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TOPICAL REVIEW

Hydromorphology of coastal zone and structure of watershed agro-food system are main determinants of coastal eutrophication

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Abstract

For a number of well-documented watersheds and their adjacent coastal zones, a simplified, but generic approach was developed to explore current nutrient deliveries to their corresponding marine system, characterized by their flushing rate/residence time and morphology. An indicator of eutrophication was defined derived from both the C:N:P:Si stoichiometry of the riverine nutrient delivery and the physical features of the receiving marine bay (B_ICEP). Results show that the morphological and hydrological conditions characterizing coastal zones are the main determinants of the manifestation of eutrophication caused by an imbalance of nitrogen (and/or phosphorus) with respect to silica in the river nutrient loading. Action on the structure of the agro-food system of the upstream watershed, which determines the nitrogen losses to the hydrosystem, is identified as the most efficient control for attenuating coastal eutrophication. A comprehensive and generic concept of the systemic processes responsible for river and coastal water degradation can be achieved with a chain of nested models, describing the terrestrial agro-food system of the watershed, the river network, including the biogeochemical processes responsible for water quality, and the ecological functioning of the receiving marine area, in terms of carbon, nitrogen, phosphorus, and silica cycles. This leads to a land-to-sea continuum view, promoting interdisciplinarity and dialogue among the various scientific communities and their modeling approaches. This would also help the actors in multiple sectors (farming, fisheries, tourism, etc) and policy-makers make harmonized choices for a sustainable environment through an economically and socially viable way of life for all citizens.

1. Background

Coastal zones worldwide are experiencing increasing eutrophication with considerable environmental and economic damage since they receive nutrients transported from the land. The problem constitutes a key environmental issue of the Anthropocene era (Diaz and Rosenberg 2008, Glibert *et al* 2014, Beusen *et al* 2016, Vilmin *et al* 2018). Extreme cases of coastal eutrophication have been reported in most regions of the world, particularly in Europe (Billen *et al* 2011, Mccrackin *et al* 2018a), China (e.g. Bohai Sea: Cui *et al* 2018; Yellow sea: Liu *et al* 2018), and North America (e.g. Gulf of Mexico: Turner and Rabalais 1994, Turner *et al* 2003a).

The manifestations of eutrophication are diverse and take various forms including harmful algal blooms (HABs), either accumulating mucilaginous material (such as Phaeocystis), producing neurologic or diarrheic toxins (such as dinoflagellates, Cyanobacteria, Pseudo-nitzschia), or leading to an accumulation of biomass on the beaches (such as *Ulva lactuca*). The overproduction of this algal material, which is not suitable for consumption by zooplankton, possibly leads to hypoxia of the bottom water layers or to hydrogen sulfide emission when accumulated on the beaches. It is now widely recognized that eutrophication of surface waters, both marine and freshwater, is not merely a result of high inputs of anthropogenic nutrients, nitrogen (N) and/or phosphorus (P), but rather of their imbalance, when N and P of anthropogenic origins are introduced in excess of silica (Si) arising naturally from rock weathering (Billen and Garnier 1997, Turner *et al* 1998, 2003a, 2003b; Liu *et al* 2012).

N and P originate from different sources within a watershed. Because of its propensity for adsorption on soil particles, P is mostly released from agricultural soils in particulate form as a result of erosion, while the highly soluble nitrate ion is leached in high quantities from intensively fertilized agricultural soils. Moreover, N and P are both abundant in untreated urban wastewater, representing a significant source of nutrients for river waters. Indeed, rivers have long been considered as systems suitable for the disposal of pollution from human activities far into the sea. However, since the 1970s, eutrophication of both rivers and marine waters has been recognized as a serious environmental threat. Measures to combat eutrophication first targeted point sources of P, assumed to be the most limiting nutrient in freshwaters, while N was most often limiting in marine systems (Elser *et al* 2007). The success of these policies, particularly in Europe (Grizzetti *et al* 2012, Romero *et al* 2013, 2016, Pistocchi *et al* 2019), the United States (Rabalais *et al* 2002, Turner *et al* 2007), and China (Shu and Finlayson 1993, Wang *et al* 2006, Ma *et al* 2020), led to a significant reduction in P load and to the elimination of eutrophication problems in some river systems (e.g. Loire: Minaudo *et al* 2015; Seine: Garnier *et al* 2019b; Ebro: Torrecilla *et al* 2005). Because N loading, now mainly deriving from diffuse agricultural sources, was not simultaneously reduced, a significant excess of N over P in river loading became the rule, in addition to the excess of N over Si, and eutrophication problems persisted in coastal zones. Measures taken to date to reduce N losses from agriculture have been much less effective than those devoted to urban wastewater treatment, both because of the difficulties in tackling diffuse versus point sources of pollution and because of agricultural intensification. The latter has taken place in the United States since the 1930s, after the Great Depression (Berlan *et al* 1981), in Europe since the 1950s, with the postwar Marshall Plan (1947) and the implementation of the Common Agricultural Policy (1962), and in China since the reforms and opening of the market in the 1980s (Schwoob 2014). Agricultural activities are indeed an important part of the economy, specifically in Europe, the United States, and China, and intensification and specialization of agriculture are clearly the major source of nutrient contamination in water bodies (and particularly N) for river networks and their receiving coastal zones.

Livestock production has significantly increased during the last 50 years in China and during the 1960–1990 in Europe and the USA, mainly thanks to an intensification and disconnection of the production system (Lassaletta *et al* 2016). In Europe and China, the livestock production increase is mainly based on feed import, which has grown tremendously (Lassaletta *et al* 2014, Bai *et al* 2018). In the USA, cropping systems have become highly disconnected from livestock (van Grinsven *et al* 2015, Spiegal *et al* 2020). As a result, cropping systems without associated stock boosted application of mineral fertilizers, whereas in areas of high livestock concentration manure is produced far above crop requirements (Garnier *et al* 2016, Zhang *et al* 2019, Mueller and Lassaletta 2020). The concentrated manure surplus may be either directly dumped into the rivers (Strokal *et al* 2016) or excessively applied to cropland (Perego *et al* 2012, Van Grinsven *et al* 2016, Mccrackin *et al* 2018b, Billen *et al* 2019), from which large nutrient losses are ultimately washed to the freshwaters. Moreover, significant amounts of N are emitted to the atmosphere during manure management, which is not the case for P, resulting in new imbalances of the N:P ratio in the nutrient loads exported to the waterbodies (Bouwman *et al* 2017, Penuelas *et al* 2020).

N due to intensive agriculture in the watersheds is therefore greatly in excess over P and Si with respect to the needs for diatom growth causing coastal zone eutrophication (Billen and Garnier 1997).

Besides agricultural intensification, climate change might also enhance eutrophication. In areas with higher net precipitation higher river discharge would increase nutrient deliveries, leading to more primary production, high biomass possibly accelerating hypoxia, and reducing habitats (Glibert *et al* 2014 , Wåhlström *et al* 2020). Conversely, in regions with lower net precipitation, reduced water flow could increase residence time along the land-to-sea continuum and decrease dilution of nutrients (Whitehead *et al* 2009, Garnier *et al* 2018, Raimonet *et al* 2018). In addition to water flow, temperature increases might affect species succession favoring HAB (Peperzak 2003 , O'Neil *et al* 2012). Ecosystems responses to both climate change and eutrophication may be hard to unravel and thus difficult to predict, but current knowledge suggests that climate change will worsen the present situation.

Nutrient-enriched enclosed bays are more vulnerable to HAB compared with open coasts. Besides nutrient inputs, coastal eutrophication depends on many factors linked to the morphology and hydrological pattern of the receiving water body—flushing rate or residence time and light conditions, etc (Duarte *et al* 2009, 2015, Berthold *et al* 2018, Friedland *et al* 2019). Clearly, a high flushing rate of coastal water bodies by oceanic water masses can rapidly dilute the riverine nutrient load and prevent any manifestation of eutrophication, which can also occur because of high turbidity.

Understanding the conditions for the recovery of coastal ecosystems requires consideration of the complexity and non-linearity of the underlying biological processes that respond in a complex way to nutrient reductions in a changing environment (Duarte *et al* 2009, 2015, Berthold *et al* 2018). Very few studies deal with the land-to-sea continuum, integrating agricultural activity in the watershed for the quantification of diffuse nutrient loading (N especially, and P), and analyzing the impact of nutrient delivery to the marine coastal zone (Lancelot *et al* 2011, Passy *et al* 2016, Desmit *et al* 2018, Garnier *et al* 2019a). Likewise, few studies are concerned with the fact that river nutrient loading is determined by the structure of the whole watershed agro-food system, defining the way food supply to the population is organized through agricultural activities and commercial exchanges, and how the resulting wastewater discharges are managed. Moreover, most databases available for following eutrophication processes over the long term focus on N and P and do not consider Si, for which too few data exist.

The aim of this study is to present a comprehensive and generic concept of land-to-sea continuum, taking into account the dominant processes responsible for river and coastal water degradation, the delivery of nutrients resulting from inputs to the watersheds, and the losses from the agro-food system. Here, the water-agro-food system (see Garnier *et al* 2016) is extended to the ecological functioning of the receiving coastal areas, in terms of carbon (C), N, P, and Si cycles. Starting from the Indicator of Coastal Eutrophication Potential (ICEP) developed by Billen and Garnier (2007) expressing the potential for eutrophication of coastal zones on the basis of N:P:Si ratios in riverine delivery, we hypothesized that the morphology and hydrodynamics of the marine coastal zone are major determinants of the manifestation of this potential eutrophication (Turner and Rabalais 1994). For the purpose of our analysis, a simplified coastal model was developed and applied to a selected number of well-documented watersheds and their adjacent eutrophic coastal zones.

2. Methodological approach

2.1. Study areas and data sources

To link human activities (i.e. point and diffuse sources) and coastal eutrophication, information on eutrophication was collected for major rivers and adjacent coastal zones across a variety of watersheds in Europe, the United States, and China (figure 1, S1 (available online at stacks.iop.org/ERL/16/023005/mmedia)). There are several incidents of coastal eutrophication in Europe, namely, in France, Italy, Spain, Germany, and Poland where case studies were chosen (Seine, Po, Ebro, Oder, and Vistula; Billen *et al* 2011). In the recent decades, however, Greece has developed agricultural activities in the alluvial plains of the Pinios, Aliakmon, and Axios rivers, where better management of fertilization is also necessary (Fytianos *et al* 2002, Stefanidis *et al* 2016). Despite a significant reduction in P levels in most of these selected rivers, their receiving coastal bays still present some of the episodic eutrophication symptoms mentioned above (northern Adriatic bay for the Po River: Cozzi and Giani 2011; Aegean Sea for the Axios River during 2005–2007: Nikolaidis *et al* 2009 ; Seine Bight in 2004 and 2005, Passy *et al* 2016, Ménesguen *et al* 2018, Garnier *et al* 2019a, Thorel *et al* 2017; Alfacs Bay for a branch of the Ebro: Busch *et al* 2016, Quijano-Scheggia *et al* 2008, Fernández-Tejedor *et al* 2009; Gdansk Gulf for the Vistula River and Pomeranian Bay for the Oder and more generally the South Baltic Sea during 2008–2009: Kudryavtseva *et al* 2019, Berthold *et al* 2018). We further selected two other emblematic river–coast systems in the world, namely, the Mississippi and its receiving coastal zone of the Gulf of Mexico (e.g. Turner and Rabalais 1994, Turner *et al* 2003a) and the Yellow River (Huanghe) with its associated coastal zone in the Bohai Sea (e.g. Zhou *et al* 2018, Liu *et al* 2019), both marine areas subjected to significant environmental damage. For all these selected river basins, intensive agriculture is present with heavy environmental effects; water availability for agriculture has been controlled either by tile drainage (Randall and Gross 2008) (5% of the watershed area in the Seine basin: Meybeck *et al* 1998; approximately 30% in the Oder and Vistula basins: Behrendt *et al* 1999), or through irrigation, particularly in the Mediterranean rivers of the Ebro (Causape *et al* 2006), Po (Perego *et al* 2012), and Axios (Nikolaidis *et al* 2009), in reservoirs and canals (for flood control and irrigation: Yellow River, Liu *et al* 2015; and the Mississippi, e.g. Triplett *et al* 2008, Downing *et al* 2016, Royer *et al* 2020), as well as intensive aquaculture systems supplying large nutrient loads to the system (Yellow River; Wang *et al* 2020)

For each of the selected river systems, the nutrient riverine deliveries to the coastal zone were analyzed considering the total amount of nitrogen (N), phosphorus (P), and dissolved silica (Si), and their respective ratio. Information was drawn from both the scientific literature on measured water quality concentrations at the outlet after the 2000s and from the simulations of the GREEN model for European rivers (Grizzetti *et al* 2021, see in S1). For Si, only measured dissolved Si concentrations from the literature were used. Data on water fluxes are also found in the literature, which compare well to the GREEN model for EU rivers, and typically represent

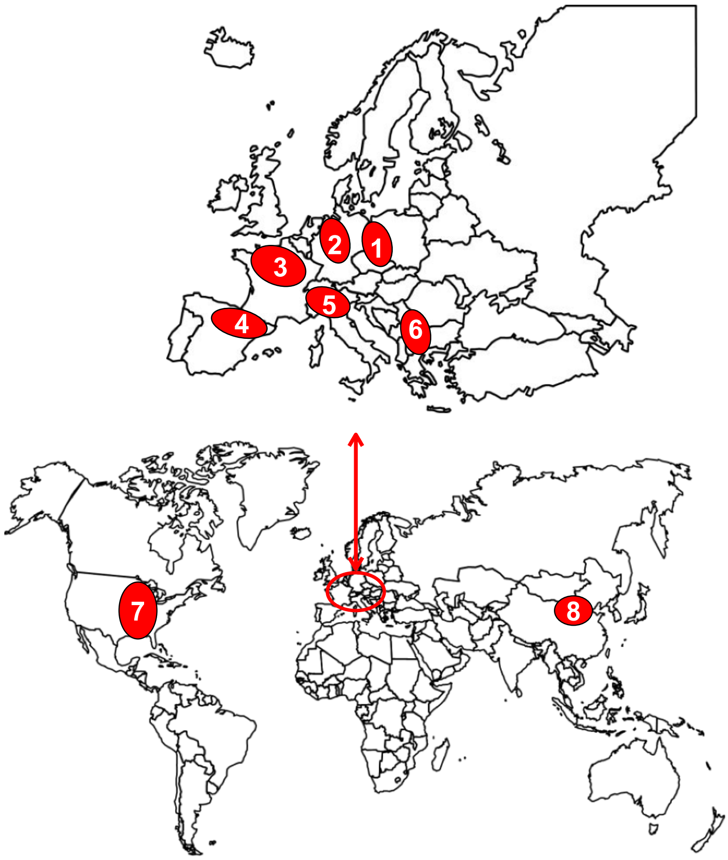

Figure 1. Location of the selected river basins: 1—Vistula (Poland); 2—Oder (Poland, Germany and Czech Republic); 3—Seine (France); 4—Ebro (Spain); 5—Po (Italy); 6—Axios (Greece and Macedonia); 7—Mississippi (U.S.); 8—Yellow River (China).

annual averages over approximately 10 years. For the requirements of the modeling exercises, the water fluxes were seasonalized using their monthly variations found in published papers or from national official sources (Mississippi: O'Connor *et al* 2016); https://waterdata.usgs.gov/; (Yellow River: Wang *et al* 2006); Seine: www.hydro.eaufrance.fr/; other EU rivers: www.eea.europa.eu/data-and-maps/data/.

The intensity of human activities, particularly agriculture, with respect to the dynamics of the N cycle was characterized using the Net Anthropogenic Nitrogen Inputs (NANI) indicator (Howarth *et al* 1996, Billen *et al* 2010, Swaney and Howarth 2019), defined as the sum of inputs of new reactive N into the watershed (not considering the recirculated N, e.g. livestock production and manure), including application of synthetic N fertilizers, symbiotic N fixation by N-fixing crops (especially leguminous crops), atmospheric deposition of oxidized forms of N (originating from transport and industrial activities) and the net import of N embedded in agricultural products such as food, feed, and fiber. This indicator reflects the level of anthropogenic N enrichment of the land-water system of the watershed, independently on the structure of the agro-food system.

2.2. An indicator of coastal eutrophication potential (ICEP)

An indicator of coastal eutrophication potential (ICEP) based on nutrient deliveries by river watersheds was proposed by Billen and Garnier (2007) (see also for the global scale Garnier *et al* 2010, and for Europe, Billen *et al* 2011, Romero *et al* 2013). It is defined as the excess of N or P over Si with respect to the requirements for balanced diatom growth according to the N:P:Si ratios (Redfield *et al* 1963, Conley *et al* 1989). For comparisons between N and P, ICEP is expressed in carbon mass unit and for comparison among rivers, ICEP is normalized per square kilometer of watershed area (Wa). An ICEP value close to zero indicates equilibrium between N or P and Si, whereas positive or negative values mean an excess or deficit, respectively, with respect to Si (Billen and Garnier 2007). ICEP is based on total N, P, and Si fluxes, since nutrients contained in dissolved or particulate organic matter and/or adsorbed onto particles can be mineralized/desorbed and contribute to eutrophication. ICEP can be calculated seasonally, and is expressed per day.

2.3. A marine bay-specific indicator of eutrophication (B_ICEP)

To assess whether or not this potential eutrophication can occur, we expanded the scope of the ICEP indicator. According to their morphological and hydrological particularities, the pelagic ecosystem of coastal bays can react quite differently to a given nutrient loading. The B_ICEP (for bay-integrated indicator of coastal eutrophication potential) was developed by combining ICEP (kgC d^{-1} km^{-2}) with some characteristics of the receiving bay. The B_ICEP is defined as the ratio of the riverine flux of nutrients in excess over Si to the water volume of the receiving bay multiplied by its flushing rate.

$$\mathrm{B\_ICEP} = 1000 \times \frac{[\mathrm{ICEP} \times \mathrm{Wa}]}{[\mathrm{Volb}\ \times \mathrm{fltot}]} \tag{1}$$

where Wa is the watershed area (km^2), Volb (m^3) is the volume of the receiving bay, and fltot, the flushing rate (d^{-1}) of the bay by both marine currents from the surrounding sea water bodies (flm) and by the river flow (flr).

Thus defined, the B_ICEP, expressed in mgC l^{-1}, represents the maximum concentration of non-siliceous algae that can be developed based on excess riverine N and P over Si in a given marine bay. In particular, B_ICEP-N (computed using ICEP-N) indicates the expected effect of an excess of N over Si and B_ICEP-P (based on ICEP-P) of the surplus of P over Si.

The flushing rate of a given bay can be empirically calculated from the river discharge, the volume of the bay, and the difference in mean salinity of the bay (Sal) with respect to that at the outlet of the river mouth (Salr) and that of the offshore water bodies (Salm) as follows (figure 2):

$$\mathrm{flm} = \frac{\mathrm{Qr}}{\mathrm{Volb}}\left[1 + \frac{(\mathrm{Sal} - \mathrm{Salr})}{(\mathrm{Salm} - \mathrm{Sal})}\right] \tag{2}$$

where fltot (d^{-1}) is the total flushing rate of the marine bay, Qr (m^3 d^{-1}), the discharge of the river, and Volb (m^3), the volume of the receiving bay.

The total flushing rate of the bay can be considered as the sum of the flushing rates by the river and by the marine currents:

$$\mathrm{fltot} = \mathrm{flr} + \mathrm{flm} = \frac{\mathrm{Qr}}{\mathrm{Volb}} + \frac{\mathrm{Qr}}{\mathrm{Volb}} \times \frac{(\mathrm{Sal} - \mathrm{Salr})}{(\mathrm{Salm} - \mathrm{Sal})}. \tag{3}$$

Therefore, from the analysis of the literature, special attention was given to document the geometry of the studied bays in terms of surface area and depth, as well as the salinity values at the river outlet, on average within the bay and in the offshore waterbody.

2.4. An idealized model of coastal bay eutrophication

In order to explore the eutrophication response of marine ecosystems to hydrological and chemical loading controls, and to assess the relevance of the B_ICEP approach, an idealized model of algal development in the water column of a marine bay was established. This model is similar to the ZOCO model developed by Billen and Garnier (1997) and Garnier and Billen (2002). It represents the major ecological processes in a pelagic ecosystem as material fluxes between nutrients (N, P, Si), diatoms, and non-siliceous algae, zooplankton, dissolved and particulate organic matter pools, and heterotrophic bacteria. The processes taken into account, their kinetics, and the value of the parameters are described in S1 (table S1, S2, see figure 3).

The model first considers an isolated offshore pelagic water column in order to provide the limit conditions of the marine coastal zone. The latter is considered to receive the flow of water from the former, at a rate depending on the marine flushing rate defined above, as well as the flow of river water dependent on the seasonal variations of the river discharge (figure 3). The model thus simulates the seasonal variations of diatom and non-diatom blooms in the coastal bay in response to nutrient fluxes brought by both the river and the offshore area.

2.5. A suite of models for exploring the impact of the water agro-food system at a coastal bay: the Seine land-to-sea continuum

Whereas the approach described above allows us to analyze the response of the coastal zone to particular seasonalized riverine fluxes, modeling riverine deliveries as impacted by agricultural practices requires an integrated modeling approach of land use and agricultural practices to calculate nutrient losses from agriculture (specifically N because of its higher mobility from crops to water courses), routed into a drainage network model able to calculate the transformation of nutrients along their transfer down to the outlet of the river. Several approaches have been identified, among which the SPARROW model (Smith *et al* 1997, Alexander *et al* 2008), GREEN (Grizzetti *et al* 2012, 2017, 2021), the SWAT model (Gassman *et al* 2007, Neitsch *et al* 2011, Arnold *et al* 2012, Yuan *et al* 2018), the IMAGE.GMN model (Beusen *et al* 2015, 2016, Bouwman *et al* 2017), and the GRAFS-Riverstrahler model (Billen *et al* 2013, 2014, Garnier *et al* 2018, 2019a) have been widely used.

The GRAFS-Riverstrahler model has been specifically developed for the Seine River and its adjacent bay (Passy *et al* 2016, Romero *et al* 2018, Garnier *et al* 2018, 2019a) and applied at the scale of the west European Atlantic coast (Desmit *et al* 2018). To our knowledge, it is the only model that has already simulated the ecological functioning of the land-to-sea continuum, i.e. including the impacts of riverine

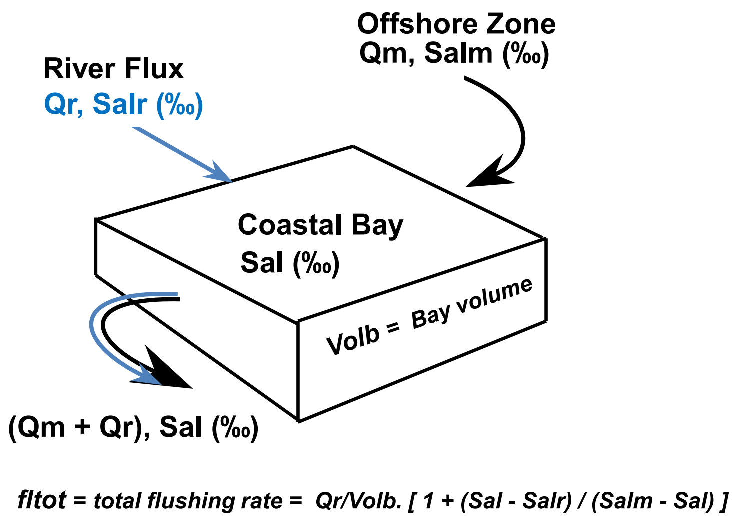

Figure 2. The conservation equation of salinity in a hypothetic coastal bay.

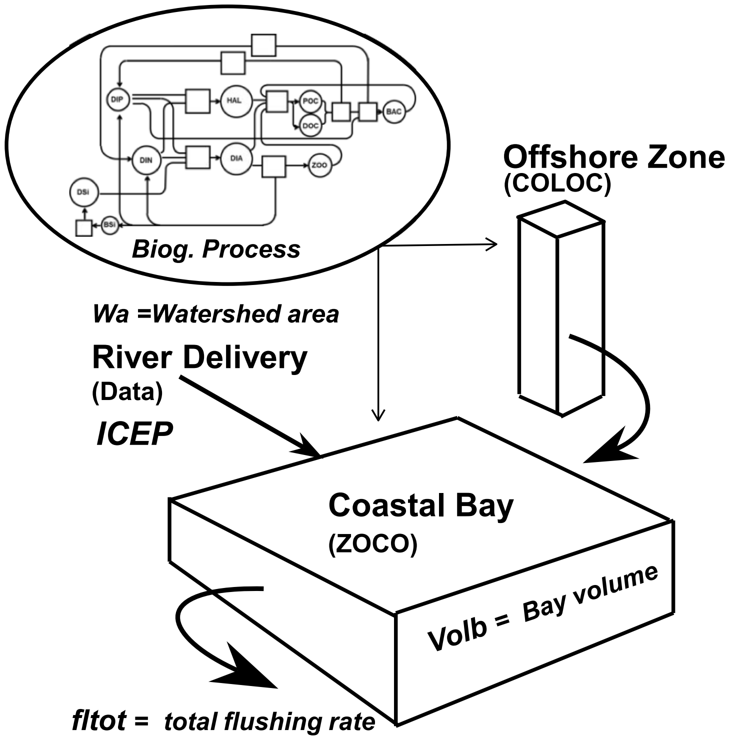

Figure 3. Idealization of the interactions between the coastal bay, the river flow, and the offshore area in the ZOCO model. Biogeochemical processes are schematically represented.

deliveries to the coastal zones when coupled with the Eco-Mars 3D ecological marine model (Cugier and Le Hir 2002, Lazure and Dumas 2008). Whereas Riverstrahler is a biogeochemical model of river networks from headwaters to large rivers (Billen *et al* 1994, Garnier *et al* 2002), GRAFS (Generalized Representation of Agro-food Systems) makes it possible to provide an estimate of diffuse agricultural sources of nutrients based on a soil balance (Oenema *et al* 2003) for which all inputs and outputs are quantified, the difference determining the surplus possibly reaching surface waters. Initially implemented for regions (Billen *et al* 2013, 2014), GRAFS may be applied to farms and small catchments (Anglade *et al* 2015, Garnier *et al* 2016), to the country scale and to historical trajectories (Le Noë *et al* 2017, 2018), as well as to the global scale (Billen *et al* 2014, Lassaletta *et al* 2016). GRAFS also allows us to explore agricultural scenarios (Billen *et al* 2018), which, when coupled with Riverstrahler, leads to an environmental evaluation, e.g. in terms of water quality (Desmit *et al* 2018, Garnier *et al* 2018, 2019a,b).

The GRAFS-Riverstrahler tool was thus chosen for its coupling with the ZOCO model described above, so as to cross simulation results from the Seine watershed under agricultural scenarios with characteristics determined for the bay.

3. Imbalanced nutrient fluxes from watersheds as related to agriculture

3.1. Riverine nutrient fluxes and potential coastal eutrophication

The watershed areas, discharge, nutrient concentrations, and loading of the chosen sample of river systems are presented in table 1. The nutrient (im)balance at the river outlet can be quantified using the ICEP indicator. Owing to the lack of data on seasonal water quality, yearly averaged ICEP values were used.

For the studied rivers, ICEP-P values are all close to zero, although slightly negative for the Po and Axios rivers, as well as for the Mississippi and Yellow rivers, and slightly positive for the rest (table 1). This result suggests that a balance has been reached between P and Si, with the generalization of dephosphatation treatment of wastewater in these basins. Regarding ICEP-N, the values are largely positive for the Seine and Po rivers (~15 kgC km^{-2} d^{-1}), and still positive for the Oder, Vistula, and Ebro rivers (3–8 kgC km^{-2} d^{-1}) as well as for the Mississippi and the Yellow rivers (1–2 kgC km^{-2} d^{-1}). Similar to ICEP-P, ICEP-N is negative for the Axios River (<−1 kgC km^{-2} d^{-1}; table 1).

In Europe, the EU Urban Waste Water Treatment Directive (EU-UWWTD 1991) and the EU Water Framework Directive (EU-WFD,2000) has resulted in member states attaining a good ecological status for natural rivers, while adjustments are being made for strongly impacted rivers, with flexibility among member states depending on the reference conditions. In France, point sources have been considerably reduced with the application of the EU-WFD (2000), by 95% for P and only 60% for N for the largest wastewater treatment plants (Rocher and Azimi 2017). For the Seine, as an example of a strongly human-impacted river, N delivery to the coastal zone has increased by 10%–20% whereas P loads have decreased by 55% during the same period (Billen *et al* 2001, Passy *et al* 2013, Garnier *et al* 2019a). Germany has achieved reductions from point sources of approximately 70% for P and 13% for N for the past two decades (Nausch *et al* 2011, in Berthold *et al* 2018). For Poland, Kowalkowski *et al* (2012) also showed that P loads have been reduced more than N loads for the Oder and Vistula rivers. Similar achievements can be observed across Europe, leading to different responses among countries: while P fluxes are generally reduced (especially in the North Sea and Baltic Sea), N fluxes remain relatively stable (Grizzetti *et al* 2012, Romero *et al* 2013). For the study period after EU-WFD inception in 2000s, the levels of Si have not changed much in these river systems, because reservoirs were built mostly between the 1920s and 1980s (Cruzado *et al* 2002), so that no additional Si trapping associated with new water infrastructure has occurred. Moreover, as eutrophication in the river continuum has decreased due to P reduction, Si uptake by diatoms has even decreased, possibly increasing Si deliveries to the river outlet (Garnier *et al* 2019b). A low ICEP-P thus is expected in most EU river basins, while ICEP-N is not expected to change.

Similarly, in China huge efforts have been made since the early 2000s to solve the problem of sanitation, thereby partly reducing P pollution in the Yellow River and Bohai Bay (Liu *et al* 2015, Liu *et al* 2019), but not in the Yangtze River (Liu *et al* 2018). However, agricultural activities, including aquaculture, have been significantly intensified, increasing N contamination that could level off in this period after the policy 'Zero Growth in Synthetic Fertilizer Use from 2020' was introduced in 2015 (MOA 2015b in Wang *et al* 2020). In addition, water abstraction has also negatively affected water quality because many reservoirs have been constructed for flood control and for irrigation, reducing the dilution of point sources. The measured discharge of the Yellow River to the sea was reduced by a factor of 3.6 between the periods 1950–1959 and 1990–2000 (i.e. to ~420 m^3 s^{-1}, presently) (Wang *et al* 2006). This freshwater reduction was accompanied by a southeastward shift of the downstream course of the river (Yang *et al* 2011). In such a managed and N-enriched river, a relatively low ICEP-N, indicating low N deliveries compared with those of Si, might be surprising and explained by the comparatively high Si loads. Although Si might be retained in the reservoirs and other stagnant irrigation canals or ponds lining the channel, rocks weathering in this basin might have been enhanced leading to elevated Si concentrations and fluxes. Indeed, with a drier and warmer climate and overgrazing, lands have experienced desertification (sandification) and permafrost alteration in the most upstream areas of the Yellow River (Wang *et al* 2001), and thus the Si level is higher than in other Chinese rivers (Liu 2015). In addition, according to these authors, biogenic Si is subject to dissolution in the estuarine section, protecting the coastal zone from strong eutrophication. Even in the huge three Gorges Dam in the Yangtze River, only limited silica retention has been reported (Ding *et al* 2019).

In the Mississippi River basin, Royer *et al* (2020) reported positive ICEP-N and ICEP-P values in the Upper Mississippi and Ohio–Tennessee sub-basins. During the 2000–2015 period, ICEP-N decreased in the Ohio–Tennessee region, but not in the Upper Mississippi or in the whole of the Mississippi. It is

Table 1. Characteristics of the studied rivers (drainage area, annual discharge, and nutrient concentrations), net anthropogenic nitrogen input (NANI), and calculated indicator for coastal eutrophication potential (ICEP) for N and P (ICEP-N, ICEP-P). (For further information, i.e. nutrients fluxes, see-S1).

Rivers	Vistula	Oder	Seine	Ebro[a]	Po[b]	Axios	Mississippi	Yellow R.
Drainage area (km^2)	193 894	118 938	73 224	85 611/3590	71 327/89 684	24 397	3 240 000	752 000
Discharge (outlet) ($m^3 s^{-1}$)	1040	547	583	277/11.7	1513/1819	101	21 220	420
mgN l^{-1}	3.35	4.93	5.12	2.27	2.71	1.96	2.10	4.90
mgP l^{-1}	0.23	0.32	0.21	0.17	0.17	0.21	0.14	0.20
mgSi l^{-1}	3.8	3.7	3.7	1.0	3.1	4.9	2.2	3.5
NANI (kgN $km^{-2} yr^{-1}$)	3432	4191	6307	3681	5623	2536	3061	4632
ICEP-N (kC $km^{-2} d^{-1}$)	4.8	7.8	14.2	3.0	14.8	0.0	3.9	1.0
ICEP-P (kC $km^{-2} d^{-1}$)	0.3	1.9	0.3	1.3	−0.1	−0.8	0.4	−0.3

[a] Branches and Canals of the Ebro Delta feed two small confined bays, the bays of Alfacs and Fangar, south- and northward, respectively. The Alfacs Bay, which is more documented in the literature, was chosen here. For drainage area and discharge values, the first figure corresponds to the whole Ebro basin, the second to the Alfacs Bay.

[b] The coastal zone taken into account for the Po additionally receives input fluxes from three smaller rivers (Adige, Brenta, and Piave Rivers) that are taken into consideration for their respective discharges and watershed areas. For drainage area and discharge values, the first figure corresponds to the whole Po basin, the second includes the Adige, Brenta, and Piave Rivers.

noteworthy that ICEP-P has increased and Cyanobacteria development has become a threat in P-enriched riverine environments (Royer 2020). Considerable Si trapping in the numerous—large or small—reservoirs in the Mississippi basin further increases ICEP-N or ICEP-P values and strengthens the potential for eutrophication.

As N appears to be the most critical factor for the eutrophication of land-to-sea systems, further analysis will focus on N.

3.2. Relationship between watershed nitrogen inputs (NANI) and potential coastal eutrophication (ICEP-N)

The Net Anthropogenic N Input (NANI; Howarth *et al* 1996) is a good indicator of the intensity of N input to watersheds, reflecting the amount of reactive N potentially exposed to be lost to the environment and therefore to the hydrosystem, either through leaching from agricultural soils or through the release of urban wastewater. The NANI values of all studied watersheds (expressed per km^2 of watershed area) reported in table 1 were taken from a previous study by Billen *et al* (2010), based on the Global News database (Seitzinger *et al* 2010). Plotting the ICEP-N values for each river against their respective NANI clearly shows a significant linear relationship (figure 4). Therefore, ICEP-N appears to be clearly driven by NANI. A target for delivered nutrient fluxes would be to reach values close to zero for ICEP-N (which is already the case for ICEP-P here) in line with several works postulating the need for a reduction in both N and P to control primary production (Howarth and Marino 2006). However the role of Si cannot be ignored. In fact the depletion of Si leads to a shift from diatoms to HAB, the former palatable for the zooplankton constituting the food of plankton-eating fish, the latter fueling the microbial loop (Justić *et al* 1995a, Justić *et al* 1995b, Billen and Garnier 1997, Heisler *et al* 2008, Viaroli *et al* 2008, Howarth *et al* 2011).

Noteworthy, some studies consider that P should be further reduced, especially in the rivers flowing to the Baltic Sea, where P can be adsorbed and resuspended with very long residence times, and where Cyanobacteria, an algal group particularly able to store P in their cells (Ritchie *et al* 2001) and capable of fulfilling their N needs through atmospheric N_2 fixation (Molot 2017), dominate the HABs. However, several studies recommend avoiding nutrient stoichiometric imbalances, and prone integrated measures of P and N reductions (Howarth and Marino 2006, Conley *et al* 2009, Glibert *et al* 2014). The ICEP indicator is precisely based on the nutrient balance of algal requirements, although eutrophication is a complex mechanism involving not only the quantities and proportions of nutrients but also their forms (Glibert *et al* 2014).

4. Fate of nutrients delivered to coastal bays

The ICEP provides information on the nutrient balance delivered by river fluxes and the resulting potentiality to support new primary production of harmful algae. It does not, however, consider the variety of characteristics of the receiving coastal systems.

4.1. A marine bay-specific eutrophication indicator: the B_ICEP

The volume of the receiving coastal bay and its flushing rate through offshore water currents are of great significance in determining the impacts of nutrient (im)balance on the potential for eutrophication.

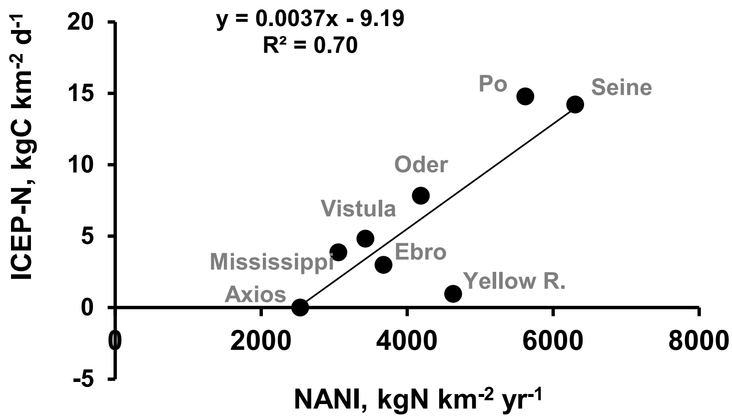

Figure 4. Relationship between the indicator of coastal eutrophication potential (ICEP-N, kgC km^{-2} d^{-1}) and the NANI of the river basin (kgN km^{-2} yr^{-1}). NB: for the Ebro, only the branch flowing to Alfacs Bay, south of the delta, is considered.

These factors vary seasonally and interannually; however, given the availability of the data, average annual values are considered here. As mentioned above, the marine flushing rate of a geographically delimited coastal zone was calculated from its average salinity values (and its difference with offshore water masses) (see equation (2)). The flushing rate by the river is derived from the volume of the bay and the delivered discharge. The total flushing rate (in day^{-1}) is the sum of these two components (figure 1; table 2). The residence time (expressed in day) can be directly inferred from the total flushing rate.

The new indicator B_ICEP-N, computed using ICEP-N by taking into account the volume and the total flushing rate of the receiving bay, is intended to be related to the maximal phytoplankton biomass. This indicator expressed in mgC l^{-1} represents the possible non-siliceous algal biomass concentration (often in the form of an undesirable algal bloom) that can be produced in the bay in excess over that built up in offshore water. Plotting the observed maximum phytoplankton biomasses, for the period after the 2000s in the receiving bay of the eight selected systems, against the corresponding B_ICEP-N values reveals a positive linear trend, validating the idea that B_ICEP-N is a good predictor of the manifestation of eutrophication in the receiving coastal bays (figure 5). This approach therefore emphasizes the importance of the morphology and water dynamics of the bay for the manifestation of coastal eutrophication.

The highest B_ICEP-N is the one for the Mississippi Bay, for which the flushing by the river, enriched in nutrients, is more than three times higher than that of the other rivers. The Po and Seine rivers, with a similarly high ICEP-N, are well differentiated in terms of B_ICEP-N, which is much higher for the Seine Bight than for the part of the Adriatic bay receiving the Po. Concurrently, the maximum algal bloom of 50 μg l^{-1} chlorophyll *a* for the Seine Bight is double that of the Adriatic Bay (25 μg l^{-1}). No eutrophication risk is found for the Vistula and Axios bays. These results clearly show that a high ICEP-N at the outlet of a river can be modulated by the flushing rate in the receiving bay; a high marine flushing rate lowers the risk of eutrophication by bringing a considerable amount of Si and less N and P, while the opposite occurs when the flushing rate by the river dominates (cf. Mississippi).

Setting B_ICEP-N at zero to simulate a lack of eutrophication potential allows us to determine target N concentrations at the outlet of each of the rivers, in accordance with the EU-WFD goals. On average, this leads to an inorganic N concentration in river water of 1.6 mgN l^{-1} (from 0.9 mgN l^{-1} for the bay of the Mississippi to 3.35 mgN l^{-1} for the inner Gulf of Gdansk). Interestingly, such threshold concentrations defined for avoiding coastal eutrophication are in the range of those already recommended for maintaining vegetal (James *et al* 2005) and animal (Camargo *et al* 2005) biodiversity in freshwater systems. Also, the thresholds determined based on ICEP are relatively consistent with the estimated N boundary between Good and Moderate Ecological Status for large European rivers (Phillips *et al* 2018) in the range 1.6–2.5 mgN l^{-1}.

Except for the Vistula and Axios rivers for which a reduction in N does not seem necessary, for the other selected rivers the average required N abatement is 67%, ranging from 83% for the branch of the Ebro flowing to the Alfacs Bay, to about 70% for the Yellow River, Oder, and Seine rivers and their bays, and 55% and 57% for the Po and Mississippi respectively (table 2).

4.2. Factors contributing to coastal zone eutrophication

Although ZOCO is a simplified model of a coastal zone, simulated values of maximum phytoplankton biomass compare relatively well with the observed value, with the slope of the relationship at 1.02 (close to 1:1) and the goodness of fit at $R^2 = 0.69$, $p < 0.05$ (figure 6). The model can be considered

Table 2. Characteristics of the corresponding coastal bay (volume, flushing rates, extinction coefficient) and B_ICEP-N. values. Nitrogen concentrations at the outlet were calculated to obtain a B_ICEP-N = 0 (no eutrophication risk) as well as the nitrogen abatement in the river water at their outlet to reach this goal. (For further information, see-S1).

Rivers	Vistula	Oder	Seine	Ebro	Po	Axios	Mississippi	Yellow R.
Adjacent bay	Inner Gulf of Gdansk	Inner Pomeranian Bay	Seine Bight	Alfacs Bay	Coastal front of the Po Delta	Outer delta	Hypoxia Zone of of North Gulf of Mexico	Outer delta in the Bohai Sea (Part of Bohai and Laizhou Bays)
Volume (km^3)	159.0	12.0	50.0	0.5	114.0	1.0	407.5	264.0
Oceanic flushing rate (d^{-1})[a]	0.0046	0.0490	0.0070	0.0250	0.0084	0.2000	0.0015	0.0053
Total flushing rate (d^{-1})	0.0051	0.0528	0.0080	0.0271	0.0098	0.2086	0.0047	0.0055
% Oceanic flushing	0.89	0.93	0.88	0.92	0.86	0.96	0.32	0.97
k (m^{-1})[b]	0.476	0.482	0.5	0.448	0.05	0.11	0.56	0.3
B_ICEP-N (mgC l^{-1})	0.0001	1.4280	2.5800	0.8250	1.1900	−0.0006	5.1456	0.4270
mg N l^{-1}	3.35	1.46	1.49	0.39	1.22	1.96	0.9	1.4
% N abatement	0	70	71	83	55	0	57	71

[a] The flushing rate is the inverse of residence time, i.e. the time necessary to replace the volume of the bay.

[b] Extinction coefficient refers to the attenuation of light in the water column, an important controlling factor of the euphotic zone depth, and hence of primary production.

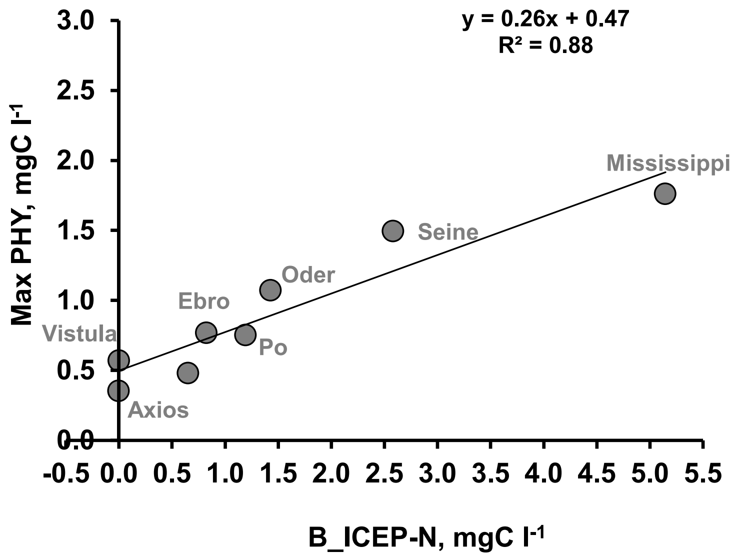

Figure 5. Relationship between the maximal phytoplankton biomasses (mgC l^{-1}) reported in the literature (see S1) and the B_ICEP-N indicator of the various coastal bays. Phytoplankton biomass in mgC l^{-1} is calculated from chlorophyll values taking a C:Chl ratio of 32 (Jakobsen and Markager 2016). NB: for the Ebro, only the branch flowing to Alfacs Bay, south of the delta, is considered.

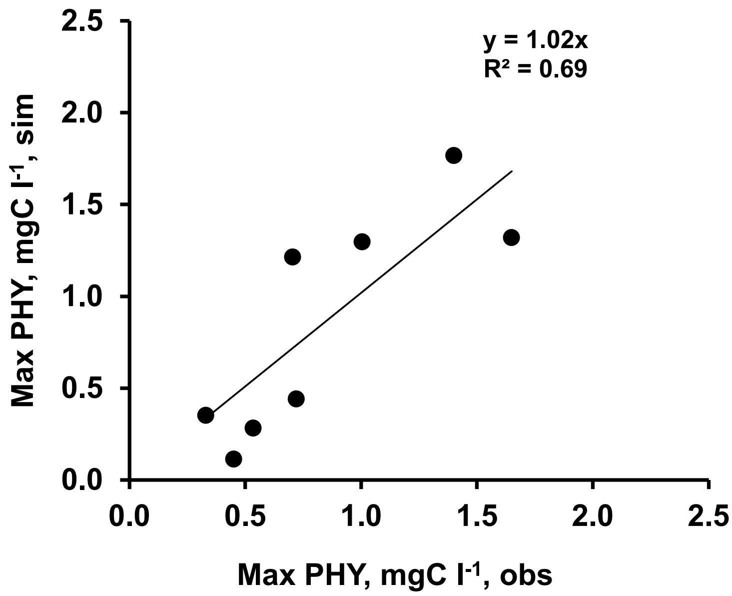

Figure 6. Relationship between the simulated maximal phytoplankton biomasses (mgC l^{-1}) and the ones reported in the literature.

robust enough to evaluate the coastal zone responses to new conditions and constraints.

4.2.1. Effects of physical determinants

We explored the complex relationships between river nutrient deliveries and marine bay characteristics (flushing rate, extinction coefficient) with the ZOCO model. The total flushing rate varied between a low rate (approx. 5% of the present one) and a high one (x3 to x10), i.e. in the range of 0.001–0.015 d^{-1}. The total flushing rate is limited at a low end by the flushing determined by the river discharge. Average salinity in the bay reflects the mixing conditions by the river and the oceanic currents (figure 7).

The maximal biomass of phytoplankton calculated in the respective bays logically decreases as a function of the increasing total flushing rate (figure 7). A low flushing rate, e.g. a residence time of more than 2 years, makes the bay a stagnant system, which leads to a high potential for phytoplankton biomass, limited only by nutrients and available light. Algal biomass decreases with increasing flushing rates. The pattern is different, however, for the Gulf of Mexico, considerably flushed by the Mississippi River, with a maximum phytoplankton biomass occurring at an

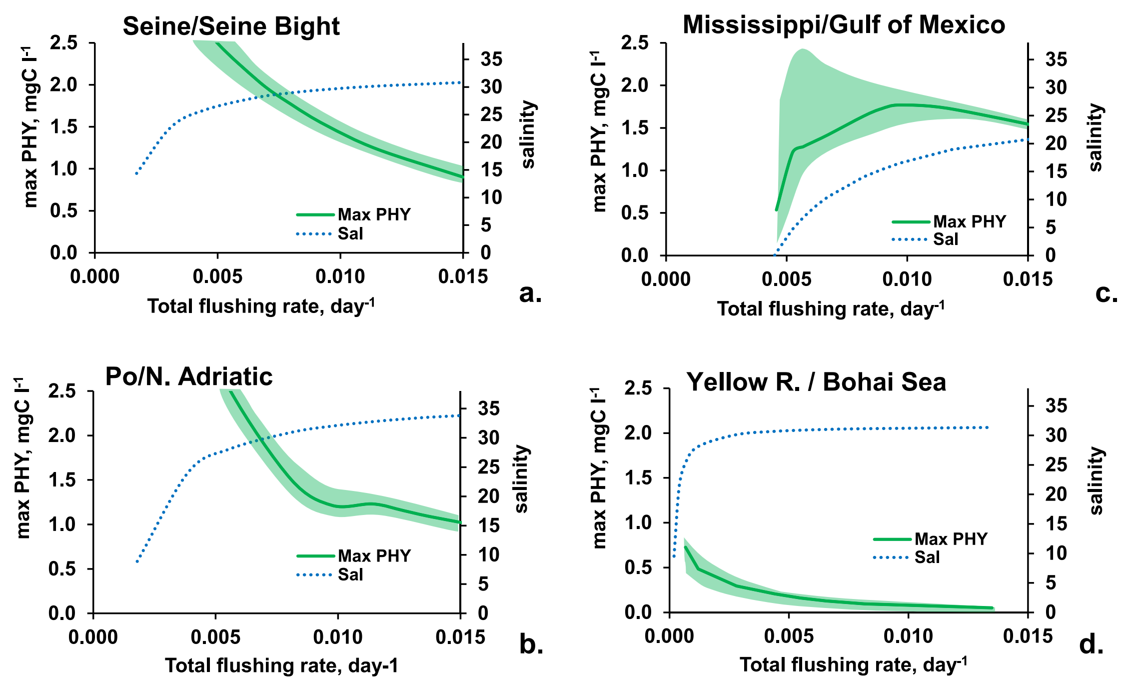

Figure 7. Relationships between maximal phytoplankton biomasses (mgC l^{-1}) and the total flushing rate of the bay receiving riverine nutrient deliveries. These relationships are calculated with the established extinction coefficient (m^{-1}) and a range of ± 0.05 m^{-1} (dotted lines). Relationships between flushing rate and salinity are shown with a secondary *y*-axis.

intermediate total flushing rate, when the proportion of oceanic and riverine flushing allow for optimal conditions in terms of nutrient supply. Algal biomass does not appear to be very sensitive to the value of the extinction coefficient, except for the Gulf of Mexico in which a more significant effect is seen. Note that a biomass of 1.5 mgC l^{-1}, corresponding to a chlorophyll value of 50 μg l^{-1}, is a relatively high value for a coastal marine system, considering a C:Chl ratio of 32 for a 7578 data set in temperate estuarine and coastal open water systems (Jakobsen and Markager 2016).

4.2.2. Role of river nutrient load

We showed in the previous sections how setting the B_ICEP-N at zero or negative values could offer a target for avoiding marine eutrophication. We quantified the N concentration and N abatement (%) necessary to achieve this target in the coastal systems under analysis. Apart from two river basins (Vistula and Axios), this would imply an abatement of approximately 50%–70% of the current riverine N loading (table 2). The two strategies that can be used are reducing (i) the point sources by improving urban wastewater treatment or (ii) the diffuse sources from agriculture. For most river systems in high-income countries, these two actions, particularly the former, have already been widely adopted, and therefore much more ambitious and radical measures have to be taken to combat coastal eutrophication and attain good water quality according to the EU directives (EU-WFD 2000; EU-Marine Strategy Framework Directive, EU-MSFD 2015).

We will illustrate this using the case of the Seine River, for which several prospective scenarios have been established for future agro-food-systems (Billen *et al* 2018) and assessed via modeling (see Desmit *et al* 2018, Garnier *et al* 2018, 2019a). All scenarios consider the fulfillment of the already well-employed implementation of P and N treatment of urban wastewater through dephosphatation, nitrification, and denitrification, in accordance with the EU- -UWWTD and the EU-WFD. As for agriculture, one scenario, called 'O/S,' considers the pursuit of the current trend toward the opening and specialization of the agro-food system, with no structural changes in farming practices, except for the application of the current environmental regulation regarding the rate of N fertilization and the intercalation of catch crops before spring crops to avoid bare soils in winter. A second scenario, called 'A/R/D,' considers a radical change in agricultural systems based on a complete reorganization of the agro-food chain, with a generalization of organic farming practices making farmers autonomous regarding the use of chemicals and importing of feed, as well as the reconnection of livestock and crop farming. The scenario also considers a change in human diet involving a 50% reduction in the current amount of animal protein consumption (Billen *et al* 2018). Two other scenarios were constructed, one ('Pristine') corresponding to the absence of any human perturbation of the watershed, considered as entirely occupied by forest, and the other ('Back to the 1980s') depicting the situation of the basin with a very low level of wastewater treatment and the absence of environmental regulation of agricultural practices. These scenarios also consider the corresponding changes in atmospheric deposition. All these scenarios were run with the model for river system biogeochemical functioning (RIVERSTRAHLER, Billen *et al* 1994, Garnier *et al* 2002), which provides the corresponding nutrients fluxes and concentrations at the outlet of the Seine River (figure 8). Applying these results to the ZOCO model of the Seine Bight interestingly shows that even the radical A/R/D scenario, despite having the best performance in terms of nutrient flow reduction, does not lead to an N level that would fully avoid eutrophication, i.e. the N level required for reaching zero B_ICEP-N (figure 8).

As a whole, for the extreme A/R/D agricultural scenario, the risk of eutrophication might still persist, supporting the view that very profound changes are required to reach the conditions for sustainable water-agro-food systems. It should be noted, however, that progress has already been made since the 1980s when compared with the current situation. Also, the negative value of the B_ICEP-N corresponding to the pristine scenario shows that the efforts required to reach zero B-ICEP should not be unattainable.

5. Science and policy integration

5.1. Policy implications

Our analysis pointed out many elements linking coastal eutrophication to N anthropogenic inputs and, for the Seine Basin, to the structure of the water-agro-food system. Generally, links between continental and marine scientific communities are relatively poor, and scientists studying continental surfaces usually focus on surface waters, ground waters, or soils and agronomy, whereas marine studies often deal with estuaries, coastal zones, and open oceans separately. Collaborations to fully understand the land-to-sea continuum are therefore very rare. One of the first integrated modeling approaches, using the GRAFS-Riverstrahler model coupled with several coastal zone (CZ) models, was applied to the Seine River CZ (Cugier *et al* 2005, Passy *et al* 2013), the Scheldt CZ (Lancelot *et al* 2007), the Seine–Somme and Scheldt CZ (Lancelot *et al* 2011), the Normandy/Hauts-de-France rivers CZ (Garnier *et al* 2019a), and the whole EU Atlantic façade (Desmit *et al* 2018). Among these, agricultural scenarios were modeled only recently, from the 2010s, mostly because an improvement in wastewater treatment plants was expected to be sufficiently efficient for improving the water quality of surface water. However, technological improvement based on end-of-pipe solution to wastewater treatment cannot solve coastal eutrophication because of the dominant impact from intensive agriculture, especially when the receiving bay is not flushed at a very high rate.

Indeed, here we illustrate the degree to which the characteristics of coastal zones could drive the intensity of eutrophication in relation to excess N. The fairly simple B_ICEP indicator of the risk of coastal eutrophication, integrating both the river nutrient load from the watershed and the major hydrological–morphological characteristics of the receiving marine systems, thus appears to be very promising for guiding policies by offering a comprehensive view.

In Europe, the successive directives on continental waters (EU-WFD) and marine waters (EU-MSFD 2015) must be better connected and should promote new integrated studies. In addition, studies of the aquatic continuum from headwaters to the sea should integrate agricultural systems and practices to assess their impact on water quality. The EU Common Agricultural Policy (CAP), which has been regularly evolving since the 1960s with several environmental measures, cannot be disconnected from environmental policies regarding continental and marine waters and biodiversity. The new CAP and the new EU Farm to Fork Strategy (2020) offer the possibility for substantially more locally adapted plans to enable strategies tailored to regional agro-environmental and watershed traits and vulnerabilities.

Regarding the large Mississippi River Basin, which represents two thirds of the surface area of the United States, it was acknowledged in 1999 that hypoxic events of the northern Gulf of Mexico were due to an excess of N inflow (Alexander *et al* 2008, Tian *et al* 2020). The size of this hypoxic zone of 5000 km^2 in the 1980s reached $\sim$15 000 km^2 in the period 2000–2010 (Turner *et al* 2012). In 2001, an action plan was thus set out to reduce 30% of the N load with the aim of reducing the areal extent of hypoxia in the Gulf (Mississippi River/Gulf of Mexico Watershed Nutrient Task Force 2001, Rabalais *et al* 2002, Turner *et al* 2007). As stated by the latter authors, addressing agricultural practices upstream of such large basins in order to reduce hypoxia so far away is a difficult challenge that demands dedicated political willingness. Due to little progress, another plan was developed in 2008 but in 2019 the Gulf of Mexico hypoxic zone was 18 005 km^2 (Ritter and Chitikela 2020).Taking into account the importance of the Mississippi discharge in the total flushing rate of the Gulf supports the need for concerted actions at a regional and a national scale.

In China, riverine and coastal systems have experienced important and rapid changes in terms of nutrients and water regimes associated with the swift economic and social developments and the influx of exogenous nutrients (Zhou *et al* 2018). As in many places, domestic wastewater is a source of N and P, which, however, has been reduced thanks to the expanded capacity of sewage treatment plants and the

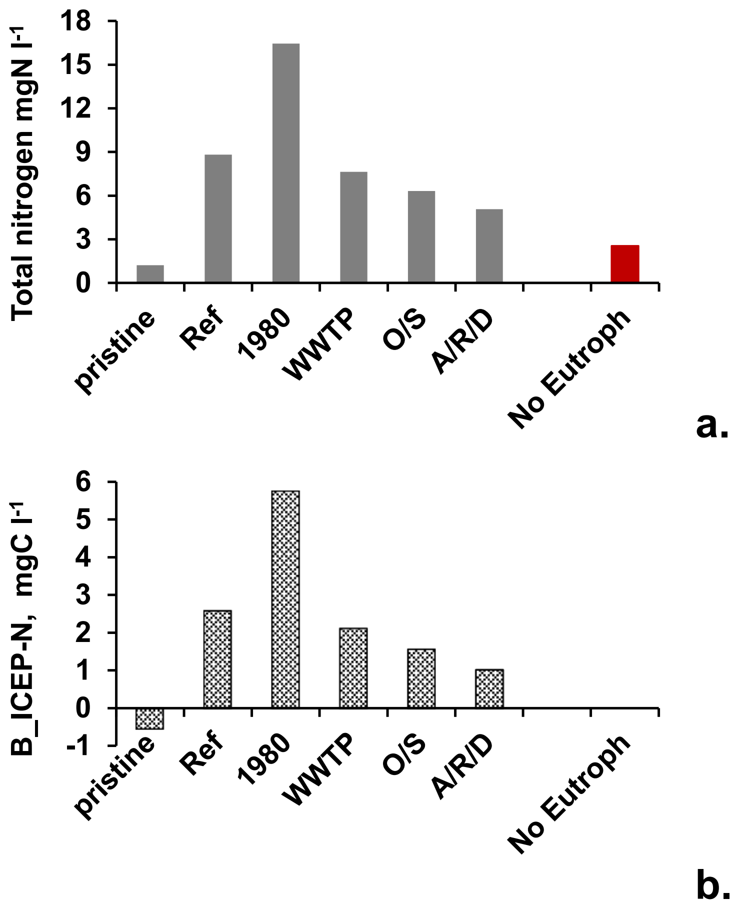

Figure 8. (a) Simulated N concentrations calculated with the GRAFS-RIVERSTRAHLER model (Garnier *et al* 2019a,b) for various scenarios of the Seine watershed (see text). (b) Calculated corresponding B_ICEP-N. (NB: 'No Eutrophication' corresponds to the estimated N concentration that would entail a B_ICEP_N value = 0, i.e. no eutrophication risk once the bay characteristics have been considered).

ban of P in detergents in the early 2000s (Ma *et al* 2020). However, in addition to domestic sewage, agriculture (including marine aquaculture, Bouwman *et al* 2013, Wang *et al* 2020) is an important and increasing source of N in the whole country and in the Yellow River basin in particular, whereas P largely related to runoff, as opposed to wastewater, is decreasing with the dramatic reduction in freshwater and sediment discharge undergone by the river (Liu 2015). Si was also reduced for the same reason, possibly increasing ICEP and B_ICEP, i.e. promoting eutrophication (Liu 2015). Indeed, since the 2000s, beside a reduction in precipitation, more than 3000 reservoirs (four major ones for a total capacity of 57×10^9 m^3) have been built to control flood events, damaging for life and property in Chinese history, but also for irrigation (Wang *et al* 2006). In addition to flood control that was listed in the 1950s as one of the main national priorities in China (with the creation of the Yellow River Conservancy Commission, Shu and Finlayson 1993), other important acts from the central government were instituted to control pollution discharge and improve water use (e.g. the 'Three Redlines' and 'Water Ten Plan'). Ma *et al* (2020) emphasize the need for a more flexible strategy that would better consider regional specificities. The nationwide Grain-for-Green Program was initiated in 1999 as an ambitious conservation measure that aimed to mitigate and prevent flooding and soil erosion (Wohlfarht *et al* 2017). Therefore, diffuse sources from agriculture should be better controlled and integrated into a river basin management plan, i.e. at a basin scale (Xia *et al* 2011), an approach already effective in countries of the EU community (2nd River Basin Management Plans, RBMPs, 2016–2021). Clearly, the Yellow River (and other Chinese watersheds) is experiencing changes that warrant studies for further analysis of coastal eutrophication.

5.2. Future efforts

5.2.1. Completing databases

There is an urgent need to consider the importance of the characteristics of the impacted coastal systems in the manifestation of riverine deliveries, such as eutrophication. Such hydro-physical databases should be widespread and made widely available, together with the variables characterizing water quality, eutrophication, and hypoxia. River water quality and hydrological data are not always harmonized. Whereas the forms of P and N are relatively well studied, data on Si are still too poorly documented, both in rivers and coastal zones, and seasonal variations of these three major macronutrients are rarely available. In addition, inorganic and total forms of these nutrients are not systematically provided together in the literature, which can prevent relevant comparisons among or within studied sites.

5.2.2. Chaining models for capturing the land-to-sea continuums

It is important to identify coastal zone models, not only dedicated to hydrodynamics but also to biogeochemical functioning, for possibly elaborating a generic simplified model to be applied to every coastal bay of interest in the world, fed by riverine deliveries, for which several existing models are good candidates (e.g. GREEN, RIVERSTRAHLER, SPARROW, IMAGE-GNM, Global NEWS, cited above., etc). The C-GEM estuarine modeling approach (Volta *et al* 2014, Laruelle *et al* 2017) can potentially make the link between rivers and coastal zones, especially when estuaries play a buffering role for nutrient retention. For a full integrated view of the land-to-sea continuum, inputs to the rivers from rural areas (diffuse sources from agricultural lands and forest, and possibly point sources from concentrated animal feeding operations and urbanized zones) must also be considered for the development of scenarios. Further, agriculture impacts groundwater that provides the river base flow. When aquifer residence time is high (e.g. up to 50 years for the Seine basin, Ledoux *et al* 2007), it confers a large inertia that may considerably delay the effect of measures taken for better agriculture practices. This effect should also be taken into account in the modeling chain as well as in policy assessment.

5.2.3. Scenario elaboration

Considering a comprehensive approach for the agro-food systems (e.g. GRAFS, Billen *et al* 2013; CHANS, Gu et al, 2012, 2015) based on territorial practices at the scale of world basins can represent an interesting option (Billen *et al* 2015, 2018). Whereas N is the nutrient in excess over Si and P, P losses from soils must not be neglected. Indeed, in many high-income countries, P has accumulated in soils (Macdonald *et al* 2012, Mogollón *et al* 2018, Le Noë *et al* 2020) and despite P having the propensity to be adsorbed onto particles, changes in climate (e.g. rainfall) can enhance erosion and P-particle losses (Zhang *et al* 2017). Scenario elaboration should integrate a variety of scientific disciplines and tools developed in order to define and meet the target of combatting coastal eutrophication. In particular, scholars of human sciences are essential for these questions related to eutrophication and agriculture, suitably qualified for analyzing the desirable solutions, the perceptions, the frustrations, and the efficiency of policies for the well-being of both the population and the environment. Stakeholders (e.g. farmers, water agencies, public authorities, etc) must also be involved in the construction processes of scenarios as they are involved in the efforts. Indeed, analysis of the agricultural scenarios for the Seine River has shown that profound changes in agricultural systems are necessary to reduce coastal water eutrophication, meaning a new paradigm for cooperation between rural and urban territories.

Coastal eutrophication can only be reduced by an active regionally adapted policy of nutrient stoichiometric rebalancing (Peñuelas *et al* 2020), based on an integrative view of the land-to-sea continuum. As an example, the role of reservoirs in silica trapping (Tripplett *et al* 2008, Harrison *et al* 2012, Ran *et al* 2018) illustrates the complexity of the processes involved. A proliferation of small or medium water reservoirs for irrigation in the context of climate change could increase the adverse effect of agriculture on coastal eutrophication, by reducing Si concentrations in river water, while it can also reduce the N load by enhancing denitrification in stagnant systems. It is a matter of urgency to connect scenarios of climate change with those of the water-agro-food systems taking all the facets of the system into account (Ducharne *et al* 2007, Glibert *et al* 2014, 2014).

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Data availability statement

All data that support the findings of this study are included within the article (and in two supplementary information files, S1 and S2).

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