hypoxia in the highly turbid Gironde estuary, applying a 3D model coupling hydrodynamics, sediment transport and biogeochemical processes

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Abstract

Estuaries are increasingly degraded due to coastal urban development, and prone to hypoxia problems. The macro-tidal Gironde estuary is characterized by the presence of a highly concentrated Turbidity Maximum Zone (TMZ). The water quality monitoring data show hypoxia occurring in the TMZ, under conditions of low river flow, increasing water temperature and during the transition from spring to the neap tide. *In-situ* data also highlights that summer hypoxic is particularly pronounced around the city of Bordeaux, located in the upper estuary. Interactions between these multiple factors limit the understanding of processes controlling the dynamics of dissolved oxygen (DO).

In this study we developed a 3D biogeochemical model coupling hydrodynamics, sediment transport and biogeochemical processes, to assess the contribution of the TMZ and the impact of urban effluents through wastewater treatment plant (WWTP) and sewage overflow (SO) on the hypoxia. Our model describes the transport of solutes and suspended material, the major biogeochemical mechanisms impacting oxygen: primary production, degradation of natural and anthropogenic organic matter, nitrification, and surface exchange. The composition and the degradation rates of each variable have been characterized by *in-situ* measurements and from experimental data performed in the study area. The DO model was calibrated and validated against observations, and simulate DO dynamics at two times scales (seasonal and neap-spring time scale) in Bordeaux city.

The simulated time series of DO concentrations shows a good agreement with field observations and reproduces satisfactorily the seasonal and neap-spring time scale variations around the city of Bordeaux. Simulations show a strong spatial and temporal correlation between the formation of summer hypoxia and the location of the TMZ, with minimum DO centered in the vicinity of Bordeaux. To understand the contribution of anthropogenic forcing, we compare different simulations with the presence or absence of urban effluents. Our results show that a reduction of POC loads by eliminating SO would increase DO concentration minimum in the vicinity of Bordeaux by 3 % of saturation in summer; omitting both SO and WWTP discharge would improve DO concentrations in summer by 10 % of saturation and mitigate hypoxic events.

1. Introduction

Hypoxic areas are increasing globally and becoming a major environmental problem in shallow water estuaries (Diaz et Rosenberg, 2008; Diaz, 2001). Hypoxic waters are defined as areas where the concentration of dissolved oxygen (DO) is less than 2 mg L^{-1} (~20-30% of air saturation or 62.5 µmol L^{-1}, e.g., Rabalais et al., 2010). DO is an indicator of water quality in estuaries as it is essential for the health of aquatic ecosystems. Indeed, hypoxic zones cause ecosystem problems for benthic and pelagic fauna (Diaz, 2001; Gray et al., 2002).

Hypoxia is a consequence of an increase in eutrophic areas induced by high concentrations of nutrients and organic substances delivered by rivers and human activities (Rabalais et al., 2010; Verity et al., 2006). This supply of material perturbs the biogeochemical cycle in aquatic systems. When oxygen consumption by community respiration is greater than the replenishment of DO by the atmosphere, vertical mixing or photosynthesis, DO depletion occurs (Conley et al., 2009). In temperate eutrophic estuaries, the input of nutrients and organic material intensifies primary production in spring and summer, producing organic matter in surface waters. Moreover, anthropogenic nutrient and organic matter enrichment intensifies hypoxic events by the rapid decay of labile organic matter during warmer months. Depending on the physical structure of estuarine waters, contrasting mechanisms drive the occurrence of hypoxia. In stratified estuaries, the settling of particulate organic matter and associated heterotrophic processes cause summertime DO depletion in bottom waters, as in Chesapeake Bay (Hagy et al., 2004) and the plume of the Yangtze River (Li et al., 2002). In contrast, in well-mixed estuaries, hypoxia is less likely to occur due to wind and tide mixing driving DO replenishment from the atmosphere. In turbid estuaries, photosynthesis is low and DO depletion can be caused by the degradation of organic matter, which is associated with the suspended sediment (Lanoux et al., 2013; Talke et al., 2009; Thouvenin et al., 1994).

Hypoxia events occur in the well-mixed and macro-tidal Gironde Estuary, characterized by the presence of a turbidity maximum zone (TMZ) resulting from the asymmetry of the tidal wave in the upper estuary coupled to the density residual circulation (Allen et al., 1980; Brenon and Le Hir, 1999). The Gironde Estuary is a heterotroph ecosystem (Abril et al., 2002, 1999; Etcheber et al., 2007; Irigoien and Castel, 1997) with oxygen concentrations relatively close to saturation in the river water and at the mouth of the estuary and between 20 and 70% saturation in the TMZ (Abril et al., 1999). Phytoplankton grows mainly in the river and at the mouth of the estuary, and the heterotrophic bacterial population grows preferentially in the TMZ (Goosen et al., 1999). The turbidity limits primary production due to light attenuation in the water column and favors DO consumption by heterotrophic processes (Goosen et al., 1999). In addition, turbidity limits gas exchange in the TMZ (Abril et al., 2009). DO depletion occurs in the water column of the TMZs of many European macro-tidal estuaries (Amann et al., 2012; Garnier et al., 2001; Lanoux et al., 2013; Soetaert et al., 2006; Talke et al., 2009; Thouvenin et al., 1994; Tinsley, 1998; Villate et al., 2013). These high turbidity zones play a key role in the sedimentation and biogeochemical processes that control the environmental water quality (Abril et al., 2000, 1999).

The analysis of 7 years of DO data from the water quality monitoring of the Gironde Estuary (Etcheber et al., 2011) showed that summer hypoxia events occur in the TMZ coincident with lower river discharge and higher water temperature. Moreover, DO minima occur a few days after the spring tide (ST) peak, with lower concentrations near the city of Bordeaux in the upper estuary (Lanoux et al., 2013). Various factors explain the hypoxia events in the Gironde Estuary, including temperature, river flow, turbidity and urban effluents, but we do not know the relative contribution of each factor. Increasing turbidity, water temperature and urban discharge, together with decreased flushing due to low river flow, could lead to severe summer hypoxia in the coming years. Therefore, it is essential to find solutions to mitigate these hypoxia events, for example, by reducing organic matter and nutrient inputs or improving wastewater management (Kemp et al., 2009). In the Thames and Scheldt Estuaries, the installation of wastewater treatment plants (WWTPs) notably increased oxygen concentrations and eliminated hypoxic zones (Amann et al., 2012; Soetaert et al., 2006; Tinsley, 1998).

The aims of this paper are to assess the impact of TMZs on the size and position of the hypoxic area and to assess the respective contribution of organic matter from the river watershed (WS) and that of organic matter and ammonia from the local urban watershed (WWTP and sewage overflow (SO)) to estuarine hypoxia. Biogeochemical models are essential tools for understanding ecosystem function. They enable the identification of diverse factors that control the dynamics of hypoxia (Lopes and Silva, 2006; Scully, 2013; Talke et al., 2009). Moreover, biogeochemical models can be used as a tool for effective estuarine management by providing guidelines for setting goals to improve water quality (Cox et al., 2009; Justić et al., 2007; Skerratt et al., 2013; Vanderborght et al., 2007; Wild-Allen et al., 2009). We implemented a 3D biogeochemical model coupled with a hydrodynamics and sediment transport model to reproduce the DO cycle in the Gironde Estuary. Previous works have demonstrated that the TMZ in the Gironde Estuary is characterized by strong vertical sediment concentration gradients, which are maintained by salt-induced density stratification (Sottolichio et al., 2000). As these gradients cannot be reproduced using a depth-averaged model, it is necessary to couple the DO model with a three-dimensional sediment transport model.

2. Materials and Methods

2.1. Study area

The Gironde Estuary is located in Southwest France on the Atlantic Coast, near the city of Bordeaux. It is formed by the confluence of the Garonne River (65% of the freshwater input) and the Dordogne River (35% of the freshwater input) and drains a watershed of 81 000 km² (Figure III.1). It is the largest European estuary, with a surface area of 635 km². The Gironde Estuary is shallow, with a depth range from 5 to 8 m in the upper estuary (upstream of PK25) and a depth of up to 20 m in the lower estuary (downstream of PK25) and near the mouth (Sottolichio and Castaing, 1999). The annual mean river flow is 680 m^3 s^{-1} (sum of Garonne and Dordogne Rivers calculated over the last 10 years (2005-2014); Jalón-Rojas et al., 2015). According to Allen (1972), the Gironde is a partially mixed to well-mixed macro-tidal estuary. The tidal amplitude at the mouth varies from 2.5 m at neap tides (NT) to 5 m at spring tides (Allen et al., 1980). The tidal wave propagates up to 180 km from the estuary mouth to La Réole on the Garonne River (PK-70, 95 km from the river confluence) and to Pessac-sur-Dordogne on the Dordogne River (90 km from the river confluence). The residence time of water in the estuary ranges from 20 to 86 days, depending on the fluvial regime (Jouanneau and Latouche, 1981).

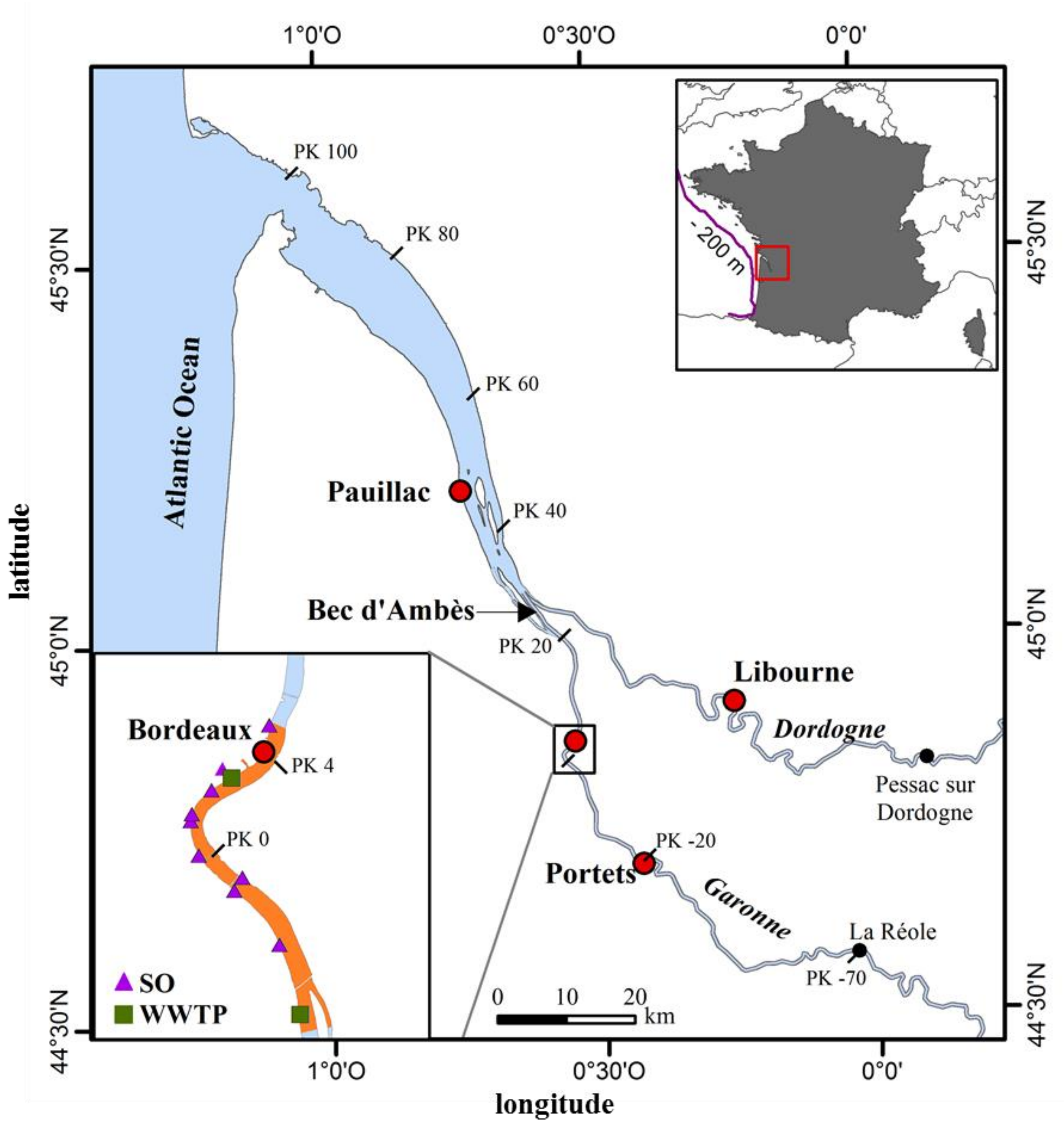


Figure III.1: *The Gironde Estuary in Southwest of France. The distance is noted by kilometric points (PK), with PK100 the mouth and PK-70 the upstream limit. Red points represents water quality monitoring station, with PK4 the location of Bordeaux station. The area in orange represents the area of Bordeaux where the biogeochemical fluxes are calculated. The purple line represents the 200 m isobath.*

In the TMZ, the suspended sediment concentration (SSC) in the surface waters varies between 0.1 and 10 g L^{-1}. The particle residence time is estimated to be between 12 and 24 months (Saari et al., 2010). The TMZ migrates longitudinally throughout the semi-diurnal tidal cycle with the ebb and flood currents and throughout the year with river flow variation and a shift in salinity intrusion. In winter, under high river flow, the TMZ is located downstream at Pauillac City, approximately 45 km from the mouth (Doxaran et al., 2009). When the river flow decreases, the TMZ moves upstream, oscillating from the junction of the two rivers (Bec d'Ambes) to Portets on the Garonne River and to Libourne on the Dordogne River, i.e., 100 km from the mouth (Castaing and Allen, 1981) (Figure III.1). The vertical distribution of the SSC is controlled by the tide. At slack water, mainly at neap tides, suspended matter settling occurs and an anoxic fluid mud appears at the bottom, forming elongated patches with concentrations greater than 100 g L^{-1} (Abril et al., 1999). The particulate organic carbon (POC) content in the TMZ is relatively constant at 1.5%, with the major constituent being refractory soil material (90%) with smaller contributions of biodegradable litter and autochthonous phytoplanktonic material (Etcheber et al., 2007).

The sewage system of Bordeaux City, located on the Garonne River (Figure III.1), drains an urban area of 578 km² and services an estimated population of 749,595 in 2015. There are two large wastewater treatment plants that discharge treated water comprised mainly of ammonia, and 9 sewage overflows that discharge untreated water mainly consisting of POC (Figure III.1) (Lanoux, 2013).

To understand the dynamics of the TMZ and the variation in DO, continuous water quality monitoring systems were installed at Pauillac, Portets and Libourne in 2004 and at Bordeaux in 2005 (Figure III.1). These stations contain real-time systems (updated every 10 minutes) that measure 4 parameters: temperature, salinity, turbidity and DO, and continue to collect data to the present time (Etcheber et al., 2011). The values recorded between January 2005 and October 2006 (http://www.magest.u-bordeaux1.fr) were used in the current study to validate the numerical model, which will be explained in more detail in section III.2.5.

2.2.Hydro-sediment model

The SiAM-3D (Simulation for Multivariable Advection) model was initially developed to simulate the dynamics of estuarine suspended sediment and associated biogeochemical variables. A detailed description of the sediment transport model and its first application can be found in Brenon and Le Hir (1999) for the Seine Estuary and in Cugier and Le Hir (2002) for the Bay of Seine. The hydrodynamics model solves the Navier-Stokes set of equations (momentum, continuity and state equations) under the Boussinesq approximation and the hydrostatic assumption in the vertical direction.

External and internal modes are separated to solve the set of equations. The external mode determines the free surface elevation and depth-integrated velocity (U and V) from the 2D St-

Venant equation. Then, the free surface elevation is introduced into the internal modes to determine the three velocity components (u, v, w). The turbulent model uses the eddy viscosity concept based on mixing length theory. The damping of turbulence by density stratification is taken into account by empirical functions using a Richardson number.

The transport mode solves the advection/dispersion equations for the dissolved and particulate variables, i.e., salinity, suspended sediment and biogeochemical variables. The equation for suspended sediment accounts for the main cohesive sediment processes: erosion, deposition, flocculation, consolidation and hindered settling (Appendix Table A1, Van Maanen and Sottolichio, 2013). The deposition flux is calculated using the Krone (1962) formulation, and the erosion flux, by the Partheniades (1965) formulation (Appendix Table A1).

2.3.Biogeochemical model

The biogeochemical model is based on the model previously developed by Thouvenin et al. (1994) for the Loire Estuary. The biogeochemical model is fully coupled to the SiAM-3D hydrodynamic and sediment transport model and simulates biogeochemical processes that produce or consume oxygen in the water column. These biogeochemical processes include the degradation of organic carbon (POC and DOC), nitrification, photosynthesis, respiration and mortality of phytoplankton, and DO gas exchange with the atmosphere (Figure III.2). For each layer and grid cell, the time evolution of each model variable (**C**) is the sum of conservative advection, diffusion and sinking processes ($\varphi\boldsymbol{C}$) and biogeochemical rate processes ($\beta_{\boldsymbol{C},}$ Appendix Table A2):

$$\frac{\partial \boldsymbol{C}}{\partial t} = -\,\varphi\boldsymbol{C} + \beta_{\boldsymbol{C}} + S_{\boldsymbol{C}}$$

with

$$\varphi\boldsymbol{C} = \frac{\partial(u_i\boldsymbol{C})}{\partial x_i} - \frac{\partial\left(K_i\frac{\partial\boldsymbol{C}}{\partial x_i}\right)}{\partial x_i}$$

where **C** is the variable concentration, $S_{\boldsymbol{C}}$ the external input, K_i is the turbulent diffusivity coefficient, and ui is the current in the x_i direction with i = x, y or z.

The dissolved variables are advected and diffused similarly to salinity. The particles settle and can be re-suspended similarly to suspended sediment. The model includes 11 state variables (Figure III.2): dissolved oxygen (DO), ammonia (NH_4^+), nitrate (NO_3^-) and particulate and dissolved organic carbon (POC and DOC) from the watershed and wastewater. For the watershed sources, the model includes the POC from litter (POC_{litter}), which is the biodegradable fraction of terrestrial POC eroded from the Garonne and Dordogne watersheds (Etcheber et al., 2007; Veyssy, 1998), and the DOC from rivers ($DOC_{fluvial}$) (Veyssy, 1998). For wastewater, the model includes the POC and DOC from wastewater treatment plants (POC_{WWTP} and DOC_{WWTP}) and from sewage overflows (POC_{SO} and DOC_{SO}) (Lanoux, 2013). The model also considers POC from phytoplankton ($POC_{phytoplankton}$) and detritus ($POC_{detritus}$) (Etcheber et al., 2007; Lemaire, 2002; Lemaire et al., 2002).

Modifications from the previous version of the biogeochemical model were introduced to include photosynthesis and nitrification processes and to differentiate the degradation rates of organic matter from rivers and urban watersheds. The organic carbon mineralization rates are derived from experimental data obtained using material from the Garonne watershed (litter, k_P^{litter} ; fluvial, $k_D^{fluvial}$), phytoplankton (k_R and $k_P^{detritus}$) (Etcheber et al., 2007; Lemaire, 2002) and the Bordeaux sewage network for WWTP (k_P^{WWTP} and k_D^{WWTP}) and SO (k_P^{SO} and k_D^{SO}) (Lanoux, 2013). These organic carbon mineralization rates account for the high lability of material from urban effluent and for the more refractory character of organic carbon from the watershed (Lanoux, 2013; Lemaire et al., 2002; Appendix Table A4). In addition, the boundary conditions of each biogeochemical variable are derived from in situ data obtained (in the previous years) from the Garonne and Dordogne Rivers, the Gironde Estuary, and the sewage network of the city of Bordeaux (Table III.1). Phytoplankton primary production is limited by nutrient availability ($f(N)$) and light attenuation ($f(I)$) in the water column due to turbidity (Irigoien and Castel, 1997), and we account for preferential uptake of ammonia vs nitrate (α_N) (Appendix Table A3). The molar C/N (Appendix Table A4) ratio of organic matter, which controls the relative rates of organic carbon mineralization and ammonification, is set to 10 for litter and wastewater (Middelburg and Herman, 2007) and to the Redfield value of 6 for phytoplankton. The nitrification of NH_4^+ consumes 2 moles of DO, giving 1 mole of NO_3^-. (Appendix Table A4). The formulation of the gas exchange rate (k_{aera}, Appendix Table A3) is derived from floating chamber measurement of the CO_2 flux in the Gironde Estuary (Abril et al., 2009) and takes into account the magnitude of the current velocity, wind, the surface area of the estuary (fetch effect) and SSC (turbulence attenuation).

Consistent with previous field and experimental research (Abril et al., 2010, 2000, 1999) highlighting slow anaerobic carbon remineralization rates in the fluid mud, seabed oxygen consumption is simulated using a simple constant benthic oxygen demand. Denitrification and Mn reduction are the major respiratory pathways in the fluid mud, forming NH_4^+ and Mn(II) in low concentrations. Owing to the modest volume of the estuary occupied by the fluid mud pools (maximum of 10% of the water column in the TMZ at neap tides), the oxidation of inorganic reduced species during resuspension events has a negligible effect on water column oxygenation, even at spring tide (Abril et al. 1999). Thus, we simplify the fluid mud biogeochemistry and apply a POC degradation rate 10 times slower than in the water column. However, the supply of particulate organic matter by fluid mud resuspension has a major effect on the water column oxygenation in the model by increasing the SSC by a factor 5 to 10 at spring tides compared to neap tides. Finally, we do not implement the denitrification process in the biogeochemical model because denitrification does not interact directly with the dissolved oxygen balance. The previous fieldwork studies (Abril et al., 2000) shows too, that even if denitrification occurs in the fluid mud, it likely has a modest impact at the estuarine scale, as nitrate concentration is first increased by nitrification in the low salinity region of the TMZ and then behaves relatively conservatively along the Gironde Estuary salinity gradient.

A summary of the biogeochemical equations and parameters values is included in Appendix table A2, A3 and A4.

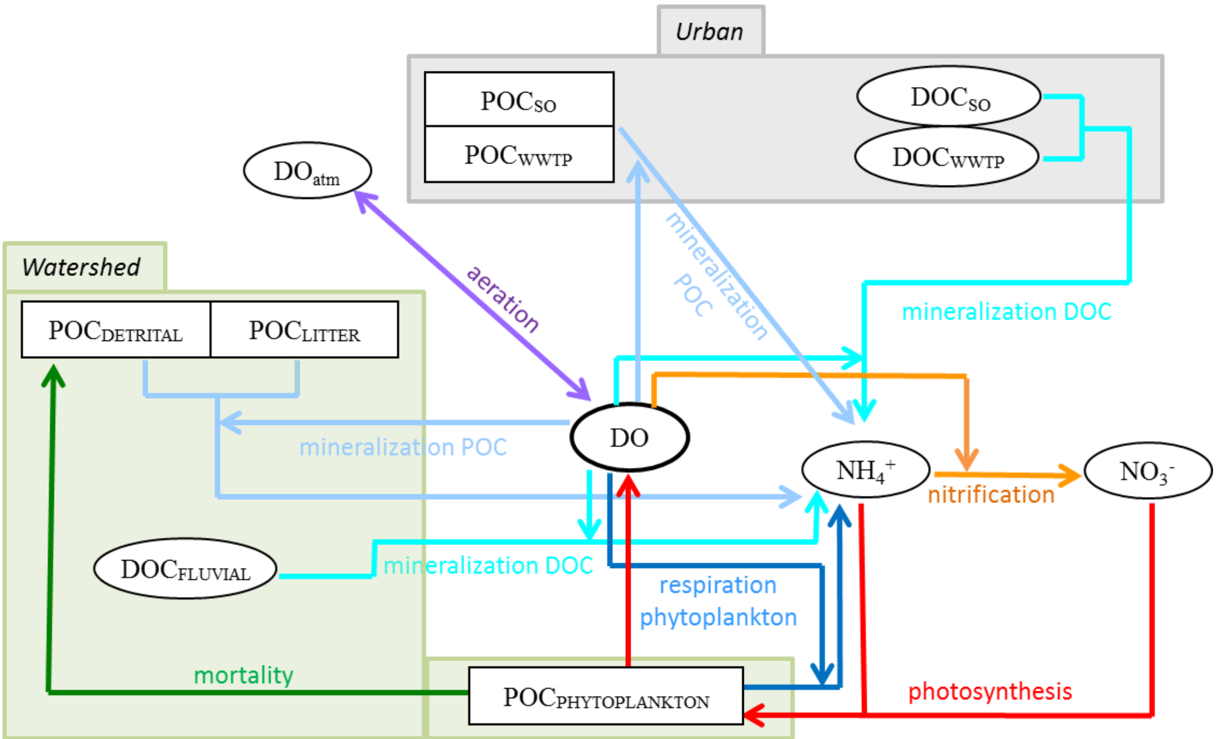


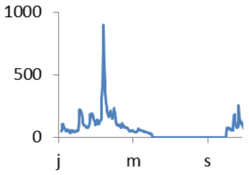
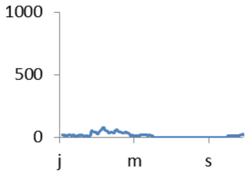
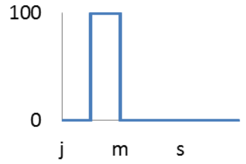
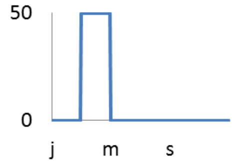
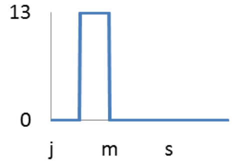
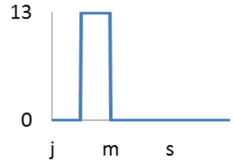
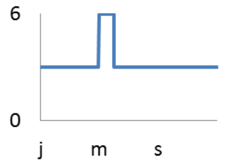
Figure III.2: *Schematic of biogeochemical model developed for the Gironde Estuary.*

2.4.Model implementation

The computational domain, defined in (Sottolichio et al., 2000) extends from the 200 m isobath on the continental shelf to the upstream limits of the tidal propagation on both rivers (Figure III.1). The model is implemented for the Gironde Estuary on an irregular rectilinear grid (2421 wet cells in the horizontal), with finer resolution in the estuary (200 m x 1 km) and coarser resolution on the shelf. The vertical grid uses real depth coordinates. The vertical axis is split into 12 layers bounded by fixed horizontal levels, with progressively finer resolution from the bottom to the free surface. In the area of interest in the Garonne tidal river around Bordeaux, the water column is described 2 m thick layers, which is satisfactory for estuarine sections (see Cugier and Le Hir, 2002 for more detail). The tidal rivers are represented by one cell in width but are discretized vertically and longitudinally. The spatial resolution in the longitudinal direction is 1 km on the Garonne River and between 1 and 4 km on the Dordogne River, with finer resolution in the upper reaches. The model uses a finite difference numerical scheme, with a transport time step of 35 s and simulations run with a run-time to real-time ratio of ~184:1. The model is forced with tidal elevation at the shelf, which is provided by a harmonic composition. At the upstream limit of the Garonne and Dordogne Rivers, a daily river flow is imposed (data available from HydroBank, www.hydro.eaufrance.fr). It is initialized using a total fine sediment budget of 3.84 million tons and a horizontal salinity gradient (Van Maanen and Sottolichio, 2013). The hydro-sedimentary model was previously validated in the lower estuary and was found to perform well against observations of tide, currents, salinity and SSC (Benaouda, 2008; Sottolichio and Castaing, 1999; Van Maanen and Sottolichio, 2013). Forcing of the seasonal water temperature (water quality monitoring, http://www.magest.u-bordeaux1.fr), wind and light intensity variation (Météo France) is included for accurate simulation of the DO dynamics.

For the biogeochemical model, the nutrient, organic carbon and phytoplankton concentrations at the boundaries are derived from Veyssy (1998) and Abril (unpublished data), and the oxygen concentrations at the boundaries are set to 90 %sat in the Garonne and Dordogne Rivers and at 100 %sat in the sea water (Abril et al., 1999) (Table III.1). Urban wastewater discharge points are included in the model: 9 SOs and 2 WWTPs (Figure III.1). The concentrations of POC, DOC and ammonia from the urban effluents are from Lanoux (2013) (Table III.1), and the discharge volume is from the wastewater management company SUEZ, which operates the wastewater network by delegation from the Bordeaux metropolis.

Table III.1: *Boundary conditions and point sources of variables in µmol.L^{-1}*

Variable	WWTP 1 Louis Frague	WWTP 2 Clos de Hilde	SO	Garonne River	Dordogne River	Sea	Initial
POC$_{litter}$*	0	0	0			0	0
POC$_{phyto}$*	0	0	0			0	0
POC$_{detritus}$*	0	0	0			0	0
DOC$_{fluvial}$*	0	0	0	200	200	0	0
POC$_{WWTP}$**	1717	584	0	0	0	0	0
DOC$_{WWTP}$**	1207	734	0	0	0	0	0
POC$_{SO}$**	0	0	6333	0	0	0	0
DOC$_{SO}$**	0	0	1250	0	0	0	0
NH_4^+*	1875	1512	214		3	2	0
NO_3^-*	0	0	0	120	100	15	0
DO***	0	0	0	90%sat	90%sat	100%sat	100%sat

* data Veyssy (1998).

** data from Lanoux (2013)

*** data from MAGEST

2.5. Model validation

The model is used to understand the DO dynamics, with a focus on the Garonne River at Bordeaux, particularly during summer conditions, to support management decisions. Following the recommendations of Rykiel (1996), to validate the performance of a sufficient DO model for our scientific purposes, three criteria are selected:

1. The model reproduces the timing of the observed DO fluctuations at two time scales: seasonal and neap-spring time scales, in the vicinity of Bordeaux City.

2. The model reproduces the observed DO amplitude in the vicinity of Bordeaux City.

3. The model vs observed Willmott Skill Score (WSS) is greater than 0.7 in the vicinity of Bordeaux City.

For the statistical validation, we calculate the bias and WSS (Willmott, 1982) for the water level, salinity, and SSC at the two water quality monitoring network stations located on the Garonne River that were mentioned in section III.2.1 for the years 2005 and 2006 (Figure III.3 and Table III.2). We compare the results of model simulations from the 1st of January to the 30th of October 2006 with observational data from the continuous MAGEST network in Figures III.3, III.4 and III.5. This period is used because 2006 was a dry year and a significant hypoxic event occurred in summer. In addition, continuous monitoring data were of high quality for this year with few gaps.

Table III.2: *Statistical validation table for 2005-2006, N = number of measure, WSS = Wilmott Skill Score. Units of bias are in the units of the parameters: water level in m, SSC in g.L^{-1} and DO in %sat.*

		Water level	**Salinity**	**SSC**	**DO**
Bordeaux	**N**	394733	62230	62294	62220
	Bias	0.07	-0.68	-0.76	10.26
	WSS	0.93	0.78	0.53	0.77
Portets	**N**	/	57726	60869	45811
	Bias	/	-0.12	-0.44	-4.27
	WSS	/	0.38	0.33	0.39

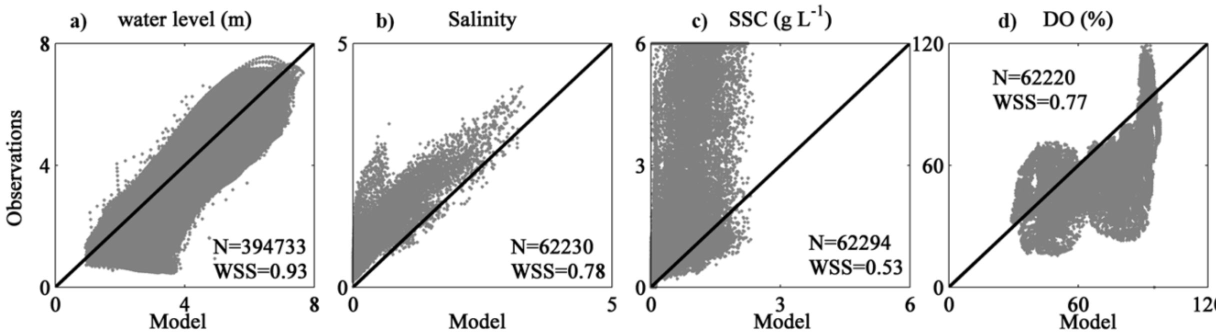


Figure III.3: *Observed versus simulated variables in Bordeaux for years 2005 and 2006: (a) water level in m, (b) salinity, (c) SSC in g.L^{-1}and (d) DO in %sat.*

As stated in section III.2.2, the sediment transport model was previously validated in the lower estuary only, i.e., downstream of the confluence of the Garonne and Dordogne Rivers. The fluvial portion of the estuary (Garonne tidal river) is addressed here for the first time. Therefore, validation of the simulated tide and SSC is performed simultaneously with the validation of the biogeochemical model. The previous hydrodynamic calibration (based on the adjustment of bottom friction) is maintained to simulate the tides and currents, but a new paramaterization is applied for suspended sediments. In particular, we include a dependence of settling velocity ws on salinity to account for deflocculation processes in brackish and freshwater (Appendix, Table A1). Salt effects are negligible in the salty lower estuary but are expected to become significant in the tidal river. Thus, ws varies between 0.1 and 2 mm s^{-1} in the lower estuary and can decrease to 0.02 mm s^{-1} in the area of salinity < 5 psu. These lower values are optimized to ensure relatively high SSC by reducing deposition fluxes and to maintain the TMZ within the river by limiting seaward SSC dispersion (Van Maanen and Sottolichio, 2013).

We can validate the transport based on available measurements data, which records turbidity below the surface, at a station near the bank. As the model calculates the width-averaged SSC, some discrepancies between the model and measurements are anticipated. Figure III.4b shows the evolution of the simulated and measured SSC in Bordeaux. At the start of the reference simulation in winter conditions, the river flow is high (Figure III.4a) and the TMZ is located in the lower estuary (Figure III.6a), so surface SSC is minimal in Bordeaux (Figure III.4c). After flood peaks in February, March and April, the river flow progressively decreases, reaching the minimum value at the end of August (Figure III.4a). The SSC at Bordeaux starts to increase between May and June and reaches its maximum values between July and September, when TMZ is shifted in the upper estuary (Figure III.6b). In summer, the simulated SSC is low compared with the observations, even when using the optimized settling velocity. As already mentioned, the measurement is performed in a shallow area of the shore, near a tidal flat, which tends to increase SSC; by contrast, in the model, the volume of the grid cell integrates the channel, where SSC can be 2-3 times lower. In addition, turbidity is measured in NTU and is converted to g L^{-1} using an empirical calibration. A simple conversion is used (10000 NTU= 6 g L^{-1}) based on the laboratory calibration reported by Jalón-Rojas et al. (2015), for which overestimations are possible due to the high uncertainties of suspended properties and sampling. Despite these discrepancies, the dynamics and relative variability of SSC are accurately reproduced by the model at the seasonal scale (Figure III.4c) and at the fortnightly tidal scale (Figure III.5c), with an increase in SSC during the spring tide.

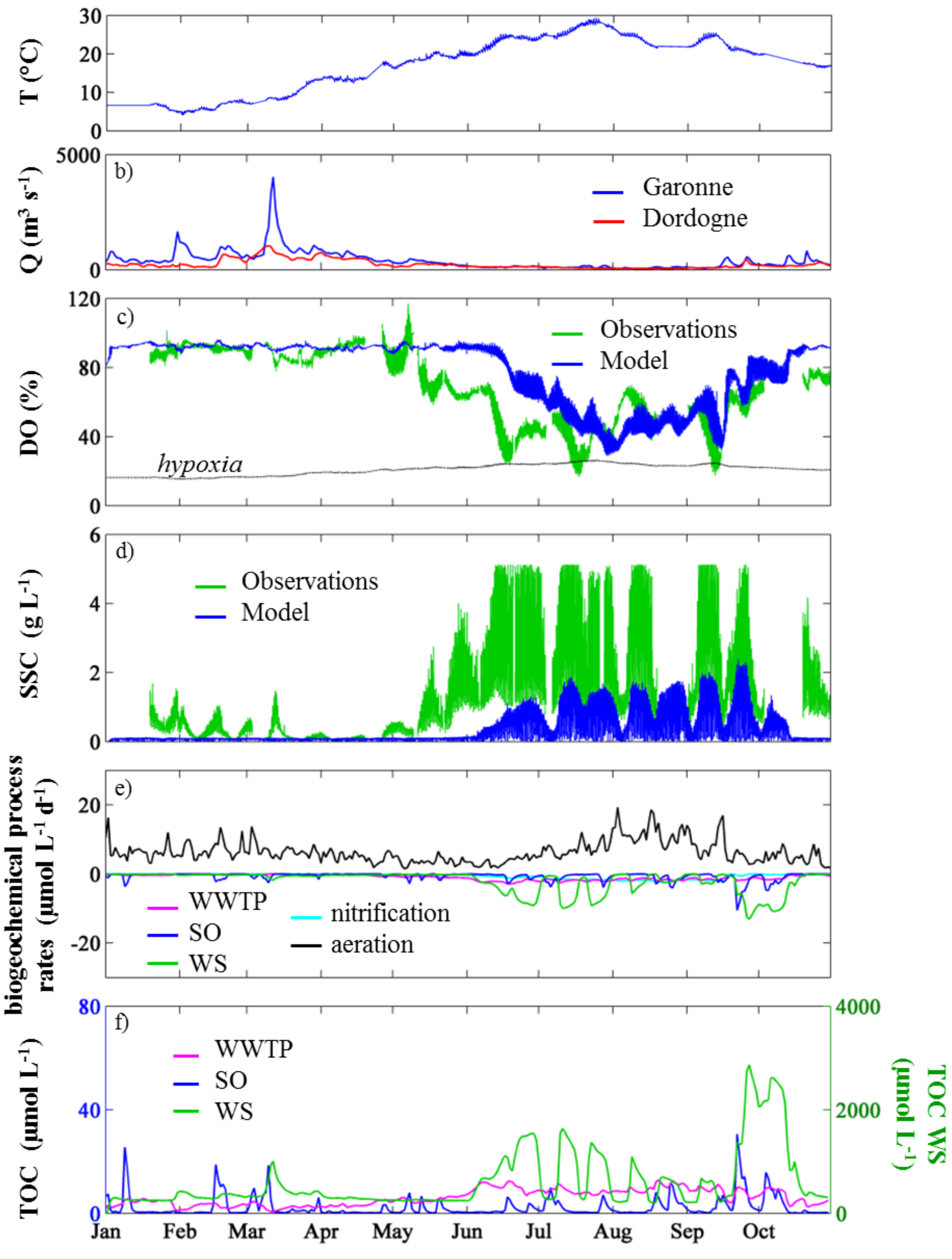


Figure III.4: *Temporal evolution between the 1st of January and 31st of October 2006 at Bordeaux of: temperature (a), river flow in m^3 s^{-1} (b), simulated (blue) and observed (green) DO concentration in %sat (c), simulated (blue) and observed (green) SSC in g L^{-1} (d). In the vicinity of Bordeaux, the daily average biogeochemical process rates affecting DO in µmol L^{-1} d^{-1}, mineralization of TOC (POC+DOC) from WS (green), SO (blue), WWTP (magenta), nitrification (cyan) and aeration (black) (e); and the TOC concentration from WS (green), SO (blue) and WWTP (magenta) (f). Please note that SSC reaches a maximum value of 5.7 g.L^{-1}, because the turbidity sensors were saturated at the constant value of 9999 NTU in summer.*

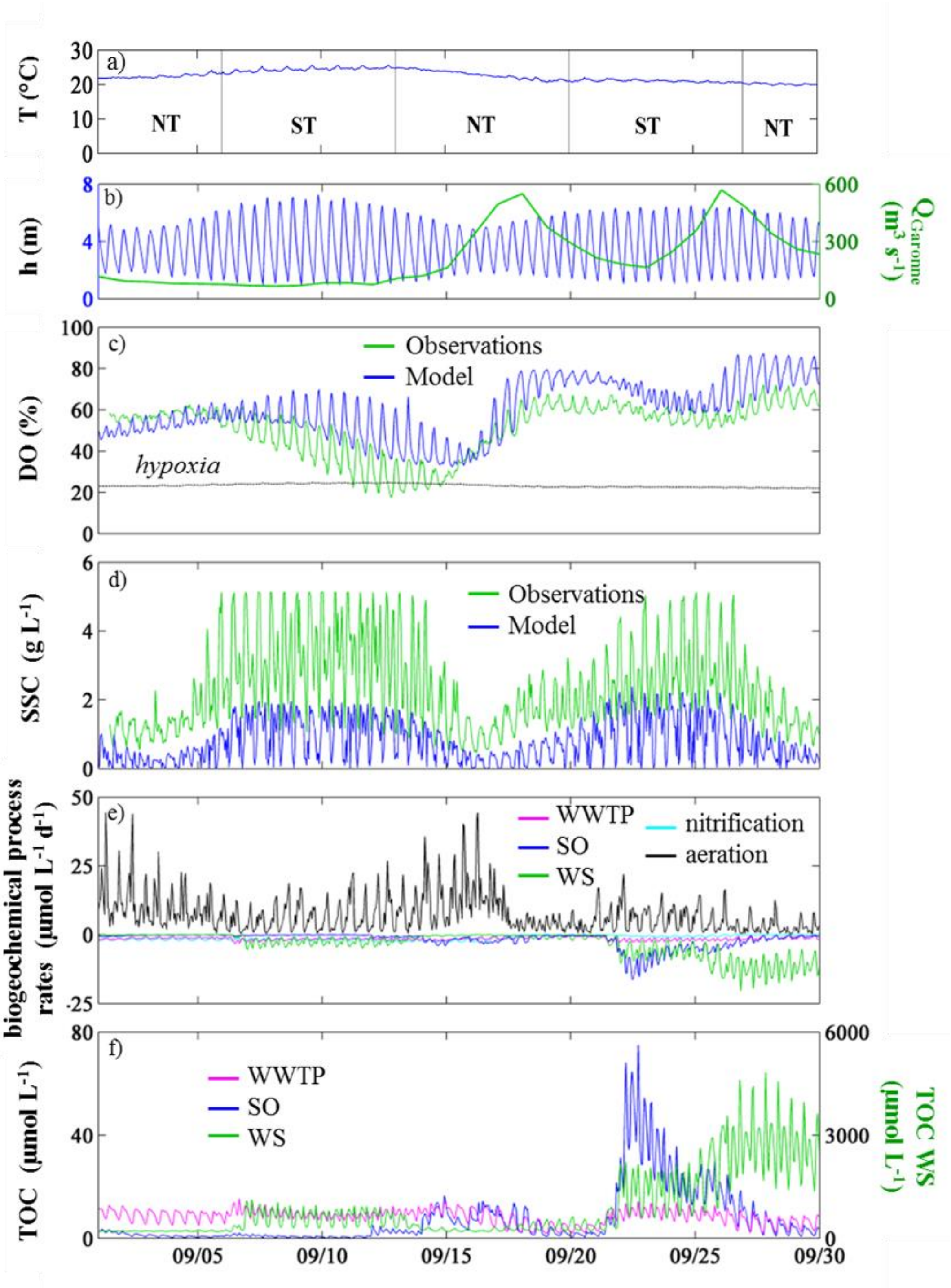


Figure III.5: *Temporal evolution during September 2006 at Bordeaux of: temperature (a), water level in m (blue) and river flow in m^3 s^{-1} (green) (b), simulated (blue) and observed (green) DO concentrations in %sat (c), and SSC in g L^{-1} (d). In the vicinity of Bordeaux, the daily average of the biogeochemical process rates affecting DO in µmol L^{-1} d^{-1}, mineralization of TOC (POC+DOC) from WS (green), SO (blue), WWTP (magenta), nitrification (cyan) and aeration (black) (e); and the TOC concentration from WS (green), SO (blue) and WWTP (magenta) (f). Please note that SSC reaches a maximum value of 5.7 g.L^{-1}, because the turbidity sensors were saturated at the constant value of 9999 NTU in summer.*

In Bordeaux City, the hydrodynamics are well simulated; the model skill for the water level and salinity is greater than 0.7 (Table III.2). For SSC, the model skill is lower (0.53) for the reasons explained above. Tidal records are not available further upstream. At Portets, the evaluation of the model skill is limited to salinity and SSC; the skill is less than 0.5, which is partly due to low variability in salinity in the upper estuary (salinity < 1). Moreover, the simulated late arrival of the TMZ in the upper Garonne River limits further upstream propagation and associated impact on DO dynamics to Portets.

For the biogeochemical model, the main parameters used in the reaction equations are taken from the available experimental data (see Table A3); there is no specific adjustment or correction of these parameters in the model, and no calibration simulations are run. Taking account of these factors, the observed and simulated seasonal and neap-spring time scale DO dynamics are satisfactorily simulated near Bordeaux City (Figures III.4b and III.5b). The model skill at this station has a satisfactory value of 0.77 (Figure III.3 and Table III.2). At the seasonal scale, the model reproduces the progressive DO reduction in summer, which is triggered by the increase in temperature (not shown) and SSC at the end of May (Figure III.4b). The model also accurately reproduces the DO amplitude, with model variation between 30 %sat and 100 %sat, compared with variation between 20 %sat and 120 %sat for the observations. The observed DO maximum at 120 %sat in May was due to a spring bloom of phytoplankton that was not simulated by the model. In general, simulated DO depletion occurs a few days later than observations, which can be explained by the movement of the TMZ upstream to Bordeaux ~20 days later than observed (Figure III.4c). Nevertheless, starting in July, there is good agreement between the model and the in situ data, with a DO bias of 10 %sat. Figure III.5 shows a simulation during the month of September 2006 with more detailed patterns at tidal frequencies. DO is modulated by the semi-diurnal tide (Figure III.5b). As described by Lanoux et al. (2013), during the spring-to-neap tide transition (from 09/10 to 09/15), when the turbidity is high, a decrease in DO occurs. Later, an increase in DO is recorded between 09/15 and 09/20, corresponding to a peak in the river flow. The model accurately reproduces the observation variability at all significant time scales, with no further calibration of biogeochemical parameters. The mechanisms involved in these dynamics will be explored in greater detail in section III.3.2.

In summary, the simulated hydrodynamics in the Garonne tidal river near Bordeaux are in good agreement with observations. We simulated the seasonal movement of the TMZ from the lower estuary into the Garonne River at low river flow in summer, as described in the previous work of Jalón-Rojas et al. (2015) and Sottolichio and Castaing, (1999). The modelled DO variation is also well simulated in the vicinity of Bordeaux. Thus, the model has sufficient skill to provide robust simulation and explanation of the seasonal and tidal dynamics of oxygen in Bordeaux. However, the results in the upper reaches of the Garonne River should be considered with more caution.

2.6.Sensitivity analysis

To perform the sensitivity analysis of the coupled model, we identify the key processes affecting oxygen in the vicinity of Bordeaux and then select which parameters of these processes to perturb. The largest fluxes contributing to DO variation in the water column are in order of increasing importance: the degradation of TOC (total organic carbon POC + DOC) from watershed (POC litter and DOCfluvial), of POC from WWTP, of POC from SO, nitrification and aeration. In addition, the sensitivity of the river DO boundary is assessed.

The parameters selected to assess the sensitivity of DO variation are: (1) kinetics of degradation of POC from SO, k_P^{SO}; (2) kinetics of degradation from WWTP, k_P^{WWTP} ; (3) kinetics of degradation from litter, k_P^{litter} ; (4) kinetics of degradation from fluvial, $\mathrm{k_D^{fluvial}}$; (5) kinetics of nitrification, k_{nit} (6) wind velocity, U_{10} ; and (7) river DO boundary conditions.

The sensitivity of DO to each parameter (Carmichael et al., 1995) is calculated as:

$$Sensitivity = \frac{V((1+a)p) - V((1-a)p)}{2aV(p)}$$

where ***V(p)*** is the average or minimum DO in Bordeaux in 2006, ***p*** is the parameter value and a is the perturbation. The perturbation values are given in Table III.3. If the sensitivity is close to 2, ***V*** is proportional to ***p***^{2}. A negative sensitivity value indicates a decrease in DO as the parameter increases.

According to Hadley et al. (2015), an absolute sensitivity value of 0.3 is the threshold for the model to be sensitive to the parameter. The DO mean is relatively insensitive to parameter variation for most of the selected processes (S1 < 0.1; Table III.3). However, the oxygen minimum is sensitive to k_P^{litter} and wind velocity (S2 > 0.60; Table III.3). DO in Bordeaux is also sensitive to the variation in the DO concentrations at the river boundaries (S1 and S2 >

0.6; Table III.3). For example, a reduction in the DO concentration from 90 %sat to 80 %sat in the river waters results in a reduction of 6 %sat in the Garonne River at Bordeaux. Therefore, it is important to use good quality data for the DO concentrations at the river boundaries to accurately determine the DO concentration at Bordeaux.

Table III.3: *Sensitivity analysis results of parameters variations affecting DO concentration in Bordeaux.*

Parameters	$\mathbf{k_P^{SO}}$	$\mathbf{k_P^{WWTP}}$	$\mathbf{k_P^{litter}}$	$\mathbf{k_D^{fluvial}}$	$\mathbf{k_{nit}}$	$\mathbf{U_{10}}$	**DO rivers**
Perturbation	± 0.1 d^{-1} or 25%	± 0.07 d^{-1} or 23%	± 0.0005 d^{-1} or 10%	± 0.0005 d^{-1} or 10%	± 0.01 d^{-1} or 10%	± 10 %	± 10 %sat or ± 11%
S1: Sensitivity of DO mean	-0.002	-0.003	-0.050	-0.019	-0.011	0.069	0.646*
S2: Sensitivity of DO min	-0.038	-0.008	-0.620*	-0.134	-0.096	0.369*	1.196*

*The absolute value of 0.3 is determined as the threshold for the model to be sensitive to the parameter Hadley et al. (2015)

3. Results

3.1.DO Dynamics over seasonal time scale

As stated in the previous section, in winter, the TMZ is located in the lower estuary, downstream from Pauillac, and the Gironde Estuary waters are well oxygenated (Figure III.6a and III.6c). In February, the bottom water SSC is less than 1 g L^{-1} in the tidal rivers, as the stock of sediment is dispersed downstream of the confluence of the Garonne and Dordogne Rivers. Consequently, the DO in the surface waters is greater than 70 %sat along the whole estuary (Figure III.6c). In summer, at low river flow, the TMZ moves upstream into the rivers. In the Garonne River, the SSC in the bottom waters is higher (> 1 g L^{-1}) from Bec d'Ambès up to Portets (Figure III.6b). In the same area, a drop in DO occurs in the Garonne River, with surface DO concentrations lower than 60 %sat for the month of July (Figure III.6d). The lowest DO area is located around and downstream of Bordeaux City, with concentrations less than 40 %sat (Figure III.6d).

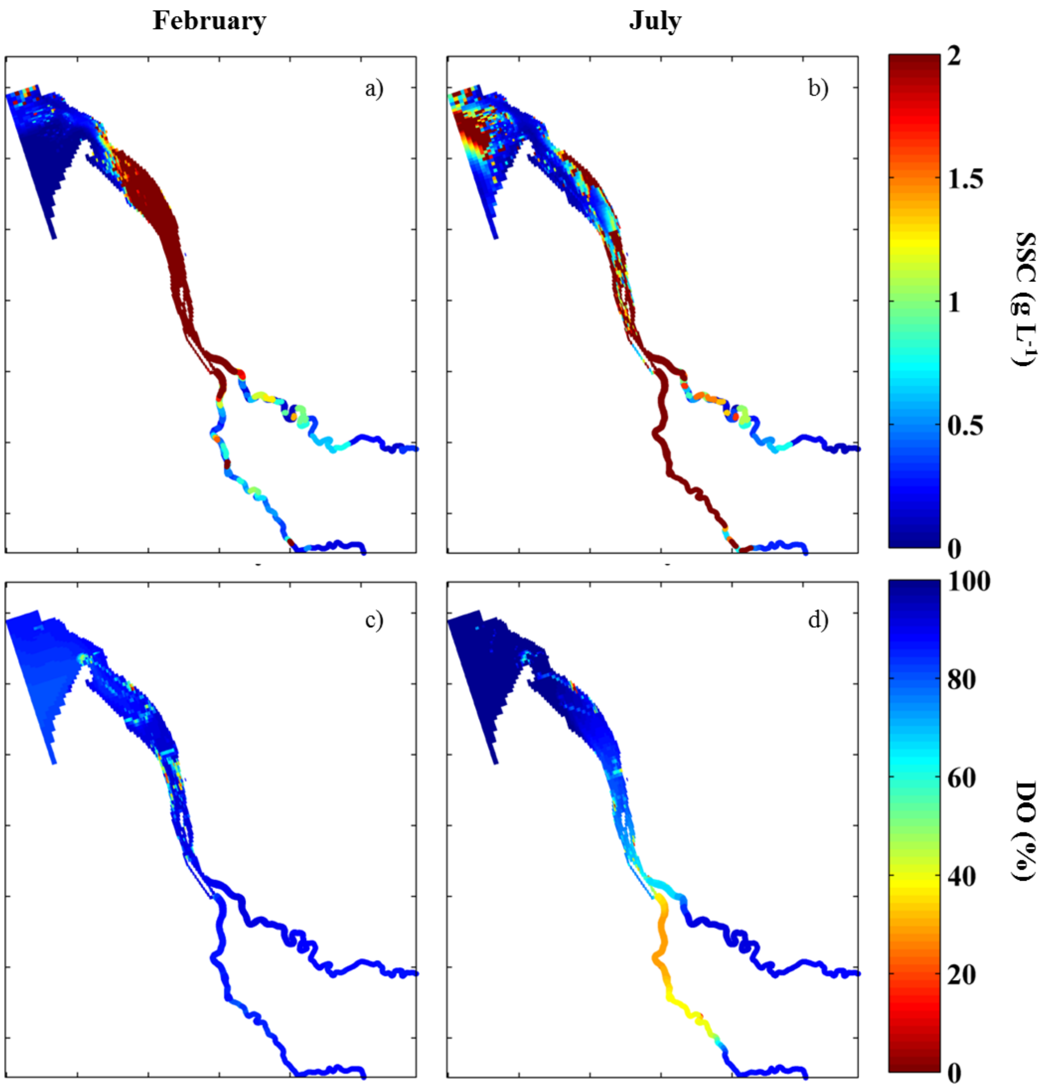


Figure III.6: *Simulated distribution of maximum near-bottom SSC in g.L^{-1} (top), and minimum surface DO in %sat (bottom) during February 2006 (left) and July 2006 (right).*

This spatial dynamics of surface SSC and DO are reflected in the temporal dynamics in the vicinity of Bordeaux (Figures III.4b and III.4c): the SSC in the surface water is approximately 0.1-0.2 g L^{-1} and DO is 90 %sat in winter. At the beginning of June (summer), the river flow decreases to less than 150 m^3 s^{-1} and reaches a minimum value of 54 m^3 s^{-1} and 52 m^3 s^{-1} in August for the Garonne and Dordogne Rivers, respectively (Figure III.4a). As a consequence, DO decreases concurrently with the arrival of the TMZ at Bordeaux, with a minimum of 29 %sat (or 71 µmol L^{-1}) (Figures III.4b and III.4c).

Figure III.7 shows the simulated evolution of the vertical structure of DO, SSC and salinity at the Bordeaux station in 2006. The distribution of oxygen throughout the water column is virtually homogenous, suggesting intense vertical mixing in the upper Gironde Estuary

(Figure III.7b, which is confirmed by the vertically homogenous distribution of salinity – Figure III.7d). A vertical SSC gradient occurs in the presence of the TMZ (Figure III.7c), which is more noticeable under spring tides conditions, when resuspension induces maximum SSC near the bottom. The salinity (Figure III.7d) is less than 2 psu, suggesting that Bordeaux is near the upper limit of salt intrusion.

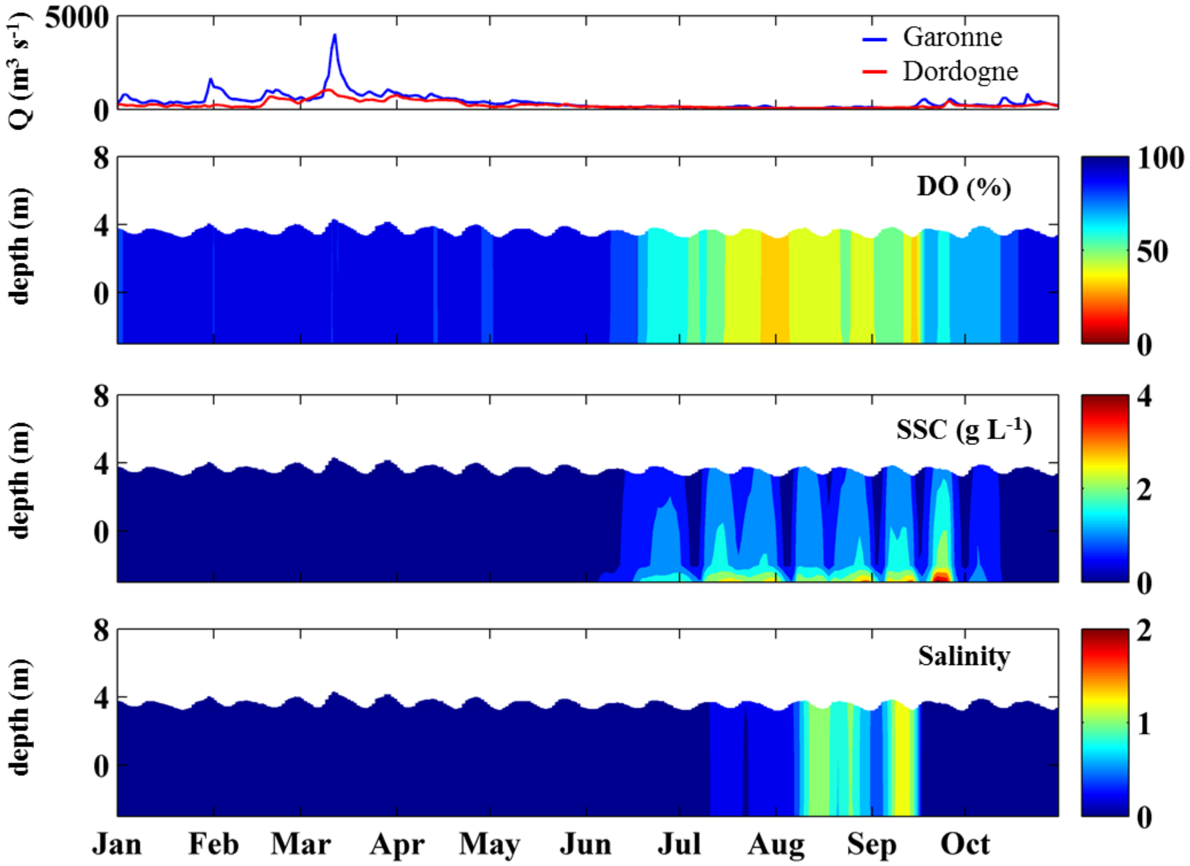


Figure III.7: *Vertical profile of temporal evolution between 1st January and the 31th October 2006 at Bordeaux of: rivers discharge in $m^3.s^{-1}$ (a), simulated daily average DO in %sat (b), SSC in $g.L^{-1}$ (c) and salinity in psu (d).*

The biogeochemical fluxes are calculated over the water volume of the Garonne River, consisting of an area of 6.6 km², covering the WWTP and the SO sites of Bordeaux (Figure III.1, orange area in lower panel). The model shows that degradation of TOC from the watershed (POC_{litter} and $DOC_{fluvial}$) is the main process affecting the DO concentration, with an annual average 50% oxygen consumption (Figure III.4d). Indeed, the inputs of $DOC_{fluvial}$ are continuous and contribute 14% of DO consumption. POC_{litter} enters the estuary during winter and spring and rapidly reaches the lower estuary, where it joins the TMZ. During summer, the TMZ moves to the upper estuary, and the trapped POC_{litter} contributes 36% of the oxygen consumption. The inputs of POC_{phyto} and $POC_{detritus}$ remain low, even during summer (6% and 2% of the total DO consumption, respectively). Gas exchange is the only process that balances oxygen, with an average input of oxygen of 5.31 µmol L^{-1} d^{-1}. Primary production is very low (due to the high turbidity), with a mean oxygen input of 0.04 µmol L^{-1} d^{-1}.

Figure III.4d shows that the mineralization rates increase greatly between June and September. This period corresponds to an increase in TOC in the area (Figure III.4e). There is a temporal correlation between the increase in mineralization rate and the presence of TMZ near Bordeaux (Figure III.4c). Moreover, the daily average TOC_{WS} shows an increase during the spring tide and a settling period at neap tide, corresponding to the POC dynamics (Figures III.4d and III.4e).

3.2.DO Dynamics at the Spring- Neap tidal scale

We now focus on the simulation of the period from the 1st to 30th of September 2006, typical of the low discharge season, covering two neap-spring tidal cycles and including two short river flow peaks (Figure III.5a). In the vicinity of Bordeaux, during the two transitions from spring-to-neap tides (i.e., from 09/07 to 09/15 and from 09/20 to 09/25), DO depletion is observed, with the minimum a few days after the spring tide peak (evidencing some inertia in the DO response to sediment resuspension), followed by recovery periods (Figure III.5b). Both are well reproduced by the model, as shown by the comparison with the observations. During the same spring-to-neap transitions, the increase in SSC due to resuspension by stronger tidal currents is simulated, followed by settling periods during neap tides (Figure III.5c). The integrated biogeochemical fluxes and TOC concentrations in the larger area of the Garonne River, including Bordeaux and the WWTP and the SO sites (Figure III.1, orange area in lower panel), are also calculated (Figures III.5d and III.5e). At neap tide (from 09/01 to 09/06 and from 09/14 to 09/21) the TOC_{WS} loads and the degradation flux are relatively low and aeration is much larger, which results in a slight increase of DO, while at spring tide, the opposite is observed (from 09/07 to 09/12 and from 09/20 to 09/26). After 09/14, there is a strong increase in the Garonne River flow of ~500 m^3 s^{-1} (Figure III.5a), which leads to an increase in DO of ~40 %sat in 3-4 days (Figure III.5b). The increase in river flow brings oxygenated water from upstream and decreases the gas exchange rate (Figure III.5d). After the river floods, a further increase in mineralization rates occurs, caused by the TOC loads advected from upstream by the river flow and re-suspended by spring tide currents (Figures III.5d and III.5e).

A view of the along-channel distribution of simulated variables (Figure III.8) shows the spatial structure of hypoxic waters and confirms the patterns observed at Bordeaux in summer. The simulated position of hypoxia persistently occurs near the city of Bordeaux, between 5 and 20 km from Bec d'Ambès (Figures III.8a and III.8b). When the currents are intensified, during the spring tide, an upstream shift of hypoxia of less than 5 km is observed, and a second DO minimum appears around Portets near the bottom, which is correlated with strong resuspension of deposited POC fixed on SSC, coinciding with an increase in POC remineralization (Figures III.8b and III.8d). Thus, during spring tides, hypoxia extends for more than 25 km between Bordeaux and Portets.

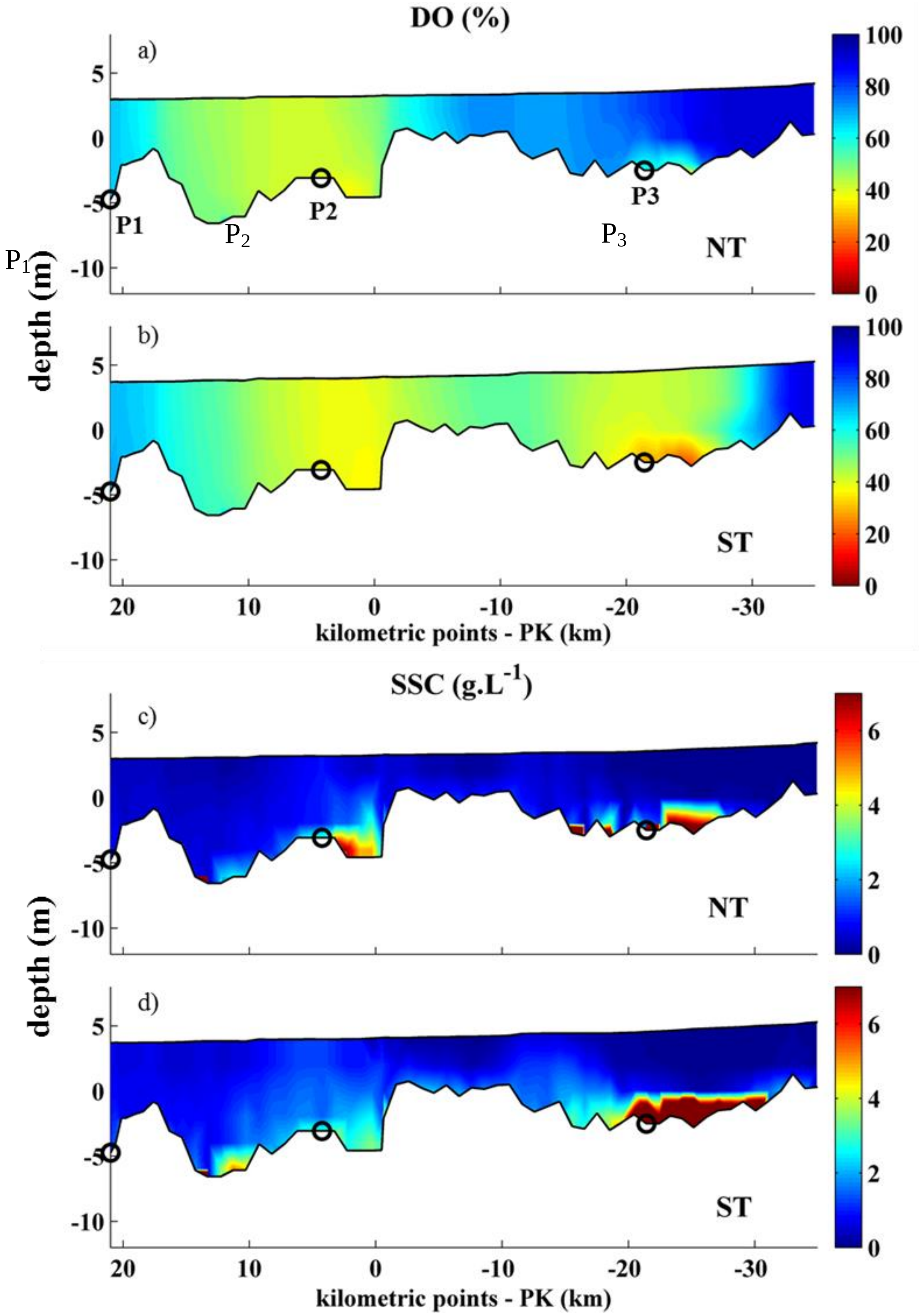


***Figure III.8:** Snapshot of the vertical transect of the DO concentration in %sat (top) and SSC in g L^{-1} (bottom) along the Garonne tidal river during high tide at neap tide (NT) and spring tide (ST). P1, P2 and P3 indicate the locations of Bec d'Ambès, Bordeaux and Portets, respectively.*

3.3. Impact of wastewater discharge

From the set of simulations run for 2006, the model shows that the mineralization of TOC from WWTP and SO consumes 19% and 12% of the DO (-1.05 and -0.70 µmol L^{-1} d^{-1}), respectively. In this region, the nitrification process accounts for 11% of total oxygen consumption on a yearly basis. However, during summer, nitrification accounts for 19% of the total oxygen consumption.

To assess the contribution of urban effluents on the oxygen concentration in Bordeaux, we compare the last simulation (presented previously) with a simulation over 10 months that integrates only inputs from the watershed (Figure III.9b). In 2006, the SO and WWTP discharged 8.3x106 m^3 and 43.1x106 m^3 of untreated stormwater and treated wastewater, respectively (16% and 84% of the total wastewater discharge), into the Garonne River. The difference in simulations with and without urban effluents (Figure III.9b) shows that in the Garonne River near Bordeaux, the DO minimum reaches 39 %sat in the absence of urban effluents and is reduced by 10 %sat by additional effluent.

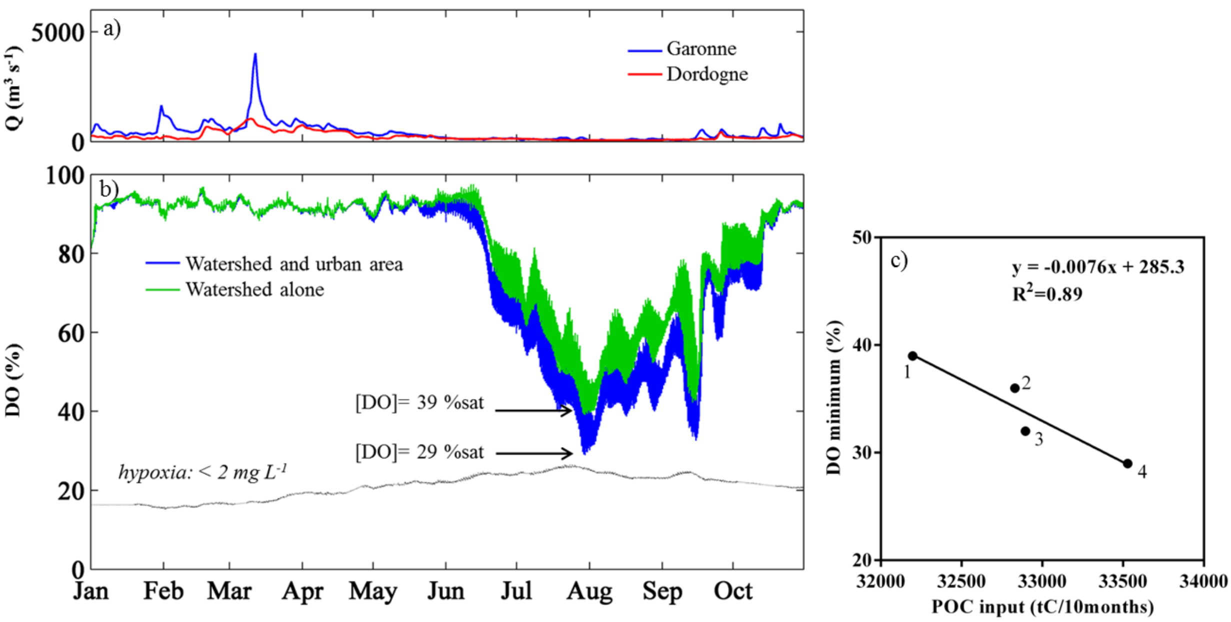


Figure III.9: *(a) River flow discharge. (b) Comparison of two simulations of DO variation in %sat at Bordeaux between the 1st of January and 31st of October 2006: Inputs from WS only (green), Inputs from WS, WWTP and SO (blue). (c) DO minimum (%sat) as a function of POC input (tC/10 months): simulation 1 reference: inputs from WS, simulation 2: inputs from WS and SO, simulation 3: inputs from WS and WWTP, simulation 4: inputs from WS, WWTP and SO.*

4. Discussion

The DO model implemented on the Gironde Estuary achieved the validation criteria specified in section III.2.5, including reproduction of the observed DO dynamics and amplitude near the city of Bordeaux at the seasonal and neap-spring tidal time scales. The model skill statistics validate that the model is sufficient for analysis of the DO dynamics in Bordeaux and assessment of the factors contributing to hypoxia in summer (WSS = 0.77 Table III.2, c.f. WSS = 0.75 in (Scully, 2013)).

Our biogeochemical model reproduced the observations established by Lanoux et al. (2013), where hypoxia occurs in the TMZ located in the fluvial section under low river flow, during high water temperature and a few days after the spring tide peak (Figures III.4 and III.5). Therefore, the model results confirm that a strong spatial and temporal correlation exists between the oxygen dynamics and the TMZ.

In summer, higher water temperature decreases oxygen solubility and accelerates biogeochemical reaction rates. Moreover, the TMZ favors mineralization processes that consume oxygen, and the turbidity limits the growth of phytoplankton due to elevated light attenuation in the water column (Goosen et al., 1999). As observed in macro-tidal European estuaries (Talke et al., 2009; Thouvenin et al., 1994), POC associated with suspended particles contributes to the majority of oxygen consumption, whereas gas exchange with the atmosphere is the major source of DO for the water column. However, in highly turbid estuaries, the presence of suspended sediment particles attenuates the turbulence and therefore slows gas exchange (Abril et al., 2009), accentuating the oxygen deficit in the TMZ. Figure III.4d shows that the gas exchange rate reaches lower values under higher SSC conditions. Abril et al. (2009) reported two effects of suspended sediment particles on turbulence: 1) the formation of SSC stratification near the bed and consequent turbulence damping and 2) enhanced dissipation of turbulent kinetic energy near the surface because of increasing viscosity.

The biogeochemical model allows us to locate the position of hypoxia in the Garonne River. The low DO zone is centered in front of the city of Bordeaux during the neap tide (Figure III.8). At spring tide, when the currents increase, the high flood currents move the hypoxic zone upstream from Bordeaux to Portets, and sediment resuspension favors POC degradation. The second DO minimum located around Portets in the bottom waters is probably caused by the strong resuspension of POC trapped in the fluid mud (Abril et al., 1999) (Figure III.8). The influence of SSC on the concentration of DO in the water is also observed at the neap-spring time scale (Figure III.5). Indeed, the POC trapped in the fluid mud earlier during the neap tide is re-suspended in the water column, favoring the degradation of organic matter and decreasing the DO concentration (Abril et al., 1999; Tengberg et al., 2003).

Estuarine urban input of nutrients and labile organic matter from wastewater leads to DO depletion. Greater hypoxia near large urban areas has been reported in many macro-tidal estuaries in Europe (Amann et al., 2012; Garnier et al., 2001; Soetaert et al., 2006; Tinsley, 1998) and China (Zhang et al., 2015). In the Gironde Estuary, the same observation was reported by Etcheber et al. (2011) and Lanoux et al. (2013). Our model showed DO depletion at Portets due to SSC resuspension during spring tide. However, the quantity of SSC is lower at Bordeaux than at Portets, and DO depletion is enhanced by urban effluents that contribute to oxygen consumption. Because of the modeling approach, we could quantitatively separate the contributions of material from rivers and urban watershed to the total DO consumption.

In the area of Bordeaux, our model showed that the mineralization of TOC from the watershed and WWTP was the main oxygen-consuming process (Figure III.4d). Nitrification accounted for only 11% of the DO consumption (19% in summer), compared with 50% of consumption in the Scheldt Estuary (before WWTP improvement; Cox et al., 2009). In the Scheldt Estuary, urban wastewater treatment did not include nitrification, so the ammonium concentrations were high. However, in the Gironde Estuary, the reduced nitrogen loads from the rivers are very low. The majority of nitrification is therefore due to WWTP effluents (Lanoux, 2013). Not accounting for the production of ammonium by anaerobic remineralization in the fluid mud could reduce the modelled NH4+ concentrations in the water column, limiting the contribution of nitrification to the total oxygen consumption. However, our modelled NH4+ concentrations are consistent with or slightly higher than those measured during the field campaigns in the late 1990s (between 2 and 25 µmol L^{-1}).

In Bordeaux, we found that with urban effluent input (WWTP and SO), the DO summer mean is 52 %sat (135 µmol L^{-1}; without urban input, it is 61 %sat – 157 µmol L^{-1}) and the DO minimum is 29 %sat (71 µmol L^{-1}; without urban input, it is 39 %sat or 96 µmol L^{-1}) (Figure III.9b). Further, our simulations summarized in Figure III.9c show that the discharge of treated wastewater from WWTPs alone reduce the minimum DO to 32 %sat (or 78 µmol L^{-1}), and the discharge of untreated wastewater from SOs alone reduce the minimum DO to

36 %sat (88 µmol L^{-1}). Consequently, if wastewater management could treat all the urban effluent discharged through the SO, DO would be higher, and low DO events would no longer occur in the Garonne tidal river, resulting in a reduction in biological oxygen stress. The WWTPs provide greater POC and NH_4^+ inputs than SO (Figure III.9c). However, the POC inputs from SO, which represent 16% of the urban effluent volume discharge, are more labile and impact the DO variation and reduce the DO minimum by 3 %sat. Thus, it is necessary to find better solutions to manage wastewater to enhance the oxygenation of the estuary.

The model highlights the sensitivity of water oxygenation to river flow for the oxygenation of water (Table III.3), particularly during the most important summer conditions. For accurate model predictions, it is important to monitor DO concentrations upstream in the river as reductions in river DO accentuate hypoxia events. Enhanced primary production in clear river waters might increase river DO and counteract hypoxia near Portets and Bordeaux; however, such blooms supply biodegradable POC to the upstream part of the TMZ. In summer, when the river flow is low, an isolated river flood can increase the DO concentration (Figure III.5). Thus, short rain events on the watershed increase river discharge and supply oxygenated freshwater to the upper estuarine area, mitigating hypoxia around Portets. In contrast, summer rain events on the urban area increase organic matter release by sewage overflow and thus enhance oxygen consumption (Even et al., 2007; Mouchel et al., 1994).

5. Conclusion

Although the biogeochemical model coupled with a hydro-sedimentary mode, includes some simplifications (e.g., no consideration of the various classes of phytoplankton and zooplankton and not reproducing the complete nitrogen cycle), it can be used with confidence to explore the seasonal dynamics of oxygen in Bordeaux and with slightly less confidence further upstream along the Garonne River. Because the presence and density of the TMZ determine the temporal and spatial dynamics of DO, including hypoxic events during summer and low river flow in the Garonne River, a 3D fully coupled model was necessary. The influence of SSC on water DO is shown at the seasonal and spring-neap time scales by the intensification of organic carbon degradation in the TMZ and during mud resuspension at spring tide. The hypoxic zone is located near the urbanized city of Bordeaux, which is impacted by urban effluents, and extends to Portets, 25 km upstream, during spring tides. The comparison of several model scenarios shows that DO depletion is accentuated mainly by wastewater discharge. Our findings suggest that an active management strategy can reduce POC inputs. For example, in our simulation, eliminating SO loads increased the DO minimum in the vicinity of Bordeaux by 3 %sat; omitting all urban effluents (SO and WWTP) increased the DO minimum in the vicinity of Bordeaux by 10 %sat and mitigated hypoxic events. To improve the oxygenation of the water, our DO model could be used as a tool to inform managers of useful future directions, and management scenarios could be tested to assess the quality, quantity and location of wastewater discharge.

Climate change projections for the Gironde Estuary, include a decrease in river flow, a sea level rise and an increase in water temperature (Etcheber et al., 2013). Variations in river flow and sea level could affect the location of the TMZ, and an increase in the water temperature could affect the biogeochemical flux rates. In this context, the proposed biogeochemical model could be used to further investigate the impact of climate change on the oxygenation of the waters of the Gironde Estuary.

Acknoledgments

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