are consistently observed near the mouth area (Fig. 2A). In the Elbe highest concentrations are observed downstream the tidal weir where waters are often oversaturated (Fig. 2C). In the Elbe water from the upper boundary comes in oversaturated (> 100 %, Fig. 2C). In the Schelde water from the upper boundary and Rupel comes in highly under-saturated (Fig. 2A). Dissolved oxygen concentrations are generally lower in the Schelde than in the Elbe. In both estuaries, dissolved oxygen concentrations are lower in summer than in winter. However, seasonality is much more pronounced in the Elbe estuary.

Setting a threshold of $156 \text{ mmol O}_2 \text{ m}^{-3}$ (5 mg L^{-1}) on the dissolved oxygen concentrations, the Schelde exhibits one large zone of low oxygen concentrations, extending to 100 km downstream the weir of the estuary. Over time this large zone gradually reduces to a smaller zone near the inflow of the Rupel tributary. There are two peaks of increased oxygen consumption and the first peak seems to be resolved over the course of the study period. By 2009 all values below $156 \text{ mmol O}_2 \text{ m}^{-3}$ in the Schelde have disappeared (Fig. 2A). In the Elbe the oxygen minimum zone recurs every year, only slightly changing in concentration, spatial extent and timing. For convenience of further interpretations, the upper Schelde estuary is divided in two oxygen minimum zones. One ranges from 0 to 50 km near the upper boundary (OMZ 1); another ranges from 50 to 95 km near the Rupel tributary (OMZ 2). In the Elbe only one oxygen minimum zone is observed from 30 to 80 km near the Hamburg port area (OMZ). Zones before and after the oxygen minimum zone, are referred to as pre- and post-oxygen minimum zone (pre OMZ and post OMZ).

Biochemical oxygen demand varies between 22 and $656 \text{ mmol O}_2 \text{ m}^{-3}$ and between 22 and $484 \text{ mmol O}_2 \text{ m}^{-3}$ in the Schelde and Elbe respectively. The spatio-temporal distribution of biochemical oxygen demand is rather patchy in the Schelde, and clearly decreases in time (Fig. 2B). BOD is generally high in summer and low in winter. In 2009 BOD measurements in the Schelde decreased to $155 \text{ O}_2 \text{ mmol m}^{-3}$. From then on, a reversed BOD pattern is observed in the Schelde; with low values in summer, and high values in winter. Biochemical oxygen demand patterns in the Elbe estuary are similar in all consecutive years. Biochemical oxygen demand is highest in summer, immediately downstream the tidal weir where water depth is about 5 meters. Biochemical oxygen demand decreases over time in the Elbe (Fig. 2D), but this decrease is less pronounced than in the Schelde and mainly observed in the most upstream part of the estuary. For the whole study period, average biochemical oxygen demand within the estuary is nearly two times higher in the Schelde than in the Elbe estuary (212 and $114 \text{ mmol O}_2 \text{ m}^{-3}$ respectively). Biochemical oxygen demand coming from the upper boundary and tributaries is also higher in the Schelde than in the

Elbe. Actually, the difference is even higher considering the 7 days incubation time in the Elbe versus the 5 days incubation time in the Schelde.

3.2 Substrate origin

To elucidate the difference in substrate between Schelde and Elbe, correlations between BOD, dissolved inorganic carbon (DOC), particulate organic carbon (POC), ammonium (NH_4^+) and chlorophyll *a* (chl *a*) have been studied (Annex I). In the Schelde BOD ($t = 5$) is highly correlated to NH_4^+ ($r = 0.68$), and to a lesser extent to DOC ($r = 0.45$), POC ($r = 0.39$) and chl *a* ($r = 0.31$). POC is mainly correlated to chl *a* ($r = 0.72$), DOC to NH_4^+ ($r = 0.65$).

In contrast, In the Elbe BOD ($t = 7$) is highly correlated to chl *a* ($r = 0.84$), and to a lesser extent to DOC ($r = 0.48$), NH_4^+ ($r = 0.39$) and POC ($r = 0.17$). DOC ($r = 0.64$) and POC ($r = 0.58$) are both highly correlated to chl *a*, and much less to NH_4^+ ($r = 0.15$). All correlations described are highly significant ($p < 0.001$).

The POC/chl *a* ratio is in general much higher in the Schelde compared to the Elbe (for all zones considered in Fig. 3, from 2004 to 2009, a median of 234 in the Schelde). In contrast, in the Elbe POC/chl *a* ratios show a marked difference between the upper parts of the estuary and the lower end of the oxygen minimum zone (OMZ). While the median ratio of POC/chl *a* in the boundary, pre OMZ and beginning of the OMZ (at 43 km) is 41; similar ratios to the Schelde can be observed at the end of the OMZ (212, at 74 km). These findings indicate the more variable labile and sewage related character of the BOD in the Schelde versus the algal derived substrate origin in the Elbe.

3.3 Model calibration

Model data is calibrated to the observed data, by manually adapting the model parameters within ranges frequently reported in literature. The observed data and calibrated model data (O_2 , BOD, CL) were plotted together to validate at least eight sampling stations (Annex II.I, II.II and II.III). Model parameters and overall linear model fits (slope and R^2) are represented in Table 2. To obtain a similar good fit, a higher decay rate was needed in the Elbe estuary.

Overall, the scatter of the fit between modelled and observed dissolved oxygen concentrations is higher in the Schelde than in the Elbe estuary (Fig. 4A, C; Annex II.II). Fits for chloride and biochemical oxygen demand can be found in Annex II.I and II.III. In the Schelde, the actual oxygen pattern is more complex than the modelled one. Nevertheless, dissolved oxygen patterns are relatively well predicted with comparable model parameters in both estuaries. One large zone of low oxygen concentrations with two peaks of increased oxygen consumption is simulated in the Schelde estuary (Fig. 4B), one smaller zone with one peak of oxygen consumption is simulated in the Elbe estuary (Fig. 4D), corresponding to the oxygen observations (Fig 2A, C). However, contrary to the observations in the Schelde, the large zone of oxygen depletion near the inflow of the Rupel tributary does not shrink to a smaller zone in time. In the Elbe modelled oxygen minima are less severe than the oxygen minima observed.

3.4 Oxygen dynamics and biochemical oxygen demand under different scenarios

The different types of scenarios are explained in detail in '2.2.4 Model simulations'. Each change is expressed per zone, as relative or absolute difference with respect to the reference scenario (R).

3.4.1 Reducing biochemical oxygen demand input (S1)

When the input of biochemical oxygen demand substances is lowered by half at the upper boundaries and tributaries, the dissolved oxygen sag is reduced (Fig. 5C-D). In the Schelde, oxygen concentrations almost double in both minimum zones whereas in the Elbe the oxygen concentration in the oxygen minimum zone increases by 20 %. Considering the absolute difference, the oxygen concentration in the Schelde rises by 87 mmol O₂ m⁻³ in the first, and by 40 mmol O₂ m⁻³ in the second oxygen minimum zone. In the Elbe estuary, oxygen concentration increases by 42 mmol O₂ m⁻³ in the oxygen minimum zone.

3.4.2 Morphological measures (S2)

Morphological measures modify oxygen patterns by geometry in two ways: by the effect of cross-sectional surface area on transport rates, and by the depth dependence of the reaeration term.

3.4.2.1 Adapting depth of the estuary (S2a)

Adding a section with increased average water depth in the Schelde estuary causes a prominent drop of 27 % (or 23.6 mmol O₂ m⁻³) in dissolved oxygen concentrations in the first oxygen minimum zone (Fig. 5C). Concomitantly oxygen consumption and oxygen reaeration decrease (Fig. 5A, E). Therefore, the decreasing oxygen concentration is not the effect of a decrease in residence time (which would involve an increase in O₂ consumption), but the effect of a decreased impact of reaeration on the oxygen concentrations due to the larger depth of the water mass. Removal of the section with large average depth in the Elbe estuary increases dissolved oxygen concentrations by 6 % (or 12.0 mmol O₂ m⁻³, Fig. 5D) and only marginally affects oxygen consumption and oxygen reaeration (Fig. 5B, F).

3.4.2.2 Adapting width of the estuary (S2b)

Introducing an abrupt widening in the Schelde estuary (from 40 to 80 km) causes the dissolved oxygen concentrations to rise there where the morphological measure is taken, increasing oxygen concentrations up to 55.2 and 170.0 mmol m⁻³ in OMZ 1 and OMZ 2 respectively (Fig. 5B). Oxygen concentrations remain clearly higher than in the reference scenario downstream. Replacing the abrupt widening by a more gradual widening (from 30 to 120 km) in the Elbe estuary shifts the oxygen minimum 20 km further downstream. Oxygen concentrations at the oxygen minimum zone increase by 29 %, but drop by 16 % in the post oxygen minimum zone of the Elbe estuary when compared to the reference model situation (Fig. 5C). Oxygen consumption decreases by more than half in the Schelde estuary, while it rises more than twofold in the Elbe estuary, when introducing an abrupt or more gradual widening respectively (Fig. 6A-B). Reaeration in the Schelde estuary decreases together with oxygen consumption by more than half (Fig. 5E). Reaeration in the Elbe estuary decreases in the oxygen minimum zone, although it increases in the former post oxygen minimum zone. Likewise the minimum dissolved oxygen concentration shift, maximum reaeration moves 20 km further downstream the Elbe estuary (Fig. 5F).

3.5 Residence times under different scenarios

We estimated the residence time in each kilometre by dividing the mean freshwater discharge of summer 2007 by the mean cross-sectional area (Fig. 6C-F), for the different morphological scenarios (Fig. 6A-B). This ignores the effect of mixing on residence time, but mixing coefficients were similar in both estuaries. Next, its influence will be limited in the upper part of the estuary. In general residence times are much higher in the Schelde estuary. Starting at the sluice in the summer of 2007, it takes about 446 days and 84 days for a dissolved substance to reach the sea in the Schelde and Elbe estuary respectively. Introducing a deepening and abrupt widening increases residence times even more in the Schelde estuary according to the location where the measure is taken. In the Elbe estuary, removal of depth and introducing a more gradual widening similar to the Schelde estuary reduces the residence time further according to the location where the measure is taken.

4 Discussion

4.1 Oxygen patterns and trends: same outcome, different causes

A large number of studies have reported the Schelde to be one of the most heavily polluted estuaries in Europe (Frankignoulle *et al.* 1998, Abril *et al.* 2002, Middelburg and Herman 2007). Low oxygen concentrations have been attributed to the input of high loads of organic matter and ammonium (Van Damme *et al.* 2005, Soetaert *et al.* 2006), largely from anthropogenic sources (Van Damme *et al.* 2005). As waste water treatment efforts increased from 1990 onwards, oxygen concentrations gradually restored (Van Damme *et al.* 2005, Soetaert *et al.* 2006). Mid 2000 wastewater treatment was finally implemented in the Brussels region, and after 2007 this became general practice for the entire catchment (Brion *et al.* 2015). The system abruptly shifted from a hypereutrophic to a classical eutrophic state which allowed photosynthesis to take place again (Cox *et al.* 2009). Our data confirm the previous reports on gradual oxygen restoration in the estuary for the period 2004 to 2009, although low oxygen concentrations are still found. Improvements in sewage treatment occurred about a decade earlier in the Elbe estuary, when Eastern and Western Germany were re-united and the input of pollutants from chemical plants in former

Eastern Germany decreased or even stopped (Bergemann *et al.* 1996, Weiss *et al.* 2013). In contrast with the Schelde, notwithstanding this effort, low oxygen concentrations as reported since the first measurements in the 80ies (Bergemann *et al.* 1996, Schroeder 1997) are still frequently observed in summer today.

There are distinct differences in the oxygen dynamics in the Elbe and the Schelde that point to this discrepancy. First of all, the oxygen minimum zone in the Elbe is highly predictable, both in space and time: it is located downstream the Hamburg port area, and mainly prominent in summer. In the Schelde, the zones of increased oxygen consumption seem to be located near the effluent of the Schelde with the major tributaries (Middelburg and Herman 2007, Vanderborght *et al.* 2007, this study). The oxygen minimum zone is more diffuse in space, while it is also quite unpredictable in time. Waters flowing in the Schelde already carry a burden of low oxygen and, within the estuary itself, this oxygen deficit is gradually restored towards saturation. BOD input from the Rupel alone increases oxygen consumption by 64 % in the second oxygen minimum zone. In contrast, waters flowing in the Elbe estuary are oversaturated and they lose their surplus of oxygen in the estuary. These observations point to a main difference in the nature of the substrate that consumes oxygen in both systems.

Derived from anthropogenic sources in the Schelde (Abril *et al.* 2002, Van Damme *et al.* 2005, Soetaert *et al.* 2006), its input through tributaries is largely dependent on what happens in the catchment area, hence the erratic nature. Input is high all year round, and the labile fraction of organic matter is variable as demonstrated by the diverse significant correlations of biochemical oxygen demand to ammonium, organic carbon and chl *a* in the Schelde estuary (Annex I). Although largely decreasing within the estuary, at the upper boundary and Rupel tributary of the Schelde estuary the contribution of ammonium remains high. About 30 to 50% of the biochemical oxygen demand input in 2009 is still attributed to the oxidation of ammonium (Maris *et al.* 2015). In contrast in the Elbe, ammonium only contributed up to 15 % of the oxygen consumption in 1995 (Bergemann *et al.* 1996, Kerner 2000). Assuming organic matter mineralisation at the boundary is defined by one mole of oxygen and nitrification of ammonium by two

moles of oxygen, it is found that less than 10 % of the biochemical oxygen demand is needed for nitrification in 2009. As indicated by the oversaturation of the incoming water and the particularly high correlation between biochemical oxygen demand and chlorophyll *a* (Annex I) in the Elbe, the oxygen minimum zone (OMZ) is rather caused by the mineralization of riverine phytoplankton, following peak primary production, both in time and in space (Bergemann *et al.* 1996). The input of biochemically reactive organic matter (BOD) is highest in spring, when phytoplankton blooms coincide with high freshwater discharge due to snow melting in the upper part of the Elbe River (Bergemann *et al.* 1996, Quiel *et al.* 2011). Quiel *et al.* (2011) demonstrated that if chlorophyll *a* values are reduced in the middle and upper Elbe River, a positive effect on oxygen concentrations can be observed downstream. The prominent role of algae in oxygen consumption in the Elbe is corroborated by POC/chl *a* ratios being as low as 41 up to 43 km, indicating the algal origin of POC. At 74 km the POC/chl *a* ratio increases up to 212, indicating a large scale breakdown in the OMZ (Fig. 3; Abril *et al.* 2002).

4.2 Model validity and applicability

For an estuary driven by consistent water input such as the Schelde and Elbe, the longitudinal gradients by far overwhelm the transversal and vertical gradients, so the monitoring of the biogeochemical data is positioned along the longitudinal gradient in both systems. Likewise to monitoring campaigns that are set up to capture the main environmental gradients, mechanistic models can be simplified to focus on the main features. Our model approach similarly considers only the spatial gradients along the estuarine length axis. Although two-dimensional models have been developed (Böhme *et al.* 2006, Vanderborcht *et al.* 2007, Arndt *et al.* 2009, de Brye *et al.* 2010) for both the Schelde and Elbe estuary, also fairly good one-dimensional model applications appeared repeatedly in literature (Soetaert and Herman 1995a, b, c, Vanderborcht *et al.* 2002, Hofmann *et al.* 2008, Quiel *et al.* 2011). Further model simplifications are to ignore any effects that occur within a tidal cycle, but rather approach the dynamics in a tidally-averaged parameterization. Another simplification is to describe the oxygen dynamics by the BOD (biochemical

oxygen demand) concept that lumps all respiration in one simple oxygen consuming process, and a reaeration term that brings oxygen towards saturation.

The advantage of these simplifications is that the model has only three parameters to be calibrated: the dispersion coefficient (E), the first-order biochemical oxygen demand unit decay rate (r), and the piston velocity (k). In the Elbe a biochemical oxygen demand decay rate of 0.10 d^{-1} fits the data best, while in the Schelde a decay rate of 0.05 d^{-1} is needed to obtain an overall good fit. Soetaert and Herman (1995c) also used a decay rate of 0.05 d^{-1} in the Schelde, but other higher values have been used (up to 0.15 d^{-1} for the fast decaying fraction in Vanderborght *et al.* 2007 and Hofmann *et al.* 2008; up to 0.25 d^{-1} in the freshwater part in Cox *et al.* 2009). It is likely that the anthropogenic signature of oxygen consuming substances in the Schelde is not easily captured using one decay rate, as clearly oxygen concentrations are much better fitted in some regions than in others. Previous Schelde model studies often use a slow and fast organic matter decay rate (e.g. Hofmann *et al.* 2008). Furthermore, decay rates might have changed over the study period considered since employment of the waste water treatment plant in Brussels in 2007, greatly reducing and possibly changing the degradability of the input of BOD near the Rupel tributary (Brion *et al.* 2015). The fit of the model to the data is overall much better for the Elbe, in line with the predominance of algal derived organic matter decay, typically composed of a more homogenous labile fraction (Abril *et al.* 2002). The values we used on the Elbe (0.1 d^{-1}) are similar to previous estimates (0.06 d^{-1} up to 0.08 d^{-1} for fast decaying organic matter in Schroeder 1997).

The two main processes that restore low oxygenated waters towards saturation and that are not measured in the BOD bottle experiments are primary production and air-water exchange. Reaeration is modelled by fitting a piston velocity (k), which is inversely related to water depth. Best-fit values were very similar in the Schelde and Elbe, 1.6 and 1.7 m d^{-1} respectively. Since the model is fitted to the data observed, the model indirectly includes primary production, inversely related to depth as well. Nonetheless, lumping both processes together oversimplifies the more complex situation in the Schelde estuary. Autochthonous primary production in the Schelde (Cox *et al.* 2009) is indeed more important than in the Elbe estuary

(Quiel *et al.* 2011), further explaining the larger mismatch between modelled and observed oxygen concentrations in the Schelde when compared to the Elbe.

Neither oxygen consuming, nor oxygen producing processes are included in detail. Organic matter mineralisation and nitrification are combined by the biochemical oxygen demand term. Primary production and air-water exchange are combined by the reaeration term. Obviously caution is required to use the quantitative estimates for both the Schelde and Elbe estuary. Nevertheless, the qualitative agreement with the data (Annex II.I, II.II and II.III) suggests that the models are valuable as first tools in evaluating causes and possible remediation of oxygen problems in both estuaries.

4.3 Water quality improvement: limitations of waste water treatment by estuarine hydro-morphology

For both estuaries reducing the BODU input by half led to the most significant improvements compared to morphological changes. In the Schelde, such a reduction can be reached improving wastewater treatment. Assuming half of the BOD input in the Schelde is anthropogenic (e.g. Van Damme *et al.* 2005), a reduction of the effluent of waste water treatment plants would result in a 25 % decrease of BOD input. This might be achieved if stronger standards for effluents are applied. This would certainly still be beneficial for the Schelde near the Rupel tributary. Import from the Zenne in the Rupel remains problematic, since at times more than half of the discharge consists of WWTP effluent from Brussels with, following much lower standards for BOD (25 mg L⁻¹; Maris *et al.* 2015). Hence, it might be not diluted enough before reaching the Schelde estuary via the Rupel tributary. In the Elbe it would require reducing algal-derived material input from the Elbe River upstream, which is not that easily achieved.

Algal growth can be controlled bottom-up or top-down. The Elbe estuary is limited in phosphorus (N/P>16) and lowering phosphorus concentrations below 2.82 mmol m⁻³ has been suggested as the first measure to control phytoplankton growth (Bergemann *et al.* 1996, Kerner 2000). Although the effectiveness of this measure is debated since algal cells are suspected to store phosphorus (Quiel *et al.* 2011). Furthermore, effects of phosphorus removal in the water column might be delayed, due to the

storage capacity of the sediment (Gunnars and Blomqvist 1997). Ultimately algal growth could be reduced, if nitrogen concentrations are severely lowered by applying ‘good agricultural practices’ in the middle and upper part of the Elbe River, as required by the Water Framework Directive (2000/60/EC). Top down control of phytoplankton in the middle and upper part of the Elbe River upstream the estuary, is dominated by a large number of rotifers (13 0000 individuals L⁻¹). However, grazing of rotifers is presumably negligible and food supply is high (Holst *et al.* 2002, Zimmermann-Timm *et al.* 2007), so that particulate and dissolved organic matter generated is transformed, accumulated, cycled and respired through the microbial loop, *in situ*, or after being transported downstream (Azam *et al.* 1983). Furthermore, a direct, strong tropho-dynamic coupling between phytoplankton and bacterial production rate in summer (1993, ‘95 and ‘97) is demonstrated in the Elbe by Karrasch *et al.* (2001).

Removal of the section with large average depth in the Elbe estuary slightly increases oxygen concentrations (Fig. 5D). It seems part of the organic matter is transported and decays further downstream. However, this effect might be underestimated, as primary production could increase when depth is lowered, while it is not explicitly included in the model. Introducing a more gradual widening in the Elbe estuary narrows the estuary upstream. There is no more time for the organic matter to decay. All organic matter is transported downstream, and oxygen deficiencies develop downstream as well. In the Elbe estuary residence times are much smaller than in the Schelde estuary (Fig. 6C-F). It only takes one day for the Elbe River discharge to reach the deep waters in the Hamburg port area. The location of the oxygen minimum zone coincides with an increase in residence time (Fig. 6D). The effect of residence time is illustrated very well by the extreme response of the Elbe estuary to the more gradual widening scenario (Fig. 5D, Fig. 6D, F).

Adding a section with increased average water depth in the Schelde estuary causes a local, concomitant drop in oxygen concentrations, more prominent than the increase in the Elbe when removed, because of a lowered reaeration flux (Fig. 5E). Widening the Schelde estuary increases the oxygen concentrations, since the increased residence time (Fig. 6C, E) decreases the total biochemical oxygen consumption per

volumetric unit almost halving its impact on the estuarine oxygen concentrations (Fig. 5A). In the Schelde estuary, the effects of morphological measures on the oxygen pattern are again more complex, because of larger residence times, higher primary production rate and the additional tributary input.

5 Conclusions and outlook

In the Schelde the occurrence of an oxygen minimum zone appears to be largely related to sewage input, coming from both the upper boundary and tributaries. In the Elbe this is mostly related to riverine phytoplankton input from the upper boundary. By consequence, the oxygen deficiencies in the Schelde can develop all year round, but they are most pronounced in summer when temperatures are higher. In the Elbe the formation of the OMZ is triggered in spring together with the peak import of allochthonous algal organic matter, and is most pronounced in summer. Because of the difference in substrate between the Schelde and Elbe estuary, the oxygen minimum zones are more easily controlled by waste water treatment in the Schelde. Indeed control of riverine phytoplankton in the Elbe seems to be more challenging, as the effect of phosphorus limitation is argued and grazing of riverine phytoplankton is dominated by smaller rotifers, which are less easily transferred to the higher trophic levels.

Not only the type and amount of organic matter input, but also the interaction of morphology and hydrology affects the location and intensity of the oxygen minimum zone through residence time. For decades estuarine morphology is strongly affected by various human alterations (embankments, deepening, dredging...). Channel deepening in the Elbe has often been associated with the oxygen minimum zone in relationship to the decreasing euphotic, mixing depth ratio (Bergemann *et al.* 1996, Kerner 2000). The plan to further deepen the navigation channel in the Elbe estuary has been delayed for more than ten years. A court procedure was launched, amongst others because of concerns that the plan would affect objectives of the Bird (2009/147/EC) and Habitat Directives (92/43/EEC); and because of concerns that the oxygen concentrations would decrease, compromising the good ecological state as agreed upon in the Water Framework Directive (2000/60/EC). In this study we show that the location of the oxygen minimum zone in the Elbe is mainly confined to the zone where the deepening increases,

because of the time the waters spends in this specific stretch of the estuary. Residence time in the Elbe estuary is much lower than in the Schelde estuary and seems to form a critical threshold. Further deepening might increase the intensity of the oxygen minimum zone, through increased oxygen consumption, increased residence time and lower reaeration. Lowering the residence times, by narrowing, will shift the organic matter breakdown activity downstream. In the Schelde residence times are much larger, and the organic matter degrades from the moment it enters the estuary. Widening the Schelde estuary would further increase oxygen concentrations, since the total amount of oxygen consumed per volume is halved. Deepening the Schelde estuary would reduce oxygen concentrations, due to lowered reaeration. Overall, the Schelde seems more sensitive to morphological measures simulated, because of the higher residence times and the lower cross-sectional affecting the transport and reaeration rates. However contrary to the Elbe estuary, autochthonous algal populations are able to build up in the Schelde estuary counterbalancing oxygen consumption, because of the higher residence time.

To understand how each of these measures simulated influences nitrification, oxic mineralisation, primary production and reaeration separately, each process should be included and calibrated individually. Moreover, the model should also contain the exchange with the sediment and information on phosphorus and nitrogen limitation.

Waste water treatment is likely to reach its maximum capacity and introduction of shallow water areas might form another measure to increase oxygen concentrations, as it may be a source of reaeration and an *inoculum* for increased *in situ* primary production. Several managed realignment measures have already been performed or planned in the Schelde and Elbe estuary (e.g. Lippenbroek in the Schelde, Kreesand/Spadenlander Busch in the Elbe; Boerema *et al.* 2013). Making a back-of-the-envelope calculation for the Elbe estuary, to increase oxygen concentrations with 31 mmol m^3 (1 mg L^{-1}) by solely reaeration, a shallow water zone with a depth of 0.4 m and a surface of more than 4.000 ha would be required (assuming a piston velocity of 1.7 m s^{-1} and supposing the water is exchanged twice a day). If oxygen supply by primary production within this area contributes equally to an oxygen concentration

increase as reaeration, still about 2.000 ha is needed. Making the same calculation for the Schelde estuary using the same assumptions, a surface of 219 ha with an average depth of 1.4 m would be sufficient. Considering the high load that needs to be reaerated in the Elbe, a shallow water zone ten times larger than in the Schelde, would need to be connected to the estuary. Hence, first crude estimate reveals large scale investments would be needed to prevent any oxygen deficiency situation to recur in the Elbe estuary.

In conclusion, reducing nutrients remains the most effective strategy increasing oxygen concentrations within the estuary channel. More detailed studies are needed to examine the effect of alternative measures, such as the effect of “good agricultural practices” and connecting shallow water zones to the estuary. These measures seem to be more challenging in estuaries characterized by high river discharge (coinciding with the productive season) and low residence times, both related to the estuary’s specific hydro-morphology.

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581	Annexes
582	Annex I Correlation analyses of biochemical oxygen demand and its influencing components
583	Annex II.I Chloride model calibration
584	Annex II.II Dissolved oxygen model calibration
585	Annex II.III Biochemical oxygen demand model calibration
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764

Tables

Table 1 State variables used in the biochemical model

State variable	Abbreviation	Unit
Biochemical oxygen demand	BOD	mmol m ⁻³
Oxygen	O ₂	mmol m ⁻³
Chlorinity	CL	mmol m ⁻³

Table 2 Model parameters and model fit after calibration

Model parameter	Schelde	Elbe
Dispersion (E; m ² s ⁻¹)	-	170
<i>Upstream Bath</i>	144	-
<i>Bath – sea</i>	180	-
First-order rate of biochemical oxygen demand (r; d ⁻¹)	0.05	0.10
Piston velocity (k; m d ⁻¹)	1.6	1.7
Q10 (-)	2	2
Slope of model fit and R ²	Schelde	Elbe
CL	1.07 (R ² = 0.98)	1.07 (R ² = 0.92)
O ₂	1.00 (R ² = 0.94)	1.00 (R ² = 0.99)
BOD	0.97 (R ² = 0.85)	1.00 (R ² = 0.95)

772
773 **Fig. 1 Estuarine morphology: average width (A), depth (B) and area (C) measured at mean water level and bathymetry of**
774 **the Schelde (D) and Elbe (E) estuaries. On top of the bathymetry maps the sampling stations for water quality are shown.**
775 **Bathymetry is expressed with respect to the local used sea level reference, TAW (*‘Tweede Algemene Waterpassing’*) in the**
776 **Schelde, NN (*‘Normalnull’*) in the Elbe. The coordinates are according to the ETR S89 geodetic system using the UTM 31**
777 **zone.**

Fig. 2 Observed oxygen concentrations and biochemical oxygen demand in Schelde (A, B) and Elbe estuary (C, D) from 2004 to 2009 from the upper tidal limit (0 km) to the estuary mouth. Thresholds are marked by a line in the surfer plot: 5 $\text{mg O}_2 \text{ L}^{-1}$ ($156 \text{ O}_2 \text{ mmol m}^{-3}$) for oxygen; 6 and 6.7 $\text{mg O}_2 \text{ L}^{-1}$ (188 and $210 \text{ mmol O}_2 \text{ m}^{-3}$) for biochemical oxygen demand in the Schelde and Elbe for 5 and 7 days resp. In the Dutch part of the Schelde estuary biochemical oxygen demand was not measured. Three zones are indicated by dotted lines: OMZ 1, OMZ 2 and post OMZ in the Schelde; pre OMZ, OMZ and post OMZ in the Elbe (limits in distance are described in the text).

Fig. 3 Boxplot of the ratio of particulate organic carbon to chlorophyll *a* (POC/chl *a*) for the upper boundary of the Schelde (*n* = 48) and the Elbe (*n* = 29), tributaries the Dender (*n* = 48) and Rupel (*n* = 47) in the Schelde, pre-oxygen minimum zone (*n* = 59) in the Elbe, oxygen minimum zones (OMZ) at 43 km (*n* = 28) and 74 km (*n* = 29) in the Elbe, OMZ 1 (*n* = 50) and OMZ 2 (*n* = 48) in the Schelde. Each boxplot indicates P25, the median (P50) and P75. The upper whisker = $\min(\max(\text{POC/chl } a), P75 + 1.5 * \text{IQR})$, the lower whisker = $\max(\min(\text{POC/chl } a), P25 - 1.5 * \text{IQR})$ and IQR = interquartile range = $P75 - P25$.

Fig. 4 Calibration of modelled to observed oxygen concentration data in Schelde and Elbe. A, C: linear fit between observed and modelled oxygen concentration; B, D: distribution of modelled oxygen concentrations from 2004 to 2009, contour lines represent the 156 mmol O₂ m⁻³ threshold for hypoxia.

Fig. 5 Average summer 2007 model simulations for the Schelde and Elbe estuaries, effects on oxygen consumption (O₂ consumption, mmol m⁻³d⁻¹), the oxygen concentration (O₂, mmol m⁻³) and reaeration (O₂ reaeration, mmol m⁻³d⁻¹).

Fig. 6 A, B: Estuarine cross-sectional area per box unit (m^2); **C, D:** average summer 2007 residence time (days) calculated as freshwater discharge divided by crosssectional area per box unit, and **E, F:** average summer 2007 cumulative residence time (days), in the Schelde and Elbe estuary, C-F: on a logarithmic scale (log scale).

Annex I

Correlation analyses of biochemical oxygen demand and its influencing components

Schelde

Elbe

Annex II.I

Chloride model calibration

Schelde

Elbe

Annex II.II

Dissolved oxygen model calibration

Schelde

Elbe

Annex II.III

Biochemical oxygen demand model calibration

Schelde

Elbe