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ABSTRACT: Eutrophication of the Seine estuary and the Bay of Seine is a crucial environmental issue for the management of ecosystems and economic activities related to fisheries and tourism in the region. A large quantity of nutrients, especially nitrogen, is brought to the coastal zone by the Seine River, the main input into that area, but also by smaller rivers along the Normandy coast. This large delivery of nitrogen leads to an imbalance between nitrogen (N), phosphorus (P) and silica (Si), which affects the growth of planktonic organisms and can exacerbate the occurrence of harmful algal blooms (HABs). These events can be damaging for shellfish fisheries, an important economic resource for the region. The study describes a new modelling chain coupling a riverine and a marine model (the Seneque/Riverstrahler and the ECO-MARS3D, respectively), which allows us to explore the effects on the coast of 2 scenarios of watershed management. The first one, focused on an upgrade of wastewater treatment plants, decreases the P fluxes by 5 to 35 kg P km^{-2} yr^{-1} on average over the 2000 to 2006 period, depending on the watershed, and would reduce about 3-fold the concentration of dinoflagellates in the adjacent coastal zone. The second one explores a hypothetical scenario of generalisation of organic farming in all agricultural areas of the basin. Although this is not realistic, it shows the best theoretical results we can achieve. With this scenario, the N fluxes decrease by almost 50%, and the dinoflagellate blooms and thus possibly the *Dinophysis* spp. blooms are drastically reduced by a factor of 20 to 40. Nevertheless, diatoms, which are the main primary producers in the bay and sustain the marine food web, are not significantly affected by this drastic scenario.

KEY WORDS: Eutrophication · Harmful algal blooms · Nutrient load · Modelling · Water quality · Seine

INTRODUCTION

Eutrophication of coastal zones and freshwaters is one of the main environmental issues today (Diaz & Rosenberg 2008, Voss et al. 2011). Many parts of the world, such as the English Channel and the North Sea (Lancelot et al. 1987, Lacroix et al. 2007), the Mediterranean Sea (Turley 1999, Danovaro 2003), the Black Sea (Moncheva et al. 2001, Заика 2003), the Gulf of Mexico (McIsaac et al. 2001, Rabalais et al. 2007) and along the coasts of China (Chai et al. 2006, Strokal et al. 2014) are impacted by this phenomenon. According to Nixon et al. (1996), eutrophication can be defined as an increase in the rate of supply of

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organic matter to an ecosystem. Eutrophication is typically caused by an excess of nutrients, mainly nitrogen (N) and/or phosphorus (P), delivered to river systems and ultimately to coastal areas (Billen & Garnier 2007, Bouwman et al. 2013). The imbalance between N and P compared to silica (Si) is also a crucial driver of eutrophication (Billen & Garnier 2007). A main manifestation of eutrophication is the development of micro or macro algal blooms (Heisler et al. 2008). Within river systems, blooms can cause problems in drinking water production (Grizzetti et al. 2011). Accumulation of organic matter can also lead to a decline in water clarity and submerged vegetation, and induce hypoxia or anoxia events in estuaries or coastal zones (Rabouille et al. 2008, Howarth et al. 2011). Eutrophication may also be the cause of harmful toxic algal blooms (Anderson et al. 2008).

Since the mid-1980s, and following the implementation of European directives, wastewater treatment plants (WWTPs) and their outputs in the Seine watershed have greatly improved. Secondary and tertiary treatments are now widely implemented and this has led to a decrease of P and N release by point sources. N fluxes originating from diffuse agricultural sources, however, have not decreased. As a result, since the end of the 1990s, N has been largely in excess of P in both the river and the coastal zone (Billen et al. 2005, Billen & Garnier 2007). The effects of this imbalance are not fully understood, but some studies relate these shifts in the nutrient ratios to an increase in the frequency of occurrence of harmful algal blooms in the Bay of Seine (Cugier et al. 2005a) and areas nearby (Lancelot et al. 2007, 2014, Passy et al. 2013).

In the Bay of Seine, *Dinophysis* spp. (Riou et al. 2010) and *Pseudo-nitzschia* spp. (Nezan et al. 2006) blooms occur regularly. These 2 species are damaging for shellfish production and fishery activities (Hoagland & Scatasta 2006), important economic resources in the region. The *Pseudo-nitzschia* spp. diatom produces an amnesic shellfish poisoning (ASP) toxin (Lundholm et al. 1994), which is absorbed by shellfish and especially scallops, one of the main fishery products in the English Channel. The dinoflagellate *Dinophysis* spp. produces a diarrhetic shellfish poisoning (DSP) toxin (Tripuraneni et al. 1997). This toxin is absorbed by shellfish, which then become inedible. Several shellfish kills caused by the DSP toxin have been reported (Jeffery et al. 2004). During the fall 2004, one of the biggest ASP crises occurred in the Bay of Seine. This crisis was followed by a severe DSP crisis in 2005. These events were responsible for the closing of many fisheries in the bay and so deeply impacted the economy of this sector.

The Seine watershed and the coastal zone of the Bay of Seine have been previously modelled, and nutrient load reduction scenarios have been explored (Billen et al. 2001, 2013, Cugier et al. 2005a, Garnier et al. 2010, Passy et al. 2013). These previous studies showed the ability of the models to reproduce the main trends in nutrient loads and algal bloom development in this region. They also pointed out the drastic decrease of P loads and the stability in N loads during the past 2 decades, and the impact of these changes on the decrease of *Phaeocystis* blooms in the North Sea. Some scenarios dealing with the management of point sources and diffuse sources have shown that further decreases in N loads and *Phaeocystis* blooms can be achieved. None of these studies, however, considered the whole Seine River–estuary–Bay of Seine continuum. Furthermore, the impacts of the Seine, Somme and Scheldt rivers flowing into the English Channel were determined without considering their estuaries (Lacroix et al. 2007, Lancelot et al. 2007, 2011, Thieu et al. 2009, Gypens et al. 2013, Passy et al. 2013). Only 1 previous study included, although roughly, the continuum Seine River–estuary–Bay of Seine (Cugier 1999). The present study addresses, for the first time, the transfer of nutrients within the whole system, analysing the functioning of the different compartments and including some noteworthy novelties compared to the previous studies.

- All the watersheds in Normandy that contribute to the Bay of Seine are modelled to take into account their contribution to eutrophication in the area. However, the estuary of the Seine River is the only 1 explicitly modelled in this study
- The modelling chain (see 'Materials and methods') is developed with the newest version of the models. Most particularly, the marine model is run with a more detailed description of the estuary that takes into account the processes occurring within the maximum turbidity zone
- The modelled results are validated with new field data covering a 7 yr period, and with satellite images showing chl *a* concentration and spatial extent.

In summary, the present study seeks to address the link between terrestrial activities and coastal impacts by coupling 2 different models, one dealing with watershed processes and freshwater flows, and the other describing the functioning of the estuary and the coastal system. This way, changes in the basin are dynamically transferred to the estuary and the bay, and we can directly assess the potential consequences offshore. We set out 2 future scenarios and we aim to hierarchize which of the nutrients is particularly noxious to land-to-sea water quality, or damaging for fisheries activities. The central part of this work is the assemblage of a modelling chain describing the biogeochemical processes occurring along the aquatic continuum, from the sources of the rivers to the coastal zone.

We believe our results can assist managers in their search for innovative strategies to prevent eutrophication concerns and, to some extent, harmful algal blooms in the Bay of Seine.

MATERIALS AND METHODS

Study area

Terrestrial domains

Together with the Seine basin, all the watersheds in Normandy flowing into the Bay of Seine are included in the modelling chain (Picoche et al. 2013) (Fig. 1). The following watersheds were explicitly modelled: the Seine, Eure, Risle, Touques, Dives, Orne, Seulles, Vire, Douve, Taute, Saire and Aure. The surface areas of the watersheds in Normandy range from 405 (Taute River) to 8130 km^2 (Orne River). Globally, the Normandy watersheds are dominated by grasslands, followed by arable lands, forest and urban areas (European Environment Agency 2007). Table 1 shows the main characteristics of each watershed modelled.

The Seine basin is the largest in our study area. At Poses, which is the upstream limit of the Seine estuary, the basin covers 64 866 km^2. Arable land dominates the basin, covering 53 % of the total area. Although the population density is high, only 7 % of the basin is covered by urban areas. About 10 % of the basin is dedicated to grasslands, which are located on the edges of it. Forests, also located at the edges of the basin, cover almost 25 % of it (Fig. 2). In 2011, the population of the basin was about 15 109 000 inhabitants (inh.) (Tavernier 2013), most of them living in the Parisian agglomeration and along the major river corridors. The mean population density is 233 inh. km^{-2}. However, the upper part of the basin and the interfluves are less populated, with

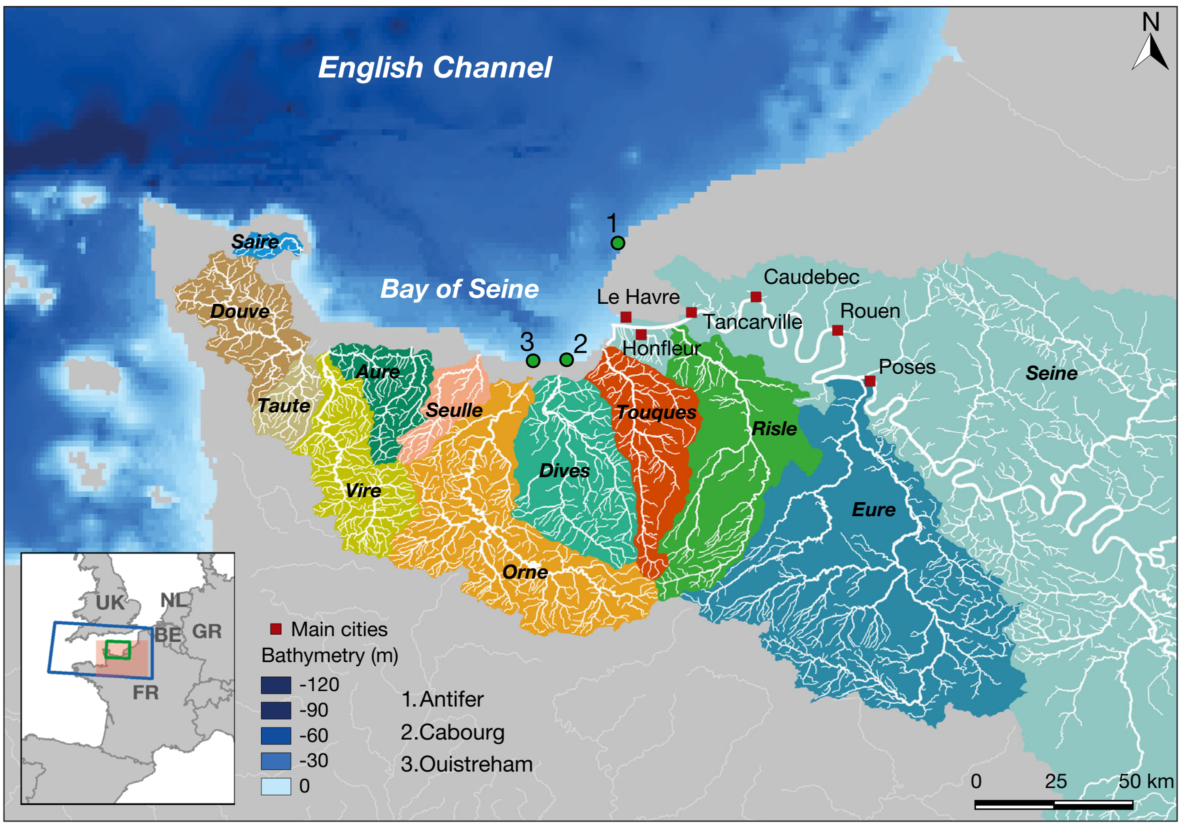

Fig. 1. Study area. Watersheds distinguished by colour with respective names in italic text. Locations of Antifer, Cabourg and Ouistreham (●). Inset: blue rectangle is the 'English Channel' configuration; green rectangle is the 'Bay of Seine' configuration; red rectangle is the spatial extent of the main map

Table 1. Main characteristics of the modelled watersheds. 'Seine' refers to the section of the river upstream of Poses. Q mean is the average of the discharges at the outlets of the rivers over the whole studied period (2000–2006). 'Strahler maximum' refers to Strahler order (https://en.wikipedia.org/wiki/Strahler_number) at the outlet of the river. inh.: inhabitants

River	Area (km^2)	Total length (km)	Drainage density (km km^{-2})	Strahler maximum	Q mean (m^3 s^{-1})	Population 2011 (inh.)	Population density 2011 (inh.km^{-2})
Seine	64 866	24 151	0.37	7	548.4	15 109 000	232.93
Eure	6015	2124	0.35	5	31.0	754 000	125.35
Risle	2312	679	0.29	4	15.7	187 000	80.88
Touques	1273	583	0.46	4	11.8	119 000	93.48
Dives	1776	906	0.51	4	8.5	133 000	74.89
Orne	8130	1696	0.21	5	35.8	453 000	55.72
Seulles	431	224	0.52	3	4.0	72 000	167.05
Vire	1239	887	0.72	5	18.9	119 000	96.05
Douve	1035	623	0.60	5	15.4	85 000	82.13
Taute	405	264	0.65	4	3.3	41 000	101.23
Aure	710	460	0.65	4	5.7	65 000	91.55
Saire	123	76	0.62	3	1.9	6 100	49.59

densities not higher than 30 inh. km^{-2} (Billen et al. 2009). See Meybeck et al. (2000) and Passy (2012) for more details about the physical geography of the Seine basin.

Owing to the temperate oceanic climate, high discharges are observed at the end of winter, while low discharges are observed at the end of summer. The Seine basin accounts for roughly 80% of the total water delivered to the Bay of Seine. In the Seine basin, 3 reservoirs, built from 1966 to 1991 to reduce winter floods and to increase summer low-water discharges, were taken into account in the modelling approach. The total capacity of the reservoirs reaches 750×10^6 m^3 (Garnier et al. 1999).

Estuarine and marine domains

The Seine estuary, the largest among the various rivers of the Normandy region, is around 165 km long and 4500 km^2 in area from Poses to Le Havre, on the coast. The shape of the estuary has changed over the years. Since the end of the 18th century, it has been deeply transformed for navigation (Patey 2004). A maximum turbidity zone (MTZ) is still located around Tancarville, 30 km upstream of Le Havre (Brenon & Le Hir 1999).

The marine domain is composed of the Bay of Seine and the English Channel. The Bay of Seine is the part of the channel impacted by the plume of the Seine River and is located along the Normandy coast. The area is about 4000 km^2 and the depth does not exceed 30 m (Aminot et al. 1997). The channel is a continental shelf sea, linking the Atlantic Ocean to the North Sea. It is about 550 km long and the mean width is about 90 km. Its mean depth is about 50 m and the maximum depth is about 170 m in the western part. Long-term water masses and nutrient fluxes go from southwest to northeast of the channel (Salomon & Breton 1993). In the eastern English Channel, circulation is mainly driven by the tide, except in the vicinity of the Seine plume.

Seneque/Riverstrahler model

The Seneque/Riverstrahler model (SR model) is a mechanistic biogeochemical model (Billen et al. 1994, 2009, Garnier et al. 2002, Ruelland et al. 2007). It describes the water quality within river networks at the watershed scale. The SR model is run at the kilometric resolution for the main streams, and at the Strahler order resolution (Strahler 1957) in the most upstream parts of the basin. In the former, the water quality of the rivers is obtained at each kilometre and in the latter the results are averaged for each Strahler-order river (https://en.wikipedia.org/wiki/Strahler_number). As inputs, the SR model takes hydro-climatic, diffuse and point source pollution data into account (Thieu et al. 2009, Passy et al. 2013). The model includes 30 state variables that can be analysed along the rivers' network; nutrient concentrations, phytoplankton biomass and suspended matter (SM) are the variables of interest for this study. Nutrients and phytoplankton fluxes delivered at the river outlets to the coastal zone are used to couple the river model to the estuarine-coastal zone ECO-MARS3D model (see below).

The SR model combines surface water flow and base water flow components with a model describ-

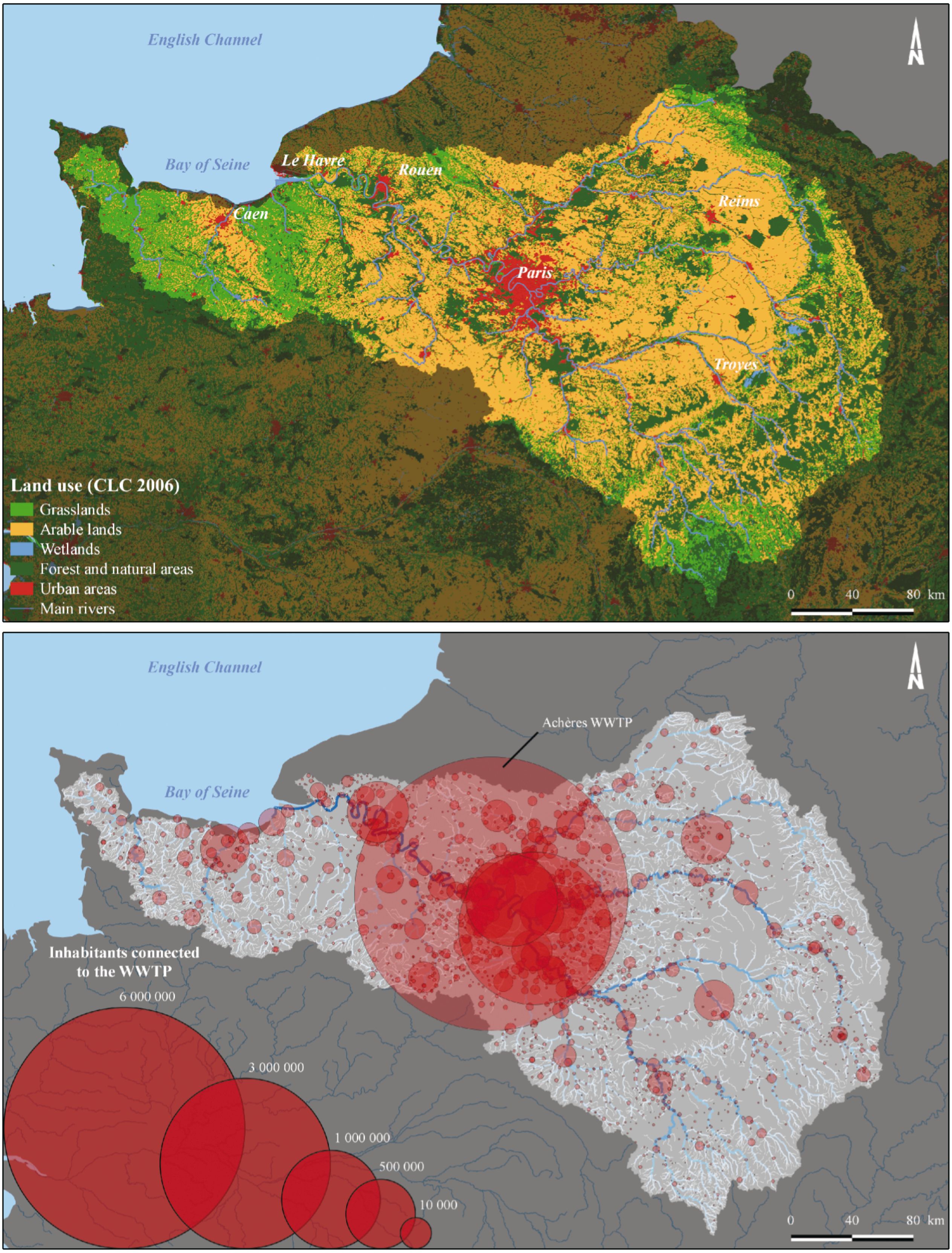

Fig. 2. Land use of the study area (top) and location of populations connected to the waste water treatment plants (WWTPs) (bottom)

ing the biological, microbiological and physical–chemical processes occurring within the water column, and the benthic phase. This part of biogeochemical computation is performed by the RIVE model. A detailed description of parameters and kinetics of the RIVE model is provided in Garnier et al. (2002). The SR model has not only been widely validated on temperate basins such as the Seine, Somme and Scheldt basins (Lancelot et al. 2007, Thieu et al. 2009, Passy et al. 2013), but also on tropical basins such as the Red River in Vietnam (Le et al. 2010, Luu et al. 2012) and the Nam Khan River in Laos (Causse 2011). Finally, the SR model provides nutrient concentrations of NO_3^-, NH_4^+, PO_4^{3-}, dissolved silica (DSi), chl *a* and SM, averaged for 10 d.

Input to the SR model

The SR model takes hydro-climatic and morphologic constraints as input data as well as data relative to point and diffuse sources of pollution. Temperature and morphology constraints are presented in detail in previous studies (Thieu 2009).

Hydrology

The SR model uses observed discharges that are divided into surface runoff and base flow. This discretisation is computed by using the Eckhardt recursive filter (Eckhardt 2008) applied on discharge data at different gauging stations. For the Normandy watersheds, missing discharges were interpolated (Picoche et al. 2013). The mean discharges are summarised in Table 1. Daily discharge data are provided from http://hydro.eaufrance.fr/.

Point sources

These correspond to the industrial effluents and the effluents of the WWTPs. About 1700 WWTPs are located within the Seine basin and 460 within the Normandy basins (Fig. 2). Each WWTP is precisely located and described by the population connected to the network and by the kind of wastewater treatments applied. These data are provided by the Seine Normandy Water Agency and allow us to compute the daily fluxes delivered to the rivers by each WWTP (Garnier et al. 2006, Servais et al. 1999).

Diffuse sources

Diffuse sources coming into the hydrological network are taken into account by associating surface runoff and base flow with mean annual concentrations of several nutrients: nitrate, ammonium, total inorganic phosphorus (TIP), SM and DSi. These different concentrations are spatially explicitly computed using the Corine Land Cover 2006 database (European Environment Agency 2007). Concerning forests, concentrations are considered as constant whatever the location within the basin (Fig. 2). As for meadows and arable lands, a discretisation procedure was carried out according to the diversity of agricultural practices (Mignolet et al. 2007, Schott et al. 2009). Spatial distribution of the agricultural areas was defined from Passy et al. (2013).

These agricultural areas were documented according to the nutrient concentrations in sub-root zones and aquifers (Thieu et al. 2009). The mean NO_3^- concentrations for arable lands are 18.1 and 9.4 mg N l^{-1} for the sub-root and aquifer compartment, respectively; for P, these are respectively 0.31 and 0.06 mg P l^{-1} and for Si these are 11.3 mg SiO_2 l^{-1} for both. See Thieu et al. (2009) and Passy et al. (2013) for more details about the other nutrients and the spatial distribution of the concentrations. The concentrations of SM were established in earlier studies (Némery et al. 2005, Némery & Garnier 2007, Thieu et al. 2009). TIP is calculated from the concentration of SM and P in soils according to the equation describing the equilibrium between adsorption and desorption (Billen et al. 2007). These P concentrations vary as a function of land use, from 0.1 g P kg^{-1} for forests to 1.7 g P kg^{-1} for the most intensively pastured areas. Intermediate values of 0.7 to 1.4 g P kg^{-1} are used for the central parts of the Paris basin (Némery et al. 2005).

Concerning DSi, initial concentrations were computed based on lithology, following the work of Meybeck (1987) and Garnier et al. (2005). This variable is now defined for each type of rock and soil associated with the agricultural areas. The biogenic silica, from the erosion of phytoliths, was calculated from SM concentrations on the basis of experimental work (Sferratore et al. 2006), leading to 4.9 mg Si g^{-1} (i.e. 81.7 µmol Si) for soils and sediments.

ECO-MARS3D model

In this study, the ECO-MARS3D model is run using offline nesting. The coarse 2 km resolution grid covers the whole English Channel and its boundary conditions are defined by climatology. This coarse resolution grid provides boundary conditions (biotic and abiotic) for a high 500 m resolution grid representing the Bay of Seine (Fig. 1). Each grid is Cartesian, using Arakawa-C and sigma coordinates (Lazure & Dumas 2008). The child grid ('Bay of Seine' configuration) covers only the Bay of Seine and has a 500 m spatial resolution. This refinement allows a better description of the Seine estuary and the MTZ. Furthermore, in this higher resolution grid, the Seine estuary explicitly takes into account realistic flow sections and estuary length from Poses.

The kernel of the ECO-MARS3D model is a 3-dimensional (3D) hydrodynamic model called MARS3D (3D hydrodynamical model for application at regional scale) (Lazure & Dumas 2008) on which 2 modules are coupled. The first describes the sedimentary dynamics of sand mud mixtures (Le Hir et al. 2011) and the other the biogeochemical processes (Cugier et al. 2005b). First, in order to produce accurate simulations, especially concerning the timing of nutrient arrival at the coastal zone, the Seine estuary characteristics were intensively calibrated (Garnier et al. 2012). Tidal propagation and currents were validated inside the Seine estuary, as well as at the scale of the whole bay.

The ability of the model to accurately represent the MTZ (in concentration) and its displacements was also carefully calibrated in order to reproduce growth limitation due to SM in this area of the estuary. For the sediment dynamics module, 2 classes of particle were introduced in the model according to Cugier (1999). The first one has a very low settling velocity so that particles are easily advected from Poses throughout the estuary. Thus, these carried SM stock from the plume in the vicinity of the estuary mouth. The second class is characterized by a higher settling velocity in order to represent the functioning of the MTZ. The initial stock of 'heavy' sediment is introduced between Honfleur and Tancarville uniformly to represent a standing stock of 400 000 t (Avoine et al. 1981).

The following measured physical characteristics (Brenon & Le Hir 1999, Dupont 2001) were forced during the modelling process: at low tide (LT), the MTZ is centred 2 or 3 km downstream from Honfleur with a downstream limit at km 352 and an upstream limit at km 340, close to Tancarville; at high tide (HT), the MTZ is centred around km 340, with a downstream limit at km 354 and an upstream limit at km 335.

Of the major and fundamental elements to living organisms, the nitrogen, silica and phosphorus cycles are taken into account. In the model, they are described under their mineral and detrital forms in the description line in the SR model. Consequently, the biogeochemical model is an NPZD model (nutrient > phytoplankton > zooplankton > detrital). NO_3^- and NH_4^+ are distinguished, and mineral phosphorus is divided into dissolved phosphate and P adsorbed on SM within the water column. The phytoplankton compartment is divided into 3 groups: diatoms, which dominate in spring; dinoflagellates, which are particularly abundant in summer and autumn; and nanoflagellates, which are rarer (Belin & Raffin 1998). These 3 kinds of algae are considered in the model via their nitrogen content. The regulation of phytoplankton by grazing is performed by 2 classes of zooplankton discretised according to size. The microzooplankton only grazes nanoflagellates and detrital organic matter, the latter being degraded by first-order kinetics. The mesozooplankton grazes diatoms, dinoflagellates and microzooplankton. The model also includes salinity, temperature and mineral SM.

The meteorological constraints (wind, atmospheric pressure, air temperature, cloud cover and relative humidity, available every 6 h at a spatial resolution of 0.5°) are provided by the ARPEGE model developed by Météo-France (Déqué et al. 1994).

Coupling between the SR and marine model

The SR model provides freshwater flow and nutrient concentrations as input for the marine model for all the rivers. The Seine River discharges at the upstream limit of the Seine River estuary, at the Poses weir in the ECO-MARS3D model, whereas all the other rivers discharge directly at the coast.

Variables from the ecosystem model were validated by comparing simulation results with 3 types of observational data. Most of the measured data come from the REPHY, (Réseau de surveillance du phytoplancton et des phycotoxines) maintained by the Ifremer (Institut français de recherche pour l'exploitation de la mer) and the RHLN (Réseau Hydrologique Littoral Normand) networks. Besides routine measurements of nutrients and chl *a* concentrations, the REPHY provides detailed data on 2 phytoplankton species that cause recurrent harmful blooms in the area. These are the diatom *Pseudo-nitzschia* spp. and the dinoflagellate *Dinophysis* spp., as well as their respective toxins, the ASP and the DSP.

This modelling coupling was implemented for the 2000 to 2006 period. This period is interesting because it includes contrasted hydrological years, 2001 being a wet year and 2005 a dry year, and because 2 different crises occurred in the Bay of Seine during this period: an ASP crisis in 2004 and a DSP crisis in 2005.

RESULTS

Model results and validation

Hydrology and water quality at the river outlets and delivered fluxes

In terms of hydrology, it is possible to validate the total discharge reconstruction at the outlets of the rivers by comparing it with the actual flow measurements. Fig. 3 shows the discharge reconstruction at the outlets of the Orne, Vire and Seine rivers over the 2000 to 2007 period. The inter-annual hydrological dynamic is similar for the 3 watersheds. The wettest year is 2001 and the driest 2005.

The nutrient concentrations at the outlets of the small Normandy rivers over the 2000 to 2010 period, and 2000 to 2006 for the Seine, were validated against the measurements. The validation data are provided by the Seine Normandy Water Agency (http://www.eau-seine-normandie.fr) and, in some cases, by our own measurements. The time step of these validation data is variable in time and depends on the watersheds. As the parameters of the model

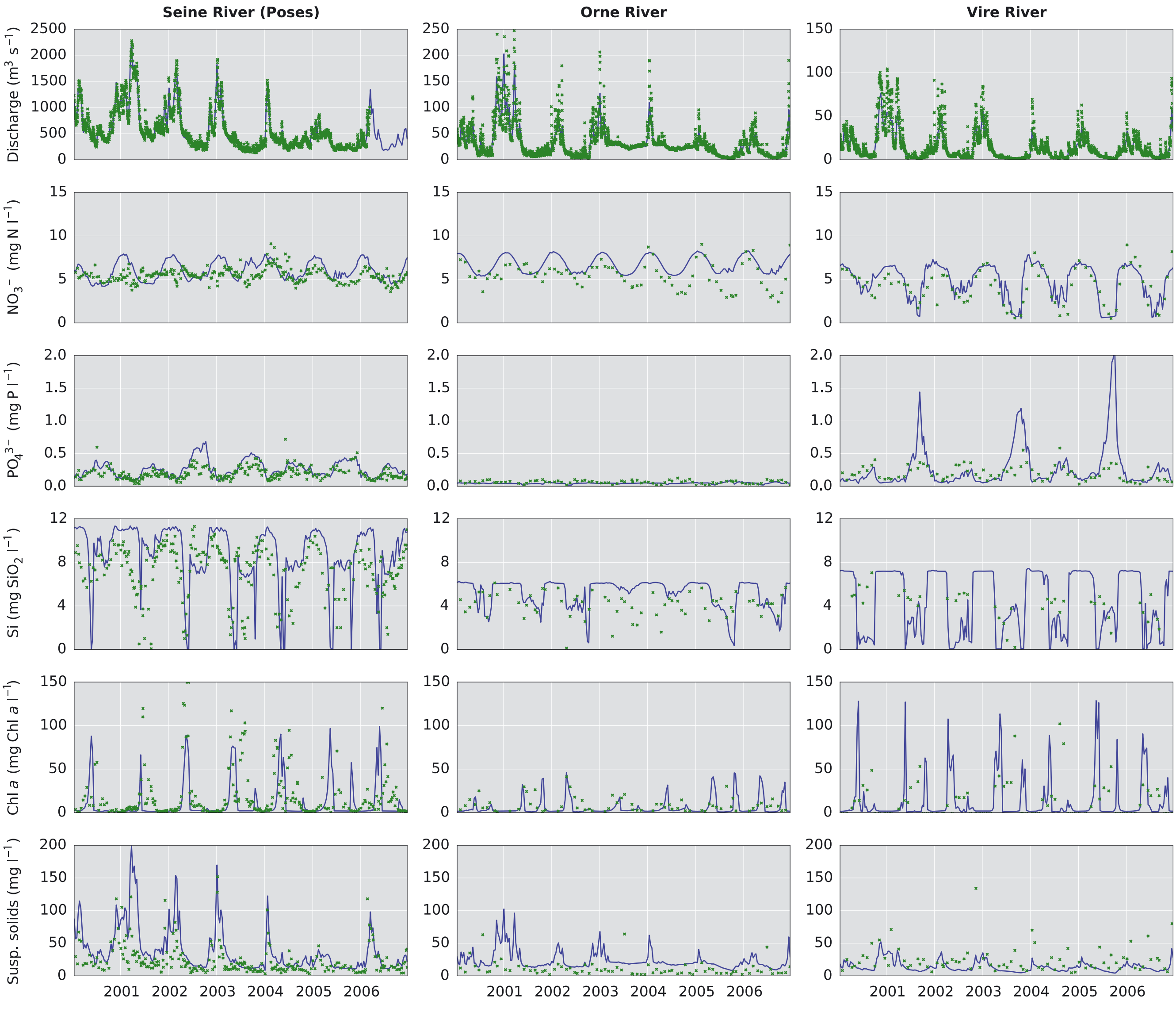

Fig. 3. Validation with the Seneque–Riverstrahler model of several water quality variables at Poses for the Seine (left), the Orne (middle) and the Vire (right) rivers for the 2000 to 2006 period. Continuous lines are simulations; dots are observations

were experimentally determined independently from the observations, the comparison between simulations and data represents a true validation.

Concerning the watersheds in Normandy, only the validations of the Orne and Vire rivers, the 2 largest rivers in the area, are presented (Fig. 3). The mean NO_3^- concentrations over the 2000 to 2010 period are 7.4 (528.6), 5.0 (357.1) and 6.0 mg N l^{-1} (428.6 µmol l^{-1}) for the Orne, the Vire and the Seine, respectively. Winter NO_3^- values are always above the summer values. Retention of nitrate within the drainage network and its riparian interfaces explains this phenomenon (Grizzetti et al. 2015). The winter peaks, for both rivers, reach about 8.5 mg N l^{-1} (607.1 µmol l^{-1}). The main difference is observed for summer low values. In the case of the Orne and the Seine, minimum summer levels are about 3 to 4 mg N l^{-1} (214.3 to 285.7 µmol l^{-1}), but for the Vire the minimum can be close to 0 mg N l^{-1}.

Values of PO_4^{3-} are very low over the whole study period. For the Vire and Seine rivers, annual averages are below 0.5 mg P l^{-1} (16.1 µmol l^{-1}), although some peaks reaching 0.6 mg P l^{-1} (19.4 µmol l^{-1}) can occasionally be observed at the outlets of both rivers.

With regard to DSi, in the Orne and the Vire rivers, winter levels are around 5.6 mg SiO_2 l^{-1} (93.3 µmol Si l^{-1}). Every year, a decrease of DSi is observed and modelled at the end of spring and beginning of summer. For some years, measurements show values around 0 or 1 mg SiO_2 l^{-1} (0 to 16.7 µmol Si l^{-1}), but in other years the minimum level reached is 4 mg SiO_2 l^{-1} (66.7 µmol Si l^{-1}). At the outlet of the Seine River, the seasonal trends of DSi are characterised by high values in winter, reaching 11 mg SiO_2 l^{-1} (183.3 µmol Si l^{-1}), and very low values at the beginning of summer, often less than 1.5 mg SiO_2 l^{-1} (25 µmol Si l^{-1}). As in the case of the Normandy rivers, the decrease of DSi is explained by the growth of phytoplankton.

For all 3 rivers, winter levels of chl *a* are very low, and maximum concentrations are found at the beginning of summer; phytoplankton concentrations are higher in the Vire River, some years reaching values of over 100 µg chl *a* l^{-1}, while values rarely exceed 50 µg chl *a* l^{-1} at the outlet of the Orne River. In the Seine at Poses, peaks can reach 130 µg chl *a* l^{-1}, and these maximum concentrations are regularly found at the beginning of summer.

Concerning SM, the annual averages are about 15 to 20 mg l^{-1} at the outlet of the 2 Normandy Vire and Orne rivers. Peaks of SM correspond to wet years, as SM is strongly correlated to hydrology. At the outlet of the Seine River, the mean annual SM values are about 39.0 mg l^{-1}. Maximum SM values are around 150 mg l^{-1} during wet years and around 50 mg l^{-1} during dry years.

Table 2. Validation indices for N, P and Si fluxes at the outlet of the Seine River at Poses

Index	N fluxes	P fluxes	Si fluxes
Correlation	0.96	0.94	0.72
Bias	7.80	5.86	0.24
Index of agreement	0.94	0.92	0.79
Nash-Sutcliffe efficiency criteria	0.63	0.56	0.31

Finally, Table 2 shows validation indices for the 3 main nutrients, which are N, P and Si. These indices are correlation, bias, index of agreement and Nash-Sutcliffe efficiency criteria (Krause et al. 2005, Willmott et al. 2012). They are calculated for the fluxes delivered by the Seine River at Poses. These indices show a generally good fit between calculation and observations for N, P and Si fluxes, although less for silica, as shown by Nash and bias indices (Table 2).

Hydrodynamic conditions within the coastal model

Tidal validation. At the scale of the Seine estuary, the modelled tidal dynamics were compared to tidal levels measured by tide gauges. After many sensitivity tests, friction was adjusted using variable roughness length within the estuary. This step was useful to reproduce the hydrodynamic characteristics of the estuary accurately: deformation of the tidal wave; asymmetry between the rising tide and the ebb tide; and transit time (Guézennec 1999). Tidal elevation in the Seine estuary agrees well with observations (Fig. 4). The tidal amplitude ranges from around 2.5 to around 4.5 m, depending on the period of the tidal cycle. Some discrepancies between simulations and observations can be found at the end of the ebb tide, especially when the tidal amplitude increases.

Tidal circulation. The French Naval Hydrographic and Oceanographic Service (SHOM) provides depth-averaged current information on its nautical charts. These hourly data come from *in situ* sampling and provide intensity and direction of the current during a tidal cycle for a given tidal coefficient of 95. To validate tidal circulation within the Bay of Seine, a simulation without wind was carried out and compared to SHOM currents. Ellipses were drawn using hourly values of intensity and direction of currents during

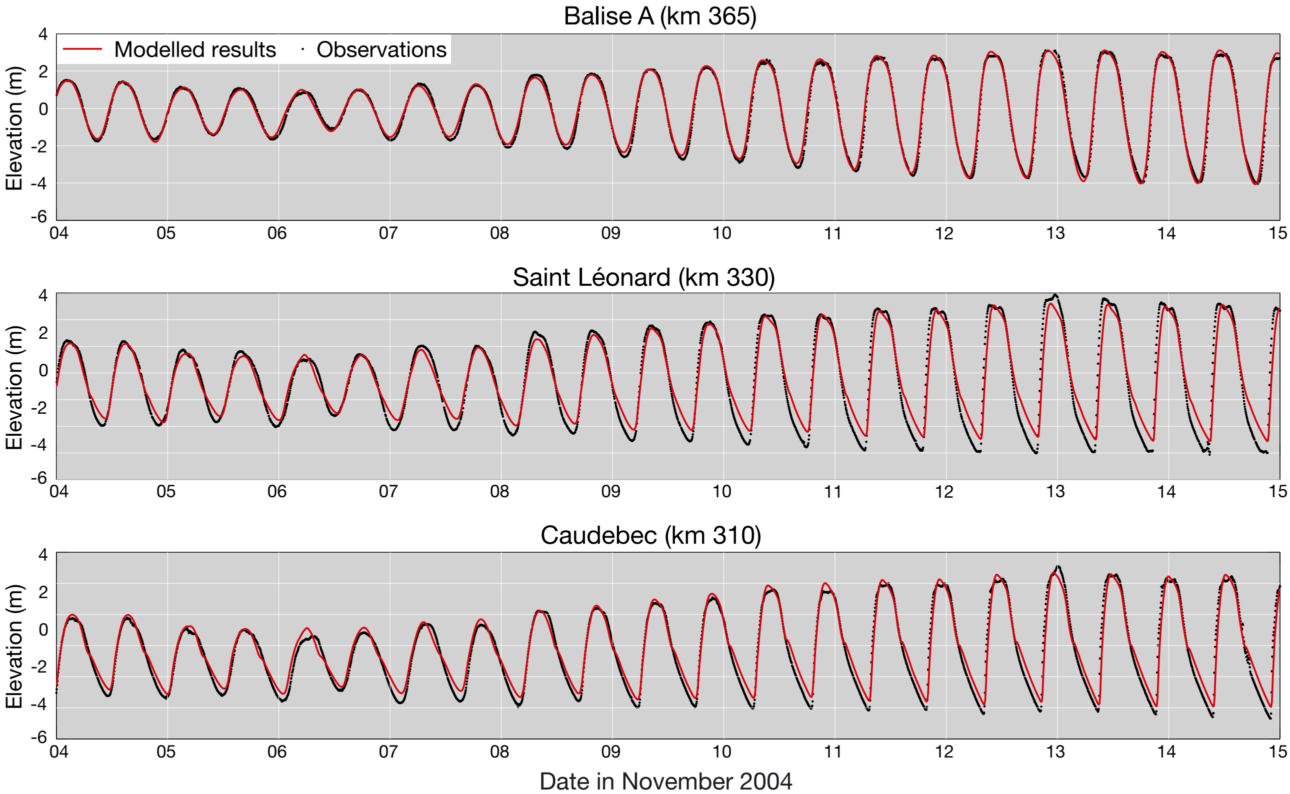

Fig. 4. Comparison of modelled (red) and measured (black) elevation from 4 to 15 November 2004 at 3 stations of the estuary. The mean associated discharge at Poses during this period is 500 m^3 s^{-1}. 'Balise A' is roughly located at Le Havre and 'Saint Léonard' at Tancarville (see Fig. 1)

one tidal cycle of approximately 12 h (from high tide to high tide). The contours of the ellipses are defined by the different positions of the current vectors during the cycle. Modelled tidal ellipses are consistent with measured data in terms of simulated intensities as well as directions (Fig. 5). The major axes and the general shapes of the ellipses are consistent between model and *in situ* measurements, which means that the orientation and intensity of the circulation are accurately reproduced by MARS3D. The strongest currents are found near the Barfleur Cape in the northeast of the Cotentin Peninsula, in the northwest of the bay. In the middle of the bay, currents are less strong and ellipses' axes are directed along the same axis as the paleovalley of the Seine River. Ellipses are mainly elongated except near the Seine outlet.

The MTZ. In the biogeochemical model, the role of SM is crucial. Indeed, SM impacts turbidity and light penetration into the water column, and ultimately primary production and nutrient dynamics (Garnier et al. 2013). The concentration and location of the MTZ within the estuary vary according to the tidal cycle

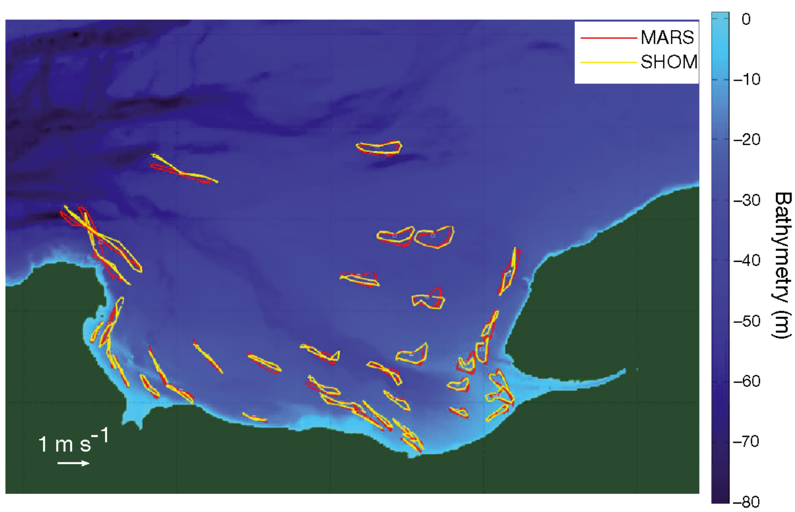

Fig. 5. Modelled current ellipses using the ECO-MARS3D (MARS; red) and measured current ellipses by the Naval Hydrographic and Oceanographic Service (SHOM; yellow) for an average spring tide coefficient (95)

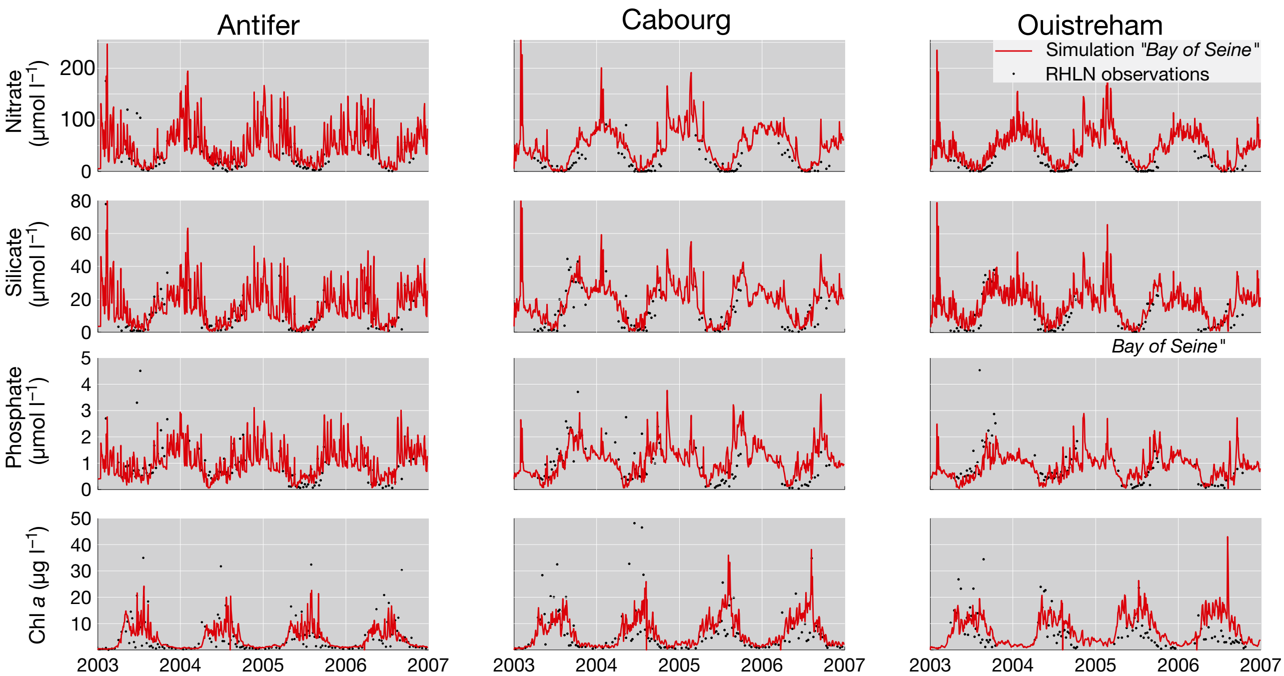

Fig. 6. Simulations of 'Bay of the Seine' spatial configuration, and the measured data at the Antifer, Cabourg and Ouistreham stations. Top to bottom: nitrogen, silica, phosphorus, chl *a*. Observation data come from the Hydrologic Network for Normandy Coast (RHLN) established in 2000 by Ifremer

(Garnier et al. 2012). At LT, the MTZ is in its most downstream location and the MTZ plume leaves the estuary. At this moment, the concentrations of suspended solids in the MTZ are above 2 g l^{-1}. At HT, the MTZ reaches its most upstream location in the estuary. At this time, MTZ SM settles because of the very weak flow.

Biogeochemical conditions within the coastal model

Fig. 6 shows the inter-annual dynamics of N, Si, P and chl *a* at 3 different offshore stations for the 2003 to 2006 period. At the 3 stations, maximum nutrient concentrations are found in winter, and they reach around 2.1 to 2.8 mg N l^{-1} (150 to 200 µmol l^{-1}), 3 to 3.6 mg SiO_2 l^{-1} (50 to 60 µmol l^{-1}) and 0.06 to 0.09 mg P l^{-1} (2 to 3 µmol l^{-1}) for N, Si and P, respectively. During summer, concentrations are lower, around 0.35 to 0.7 mg N l^{-1} (25 to 50 µmol l^{-1}), 0.3 to 0.6 mg SiO_2 l^{-1} (5 to 10 µmol l^{-1}) and 0.02 to 0.03 mg P l^{-1} (0.5 to 1 µmol l^{-1}). For the 3 stations, modelled nutrient concentrations are close to the measured concentrations and the yearly cycle is properly simulated. Phytoplankton grows on the stock of nutrients accumulated during winter. Maximum phytoplankton concentration is reached in summer, with chl *a* values of 30 to 40 µg chl *a* l^{-1}, whereas minimum concentration of 2 to 5 µg chl *a* l^{-1} is found during winter.

The chl *a* concentration obtained from satellite data, known as the OC5 product, was available at the French Research Institute for Exploitation of the Sea (Ifremer). These data have a resolution of 1.1 km and are a composite product merging 3 different data sources: MERIS, MODIS/Aqua and SeaWiFS (Saulquin et al. 2011). This validation method is commonly used in many parts of the world (McCain et al. 2006, Dogliotti et al. 2009). According to Fig. 7, the spatial extension of the bloom and biomass levels are accurately simulated. The spatial correlation and index of accordance between simulations and satellite images are satisfactory along the coasts and at the outlet of the Seine estuary. We can see in Fig. 7, directly at the outlet of the estuary, a zone without chl *a* indicating the effect of the MTZ, which limits the primary production.

Nutrient reduction scenarios

To reduce eutrophication in the Bay of Seine, we used the modelling chain to simulate 2 different scenarios of watershed management. The first one (SWT) simulates the application of the European directive relative to upgrading WWTPs. This scenario reduces the point sources, especially concerning P. In this scenario, every WWTP treating more than 2000 equivalent inhabitants is upgraded with a full tertiary

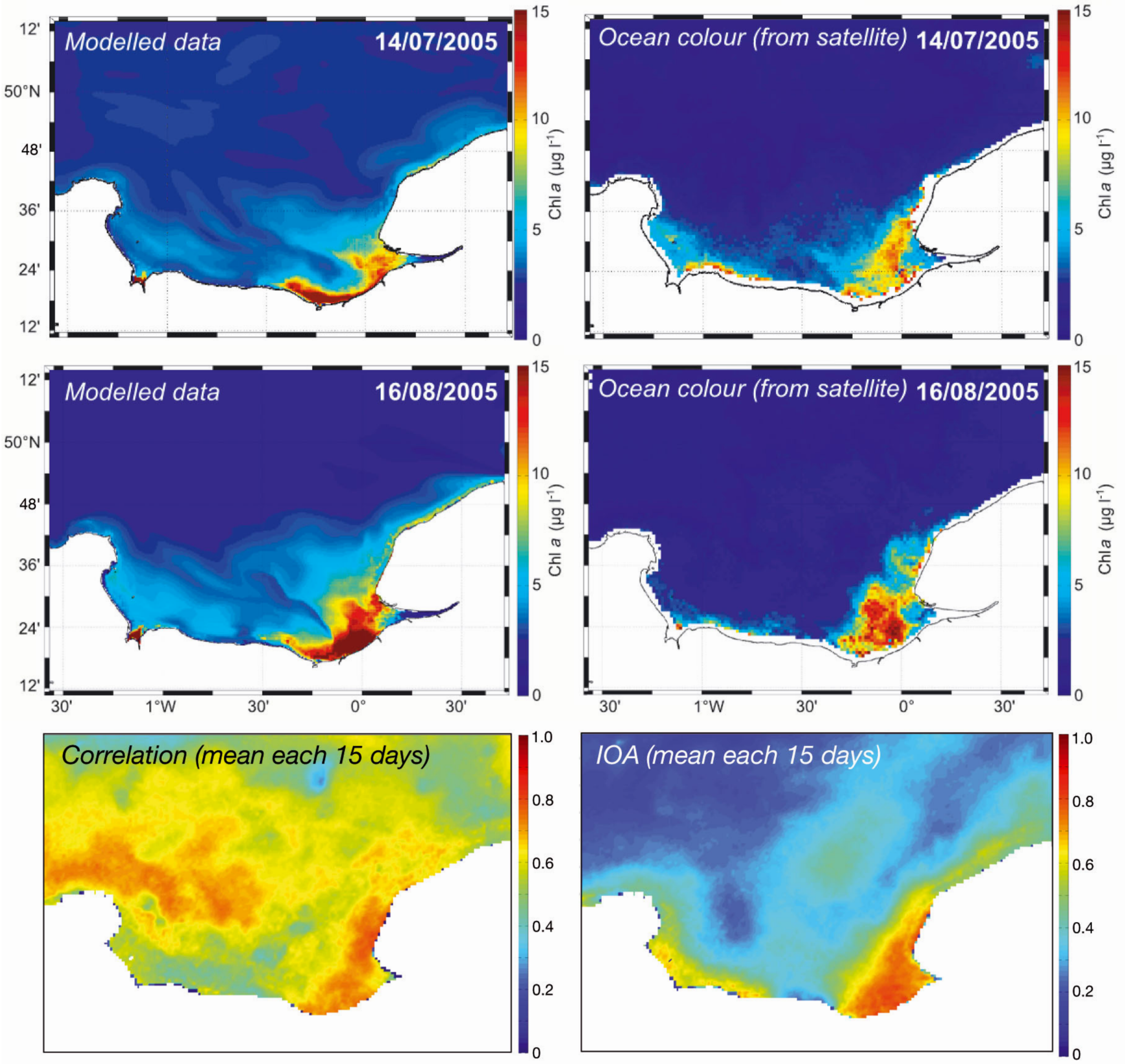

Fig. 7. Simulation of chl *a* concentrations with 'Bay of Seine' spatial configuration (left), and comparison with the corresponding ocean colour observed by satellite (right), for the 14 July (top) and 16 August 2005 (middle). Spatial validation, correlation (bottom left) and index of agreement (IOA; bottom right). Dates given as dd/mm/yyyy

treatment, releasing 1 g N-NO_3^- inh.$^{-1}$ d^{-1}, 0.25 g N-NH_4^+ inh.$^{-1}$ d^{-1} and 0.1 g P-PO_4^{3-} inh.$^{-1}$ d^{-1}. The second one (ORGFARM) is a theoretical scenario implementing organic farming in all arable lands and grasslands of the modelled watersheds. This second scenario reduces the diffuse sources and acts especially on N pollution (Benoit 2014, Anglade 2015). In this scenario, sub-root and phreatic NO_3 concentrations for arable lands and grasslands are reduced to 3 g N-NO_3^- l^{-1} according to the value defined by Thieu et al. (2009). The 2 scenarios were run for the 2000 to 2006 period. These scenarios were previously discussed for the Belgian coastal zone, where eutrophication is caused by mucilaginous species (*Phaeocystis* spp.) rather than toxic algae (Lancelot et al. 2011, Thieu et al. 2011, Gypens et al. 2013, Passy et al. 2013).

Impact in terms of nutrient fluxes

Simulations with the watershed SR model show that the SWT scenario would significantly reduce the P fluxes (by 30%) delivered to the coastal zone, e.g. from a reference state of about 150 kg P km^{-2} yr^{-1} (12 Gg P yr^{-1}) in 2001 at Poses to about 90 kg P km^{-2} yr^{-1} (8 Gg P yr^{-1}) (Fig. 8b,d). The combination of both scenarios, SWT and ORGFARM (SWT + ORGFARM), would not lead to an additional decrease of P. This is explained by the fact that in the zone studied, P orig-

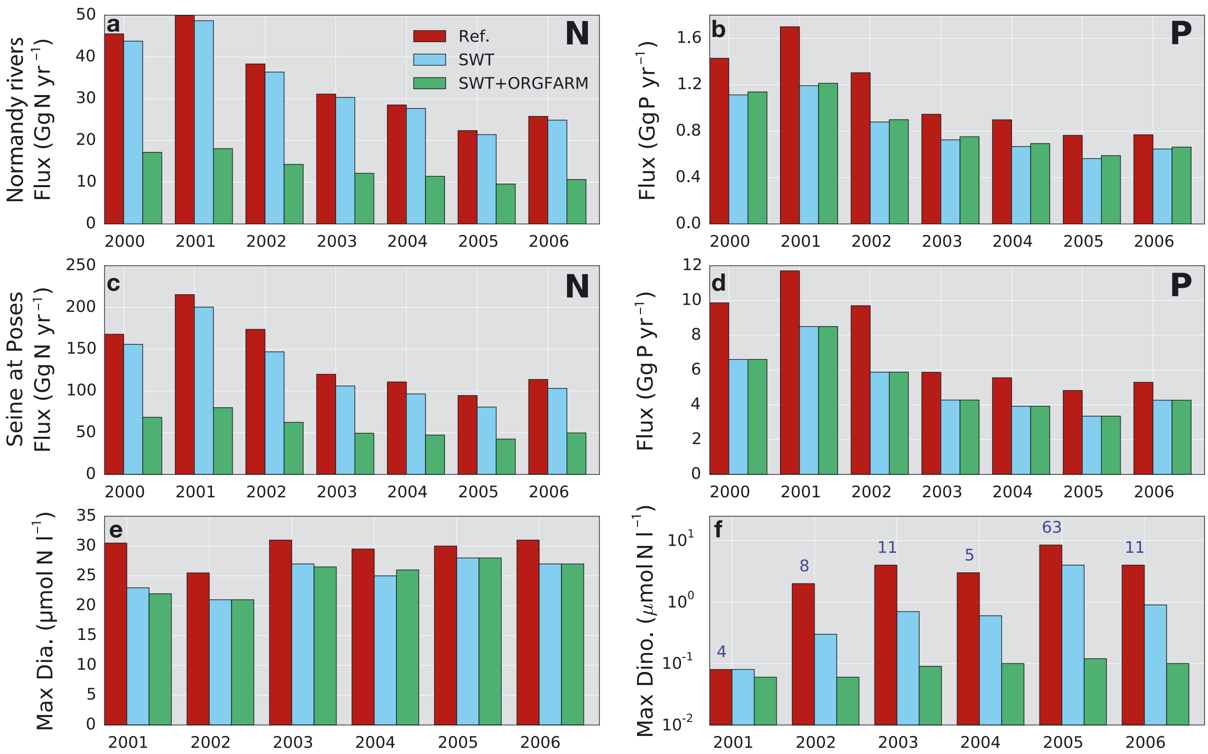

Fig. 8. Absolute annual (a) N fluxes and (b) P fluxes cumulated for the 11 Normandy rivers for the reference, SWT and SWT + ORGFARM scenarios (see section 'Nutrient reduction scenarios' for description of scenarios). Absolute (c) N and (d) P fluxes at the outlet of the Seine River (at Poses). Annual maxima of (e) diatoms and (f) dinoflagellates in the Bay of Seine over the 2001 to 2006 period for the 3 scenarios. (f) Numbers of toxic events measured for each year are indicated on the maximum dinoflagellate graph

inates to a large extent from point sources, diffuse P sources from erosion being relatively low in such lowland watersheds, despite intensive cropping (Cerdan et al. 2010). Concerning N, the SWT would only slightly decrease the N fluxes, despite the treatment of nitrification and denitrification within the WWTPs (Fig. 8a,c). The proportion of N input to the basins by WWTPs is indeed very low (11.3 %) compared to the remaining 88.7 % from diffuse sources. Consequently, only a scenario dealing with agriculture is able to modify N fluxes (Fig. 8a,c). This is shown by the ORGFARM scenario, which would drastically reduce (–50 %) the N fluxes delivered to the coastal zone. For instance, at Poses in 2001, the wettest year of the period, N fluxes decrease from about 3200 kg N km^{-2} yr^{-1} (215 Gg N yr^{-1}) in the reference scenario to about 1200 kg N km^{-2} yr^{-}1 (80 Gg N yr^{-1}) with the ORGFARM scenario. Concerning silica, these scenarios would not significantly impact the fluxes, in accordance with Thieu et al. (2011). Although the ORGFARM scenario is somewhat unrealistic over the short term, this simulation exercise evidences the importance of N management in agrosystems (Sutton et al. 2009).

Impact of scenarios on the coastal zone

We compared the annual maxima of diatom and dinoflagellate concentrations for the reference, SWT and ORGFARM scenarios (Fig. 8e,f). As shown in Fig. 8e, the maximum diatom concentration does not change much according to the scenario. On the other hand, the maximum concentration of dinoflagellates varies greatly according to the scenario (Fig. 8f). When comparing the concentration of dinoflagellates simulated by the model to the number of toxic events caused by dinoflagellates provided by the REPHY network, a strong parallel appears (R^2 = 0.96 including the 2005 year and R^2 = 0.56 without this year). Their impacts are examined over the 2000 to 2006 period at the scale of the Bay of Seine.

The top panel of Fig. 9 shows the mean chl *a* concentrations over the 2000 to 2006 period, in the Bay of Seine, for the different scenarios. The spatial distribution remains the same, but the hot spots of chl *a* concentration decrease, markedly for the ORGFARM scenario (4 to 5.5 µg l^{-1} versus 8 µg l^{-1} for the reference), and less for the SWT scenario (6 to 7 µg l^{-1}).

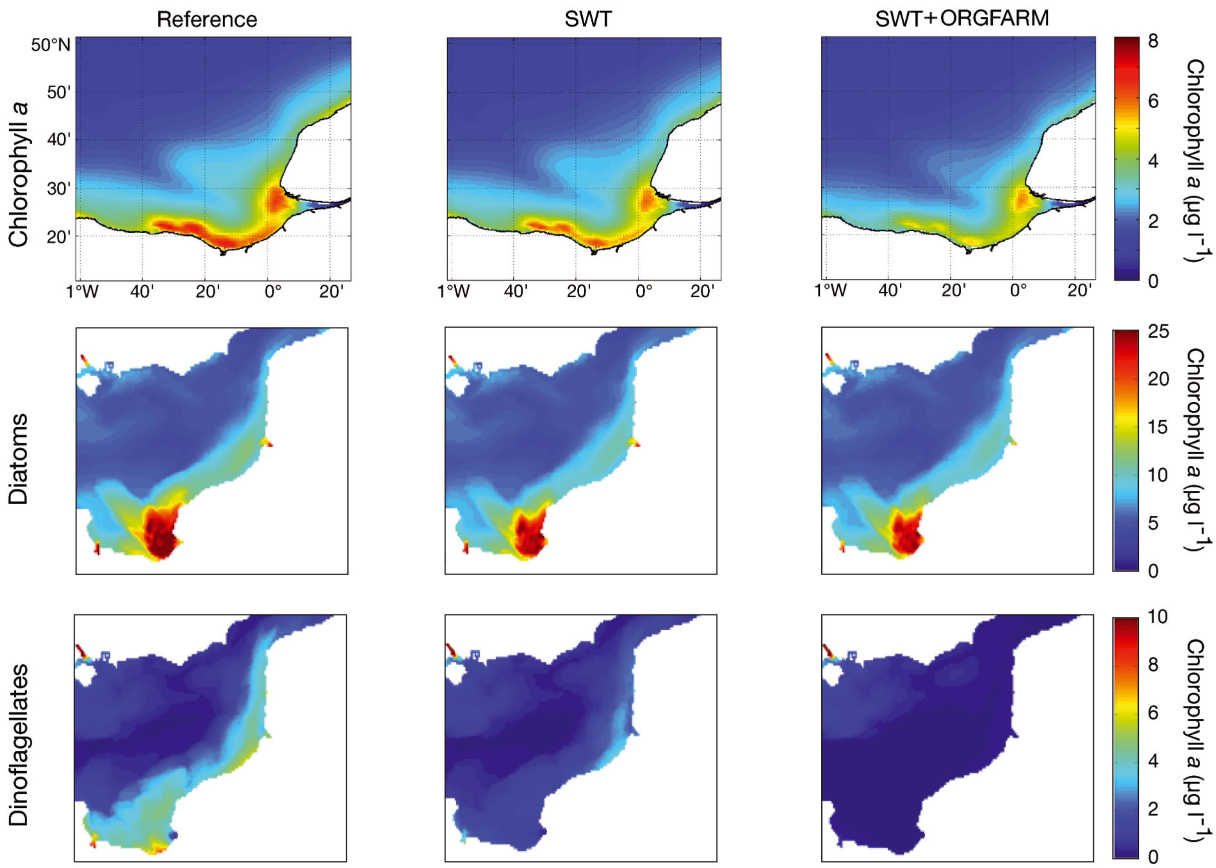

Fig. 9. Top: inter-annual mean (2000 to 2006) of chl *a* concentrations. Left to right: reference situation (2006), SWT and SWT + ORGFARM scenarios. Middle: annual maxima of diatoms for the reference situation (2006), the SWT and the SWT + ORGFARM scenarios. Bottom: annual maxima of dinoflagellates for the reference situation (2006), the SWT and the SWT + ORGFARM scenarios (see section 'Nutrient reduction scenarios' for description of scenarios)

Maximum diatom concentrations decrease for both scenarios with regard to the reference state, but abundance levels remain high at 20 to 30 µmol l^{-1} (Fig. 9). The impacts of the scenarios are stronger on dinoflagellates (Fig. 9). The SWT scenario would reduce 3- to 5-fold the concentration of dinoflagellates, but the SWT + ORGFARM would reduce concentrations by a factor of 20 to 40. For instance, for the year 2005, the concentrations would shift from 9 to 0.2 µmol l^{-1}. This ORGFARM scenario shows that any measures aiming to decrease the excess of fertilisers on agricultural lands would reduce the number of dinoflagellates.

DISCUSSION

The modelling chain coupling the watershed SR model and the marine ECO-MARS3D model appropriately captures the dynamics of nutrient concentrations at the outlet of the watersheds and in the Bay of Seine, and it is very useful to assess changes in nutrient fluxes under different future scenarios. The seasonal dynamics and the total river inputs are in accordance with the measurements. As discussed in previous studies (Grizzetti et al. 2012, Passy et al. 2013, Romero et al. 2013), P fluxes have decreased considerably owing to technological improvements in WWTPs and following national and/or European directives (Urban Waste Water Directive, Water Framework Directive). On the other hand, N fluxes from agriculturally diffuse sources remain high (Lassaletta et al. 2009, Romero et al. 2013), despite the progressive implementation of good agricultural practices.

Similar situations, with a remarkable drop in phosphate but high values of nitrate, are found in other European catchments (Grizzetti et al. 2012, Romero et al. 2013). This shift in nutrient loads has been found to reduce the total biomass of phytoplankton, but does not seem to ameliorate the occurrence of algal blooms.

Additionally, by using a 500 m spatial resolution and by realistically taking into account the Seine estuary, ECO-MARS3D can simulate the spatial–temporal dynamics and the SM content of the MTZ. This is important because the MTZ strongly affects primary production, especially at the direct outlet of the estuary, and its role is frequently neglected in the models. The importance of the MTZ was thoroughly discussed by Irigoien & Castel (1997), who found that changes in turbidity could drastically reduce the primary production in the Gironde estuary, directly due to light attenuation. This light attenuation may also lead to a proliferation of bacteria (Goosen et al. 1999).

The comparison between observed and simulated surface chl *a* showed that its magnitude and spatial extent are suitably reproduced in the Bay of Seine. The simulations of the 2 phytoplankton groups (diatoms and dinoflagellates) could only be qualitatively compared to the abundances of 2 harmful species *Pseudo-nitzschia* spp. and *Dinophysis* spp., and partially validated using data collected during ASP concentrations in 2004 for the former and DSP events in 2005 for the latter. The model results are, in any case, consistent with these data.

With these validations in the continental and marine domains, we have explored scenarios that could guide managers and stakeholders. Our result is coherent with previous studies (Cugier et al. 2005a, Lancelot et al. 2014) that explored a pristine scenario, even more unrealistic than the ORGFARM one. The scenarios have stronger impacts on dinoflagellates, whose concentrations may be reduced 3- to 5-fold with the WWTP scenario and 20- to 40-fold with the organic farming scenario. Given that *Dinophysis* spp. are part of the dinoflagellate group, we can hypothesise that DSP crisis events would also be reduced. Although the mechanism of toxic production is not yet clearly known, the imbalance of N, largely in excess compared to P and Si, is suspected to foster the production of domoic acid by the algae (Hagström et al. 2010). Besides, the concentration of diatoms, which are the dominant primary producers in the Bay, remains high even under the organic farming scenario.

The potential mechanisms are well explained in Billen et al. (2007) and in Romero et al. (2013). Diatom growth does not decrease when N decreases because the N supply is large enough to sustain their growth, even in the reduction scenarios. Indeed, it is likely that Si (and possibly P), not N, limits their growth. If so, diatoms consume all available Si and the corresponding N load, but large amounts of N still remain in the water and are available to non-siliceous algae after the diatom bloom. Dinoflagellate blooms tend to occur right after the typical spring diatom bloom, and in the summer. Contrary to diatoms, most dinoflagellates are more opportunistic because, in addition to their lack of Si requirement, they are motile. So they can move up and down within the water column to benefit from the higher P concentrations found in bottom waters. Therefore, diatom growth is not affected by N load reduction, whereas the amount of leftover N after the diatom bloom limits the growth of dinoflagellates and the occurrence of toxic algal blooms.

Despite the drastic decrease in P load over the past 2 decades, the Bay of Seine marine ecosystem remains fragile. The current challenge is clearly to reduce diffuse N fluxes originating from agriculture. There are 2 main possibilities to reduce them. The first is curative by reintroducing ponds or wetlands, which are nitrogen traps in the landscape (Passy et al. 2012, Billy et al. 2013, Mander et al. 2013). However, this solution is land consuming, and the effects are limited at the scale of the whole catchment (Passy et al. 2012). The second is preventive and aims to decrease fertiliser use by implementing precision farming (Zhang et al. 2002) or organic farming (Pimentel et al. 2005, Mondelaers et al. 2009, Benoit et al. 2014). It seems that the second is more efficient (Garnier et al. 2014).

In conclusion, it is crucial to adequately model the input of nutrients in the Bay of Seine in order to resolve the causes of eutrophication and develop potential mitigation strategies. The model provides information on the change in nutrient fluxes after the implementation of certain measures and on their effect on the consequences of eutrophication including toxic algal blooms. Our results suggest that further technological improvements in WWTPs, which are costly in terms of energy and money, may not be enough to reduce toxic coastal blooms. As already shown in previous studies, today's challenge is really to better manage and reduce the diffuse N sources from agriculture. The potential improvement in water quality associated with changes in agricultural practices is shown in our ORGFARM scenario. It should be noted, however, that the hydro-agronomic system in this scenario considers that sub-root and phreatic NO_3^- concentrations are in steady state. Owing to the residence time of water in aquifers, this steady state would not be reached for several decades. Consequently, even if this scenario was applied today, its effects at the coastal zone would only be fully achieved in a few decades' time.

Finally, the determinants of *Pseudo-nitzschia* spp. and *Dinophysis* spp. development are still too poorly characterised to explicitly take them into account in modelling approaches. Some studies have suggested a propensity for toxin production when the N:P ratio increases (*Dinophysis* spp.: Murata et al. 2006) or when P and Si are limited (*Pseudo-Nitzschia* spp.: Trainer et al. 2012), but more research is needed on the biology and physiology of these 2 species.

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