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Reducing marine eutrophication may require a paradigmatic change

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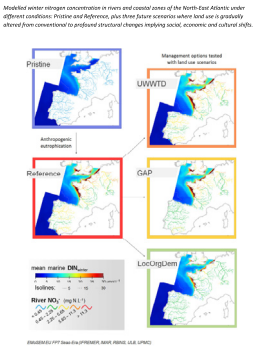
HIGHLIGHTS

- Undesirable eutrophication at sea results from land use in the watersheds.
- Human society could significantly reduce its land-sea nutrient fluxes.
- Four models cover the river-ocean continuum in the North-East Atlantic.
- Management options are tested with scenarios in which land use is gradually altered.
- Significant achievements likely require deep structural changes in human activities.

GRAPHICAL ABSTRACT

Modelled winter nitrogen concentration in rivers and coastal zones of the North-East Atlantic under different conditions: Pristine and Reference, plus three future scenarios where land use is gradually altered from conventional to profound structural changes implying social, economic and cultural shifts.



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ABSTRACT

Marine eutrophication in the North-East Atlantic (NEA) strongly relies on nutrient enrichment at the river outlets, which is linked to human activities and land use in the watersheds. The question is whether human society can reduce its nutrient emissions by changing land use without compromising food security. A new version of Riverstrahler model (pyNuts-Riverstrahler) was designed to estimate the point and diffuse nutrient emissions (N, P, Si) to the rivers depending on land use in the watersheds across a large domain (Western Europe agro-food systems, waste water treatment). The loads from the river model have been used as inputs to three marine ecological models (PCOMS, ECO-MARS3D, MIRO&CO) covering together a large part of the NEA from the Iberian shelf to the Southern North Sea. The modelling of the land-ocean continuum allowed quantifying the impact of changes in land use on marine eutrophication. Pristine conditions were tested to scale the current eutrophication with respect to a "natural background" (sensu WFD), i.e. forested watersheds without any anthropogenic impact. Three scenarios representing potential management options were also tested to propose future perspectives in mitigating eutrophication. This study shows that a significant decrease in nitrogen fluxes from land to sea is possible by adapting human activities in the watersheds, preventing part of the eutrophication symptoms in the NEA rivers and adjacent coastal zones. It is also shown that any significant achievement in that direction would very likely require paradigmatic changes at social, economic and agricultural levels. This requires reshaping the

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connections between crop production and livestock farming, and between agriculture and local human food consumption. It also involves cultural changes such as less waste production and a shift towards lower-impact and healthier diets where half of the animal products consumption is replaced by vegetal proteins consumption, known as a demitarian diet (http://www.nine-esf.org/node/281/index.html).
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1. Introduction

Human-induced eutrophication remains one of the most important anthropogenic pressures on estuarine and coastal waters, with serious consequences for ecosystem functioning (Diaz and Rosenberg, 2008; Nixon, 2009; Doney, 2010; Cai et al., 2011). Cultural eutrophication started with the Great Acceleration in 1950 due to the synergetic effect of increased human population and socio-economic development with a worldwide sixfold increase in fertilisers use, and consequently a two to fourfold increase in nitrogen (N) fluxes and a threefold increase in phosphorus (P) fluxes to the coastal zone (Howarth et al., 1996; Steffen et al., 2011; Beusen et al., 2016). Throughout European watersheds, diffuse nutrient emissions constitute now the preeminent pollution driving coastal eutrophication and the net anthropogenic input of reactive N is five times larger than the background rate of N_2 fixation (Billen et al., 2011). The North-East Atlantic Ocean (NEA) hosts a collection of different coastal ecosystems from the upwelling system of the Iberian Shelf to the turbid, shallow and well-mixed coastal waters of the Southern North Sea. Intense phytoplankton biomass blooms (expressed in chlorophyll *a* unit, Chl) occur in most coastal zones of the NEA between March and October, which is illustrated by the high 90th percentile of Chl (Chl P90, a proxy for the yearly bloom maximum; Fig. 1). Depending on the phytoplankton structure, nutrient inputs induce different eutrophication symptoms with varying ecological responses: high coastal Chl blooms, local hypoxia events, foam events, nutrient imbalance allowing toxic species occurrence, possible changes in coastal community structure (Ménesguen, 1990; Jickells, 1998; Rousseau et al., 2000; Moita et al., 2003; Philippart et al., 2007; Lancelot et al., 2009; Lancelot et al., 2014; Passy et al., 2016).

Marine eutrophication in the NEA directly relies on N and P enrichment at the river outlets and, to a lesser extent, on N atmospheric deposition and upwellings. The relative importance of terrigenous nutrients delivery into the sea increases with the degree of freshwater eutrophication that is sensitive to agricultural practices and waste water treatment policies at the level of the watersheds (Thieu et al., 2010; Hong et al., 2012; Passy et al., 2013). Mitigating eutrophication requires the reduction of nutrient inputs to the rivers (de Jonge et al., 2002; Diaz and Rosenberg, 2008; Billen et al., 2011), preferably by adopting a dual-nutrient reduction strategy to keep a healthy nutrient balance between N, P and dissolved silica (DSi) (Howarth and Marino, 2006;

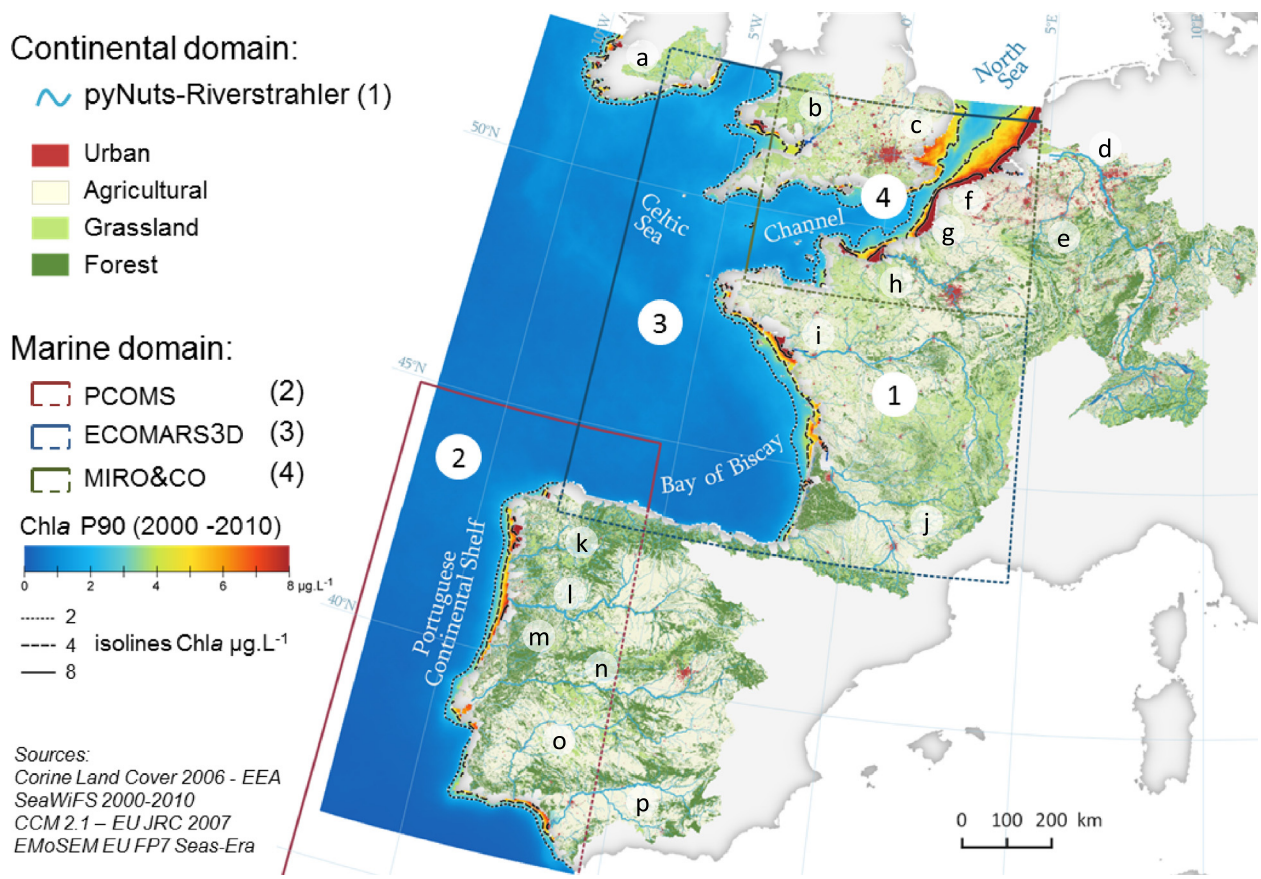

Fig. 1. Multiyear average of marine chlorophyll *a* percentile 90 (SeaWiFS remote sensing 2000–2010, courtesy of F. Gohin/Ifremer). Land colours indicate the density and type of land use. The four model domains are shown: three marine ecological models (see legend) and the ecological model for European river-basins. The selected rivers, used for validation, cover most of the studied catchment area in Western Europe: (a) The Three Sisters, (b) Severn, (c) Thames, (d) Rhine, (e) Meuse, (f) Scheldt, (g) Somme, (h) Seine, (i) Loire, (j) Garonne, (k) Minho, (l) Douro, (m) Mondego, (n) Tagus, (o) Guadiana, (p) Guadalquivir. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

Conley et al., 2009; Desmit et al., 2015a). Three main directives and conventions govern the environmental status of European waters: the OSPAR Convention covers the North-East Atlantic marine waters (OSPAR Commission, 1992), the Water Framework Directive (WFD) focuses on terrestrial ground- and surface waters including the near-shore coastal waters (Directive 2000/60/EC), and the Marine Strategy Framework Directive (MSFD) focuses on national marine waters (Directive 2008/56/EC). Other directives focus on specific causes of eutrophication as e.g., the management of waste waters (Urban Waste Water Treatment Directive, UWWTD, Directive 91/271/EEC) or the use of fertilisers (Nitrate Directive, Directive 91/676/EEC). The first eutrophication assessment in the NEA was led by the OSPAR Convention in the North Sea and resulted in the proposition of reducing both N and P inputs by 50% compared with 1985 (OSPAR Commission, 1988; Claussen et al., 2009). The comprehensive procedure used for the OSPAR assessment in the North Sea (with N, P and Chl as main indicators of eutrophication) has formed a basis for the development of the WFD, which extended the assessment strategy at the scale of Europe. The application of the European policies led to a significant reduction in P after 1990, mainly due to the ban of polyphosphates from washing powder and a better control of the point sources. However, the reduction of N (mainly from diffuse agricultural sources) has been less important and the combination of these trends resulted in an average increase of the coastal N:P ratio. The difficulty in controlling diffuse sources explains the persistence of coastal eutrophication across the NEA (Lenhart et al., 2010; Bouraoui and Grizzetti, 2011; Romero et al., 2013).

A major question to be addressed is: can the Western European society reduce its nutrient emissions to coastal zones by changing its land use without compromising its food security? To link land-based activities and marine eutrophication, the spatial patchiness of human activities in the watersheds and the complex processes of nutrient transport and transformations across the land-ocean continuum must be considered on the fine scale (Bouwman et al., 2013). This advocates for an integrated approach linking freshwater and marine waters with tools such as numerical models and GIS-based models (Ruelland et al., 2007; Thieu et al., 2010; Lancelot et al., 2011). The strength of the modelling tool holds in its capability to explore nonlinear processes at fine spatial and temporal resolution, to test hindcast and predictive scenarios, to assess management options and support future policy alternatives (e.g., Cugier and Le Hir, 2002, Lacroix et al., 2007b, Los et al., 2008, Thieu et al., 2010, Lancelot et al., 2011, Los et al., 2014, Ménesguen et al. 2018, this issue). Also, the numerical models help discussing the threshold values of indicators, such as nutrient concentrations, currently recommended by conventions and policies (Troost et al., 2014).

Previous model studies have investigated the effects of reduced riverine nutrient loads on North Sea ecosystem (e.g. Lenhart, 2001; Lacroix et al., 2007a; Lancelot et al., 2006; Skogen and Mathisen, 2009; Lenhart et al., 2010). Nutrient reduction scenarios were defined by applying a percentage reduction to a reference nutrient load. Whether the percentage reduction was homogeneously applied in all rivers or followed a more refined approach, the proposed reduction scenarios were focused on testing the sensitivity of the marine ecosystem to possible changes. These studies did not, however, focus on how to reach in practice such percentage reduction. In more recent approaches, both the river watersheds and the marine ecosystem were modelled and then coupled together. The reduction scenarios were then designed based on changes in human activities at the scale of the watersheds (Lancelot et al., 2007; Thieu et al., 2010; Lancelot et al., 2011; Passy et al., 2016). Modelling the land-ocean continuum in such a way allows testing the impact in the sea of alternative measures taken upstream in the river basins (e.g., changes in agricultural practices, upgrade of waste water treatment plants, reconnection of crop and livestock farming, of food production and consumption, etc.). Such a "continuum" approach requires to compute: (i) the nutrient point and diffuse sources in the watersheds as a function of land use (Thieu et al., 2010; Passy et al., 2016); (ii) the riverine nutrient transport and transformations along the stream to the coastal zone (Garnier et al., 2002); (iii) the marine processes leading to coastal eutrophication and export of nutrients to adjacent areas (Lenhart et al., 2010; Los et al., 2014; Dulière et al., 2017).

In this study, the land-ocean continuum is simulated across the NEA from Southern Spain to the Southern North Sea by combining four numerical models: the GIS-based generic river model pyNuts-Riverstrahler, using a same set of parameters across the modelled domain, and three marine ecosystem model applications PCOMS, ECOMARS3D and MIRO&CO (Fig. 1). The GIS-based model translates any significant change in the land use into nutrient point and diffuse emissions to the rivers and coastal zones across Western Europe. The marine ecosystem models can then be forced with these river loads and used to assess the effect of human activities on marine eutrophication. With these combined models, it is possible to test policy-relevant prospective scenarios along the land-ocean continuum at regional scales. Our scenario study accounts for multiple causes of nutrient emission and retention in the watersheds and anticipates nonlinear outcomes in the coastal zone. It also assumes possible changes in human activities and cultural choices (e.g., structural changes in agriculture, change in dietary choices) to measure their effect on the marine system. This was carried out with three objectives: (i) to assess a natural background of eutrophication indicators (as planned by the WFD) with a modelled "pristine" situation where European watersheds are forested and not subject to any human perturbation; (ii) to scale the current situation of eutrophication by comparing it with the modelled "pristine" situation; (iii) to test three prospective alternative scenarios where changes in anthropogenic pressures reduce the point and diffuse nutrient emissions to the river network and decrease the undesirable effects of coastal eutrophication. A test case in a Belgian waterbody also illustrates the effect of the scenario on the modelled seasonality of the phytoplankton succession.

2. Methodology

2.1. Area of study

In this study, the NEA consists in the part of the ocean comprised within latitudes 34°N to 53°N and longitudes 13°W to 5°E (Fig. 1). The terrestrial watershed of the NEA domain represents an area of >1 million km^2, with 174 individual fluvial basins of area over 300 km^2.

The NEA domain covers a wide gradient of climate, population density, land use and hydrological conditions. Among the main rivers systems, the Rhine, the Scheldt, the Seine and the Thames Rivers are textbook examples of highly impacted river-basins, dominated by agricultural land, with extended urban area (>1 million inhabitants) and low proportion of natural forest (Fig. 1). Within these river basins, conurbations are more numerous and proportionally bigger compared to other NEA basins. The Meuse River concentrates its population in a few medium conurbations (<200,000 inhabitants). The Minho is the only main basin where forest is dominating. In a large part of the NEA watershed area, there is severe deterioration of ground- and surface water quality. These include nitrate contamination of drinking groundwater resources (over 800 drinking water wells were closed in France during the last decade because of agricultural contamination, Direction Générale de la Santé, 2012), as well as loss of biodiversity and increased eutrophication in rivers and lakes (Sutton et al., 2011).

The Portuguese Continental Shelf width is typically smaller than 30 km. Therefore, the ecological processes are influenced by both terrestrial nutrient loads and ocean-margin exchanges (upwellings). The relative influence of both land- and ocean-based nutrient sources follows the hydroclimatic seasonal pattern with river discharges higher by one order of magnitude in winter than in summer. Nutrient loads are also higher in winter compared to summer when river flows are lower and nutrients tend to be retained or eliminated during their travel within the river and dam network before they reach the marine outlet. The coastal circulation is mostly influenced by the northern wind, which

is responsible for upwelling events (Fiúza et al., 1982). Upwelling conditions enhance the exchanges across the shelf border and act as a significant source of nutrients to the coastal zone. Eutrophication-related problems have not been reported in the coastal zone yet. However, some estuarine areas such as the Mondego estuary have been subject to eutrophication conditions (Marques et al., 2003) while other larger estuaries such as the Tagus and Sado have been considered as non-problem areas from that respect (Instituto da água, 2002).

Northward, whereas the southern Bay of Biscay does not show yet signs of eutrophication, the French Celtic Sea and the western English Channel have seen the nitrate fluxes from Brittany to coastal waters increase tenfold during the last four decades. Eutrophication has become obvious along beaches invaded by considerable outbreaks of *Ulva* sp. On the continental shelf, episodic massive blooms of phytoplankton (diatoms or sometimes dinoflagellates) occur in the plumes of the Loire and Vilaine Rivers (Ménesguen, 1990), where their subsequent sedimentation sometimes result in lethal anoxia events (Chapelle et al., 1994). The N enrichment to the coastal zone is suspected to have triggered the toxicity of some phytoplankton species. For instance, some diatoms belonging to the *Pseudo-nitzschia* genus produce the amnesic shellfish poisoning (ASP) toxin when the N:Si ratio is abnormally high (Fehling et al., 2004) and the dinoflagellate *Karenia mikimotoï* can induce widespread mortality events of wild fishes and benthic invertebrates after anthropogenic nutrient enrichment in the river plumes (Jones et al., 1982; Arzul et al., 1995; Erard-Le Denn et al., 2001). The naturally confined Bay of Seine receives large N loads from the Seine River and hosts high rates of primary production with considerable Chl concentrations some years (30 to 70 μg Chl L^{-1}; Aminot et al., 1997). Diatoms are dominant, but flagellates can become abundant in late summer, including *Dinophysis* sp. that is responsible of diarrheic shellfish poisoning (DSP) toxicity (Sournia et al., 1991).

The English Channel and Southern North Sea are subject to the nutrient inputs of nutrient-rich rivers (Seine, Somme, Scheldt, Rhine/Meuse and Thames), and these nutrients increase along the SW-NE direction (Ruddick and Lacroix, 2006). The combination of Atlantic water inflows, freshwater discharge and wind patterns results in a variety of salinity and nutrient distributions affecting the spatial distributions of phytoplankton production and Chl maximum (Desmit et al., 2015a). The distribution of Chl concentrations in Belgian and Dutch waters typically shows high values in the coastal zone and a decreasing gradient towards the offshore (Schaub and Gieskes, 1991; De Vries et al., 1998; Rousseau et al., 2006). An excess of N (as NO_3^-) is supplied relative to silica that is beneficial to *Phaeocystis globosa* colonies after the spring diatom bloom (Gypens et al., 2007; Baretta-Bekker et al., 2009; Ly et al., 2014). Colonies of *P. globosa* may reach large sizes and become inedible by zooplankton (Weisse et al., 1994; Rousseau et al., 2000). The predator-prey size mismatch and the poor nutritional value of *P. globosa* have been reported to inhibit copepod grazing and egg production in the late spring (Daro et al., 2006). Instead, *P. globosa* detrital matter feeds bacteria. The stimulation of the microbial network and the inhibition of copepod production are assumed to decrease the trophic efficiency to higher levels (Daro et al., 2006; Lancelot et al., 2009). Deprived from grazing control *P. globosa* colonies grow rapidly and form transient biomass accumulation, which disruption causes large foam accumulation in coastal zones with a risk of bottom oxygen depletion in some enclosed areas (Peperzak and Poelman, 2008).

2.2. Overview of the models

The main characteristics of the different models are summarised in Table 1 and more details on each model is given below. The pyNuts-Riverstrahler model was applied on a selection of simulated daily discharges as well as 10-day river temperatures, and provided biogeochemical concentrations (NO_3^-, NH_4^+, PO_4^{3-}, DSi, Norg, Porg). The three marine ecosystem models used the outputs of the pyNuts-Riverstrahler at the river outlets (without considering estuarine retentions) to force their river loads in all simulations (see below Section 2.4 Past and present situations, prospective scenarios). Dissolved nutrient state variables were directly imported into the marine models. Regarding organic matter, the non-refractory organic N and P coming from the rivers was converted into organic N and P in marine models (the refractory part of organic matter coming from rivers was either neglected or converted into refractory organic matter depending on the marine model). Phytoplankton from pyNuts-Riverstrahler (in unit carbon) was assumed to die in salty waters and was therefore converted into labile organic N and P within the marine models using the Redfield ratio. The atmospheric deposition of N (wet and dry, oxidized and reduced) computed by the "European Monitoring and Evaluation Program (EMEP)" was used in the three marine ecosystem models (http://www.emep.int/, courtesy of S. Valiyaveetil and J. Bartnicki, met. no). The data has been re-interpolated to be implemented on the grids of the different marine models. Atmospheric deposition of N is used in every scenario except in the Pristine situation (see below Section 2.4.2).

We ran the models ECO-MARS3D and MIRO&CO for ten consecutive years to remove interannual variability. The model PCOMS could only be run for the year 2010 due to limited time with the conjunction of model development and computation time.

The three marine ecological models were meant to run independently from each other to test the effects of the same nutrient-reduction scenarios in different geographical areas. For clarity in mapping the

Table 1
Main characteristics of the river model pyNuts-Riverstrahler and of the three marine ecosystem models PCOMS, ECO-MARS3D and MIRO&CO.

	Generic river model	Marine ecosystem models		
	pyNuts-Riverstrahler	PCOMS	ECO-MARS3D	MIRO&CO
Simulation period	2000–2010	2010	2000–2010	2000–2010
Latitude range	NA	34.4°N-45.0°N	43.2°N-52.8°N	48.5°N-52.5°N
Longitude range	NA	12.6°W-5.1°W	8.1°W-5.0°E	4.0°W-5.0°E
Horizontal resolution	1 km (rivers)	5.6×5.6 km^2	4×4 km^2	5×5 km^2
Depth description	NA	50 sigma layers	30 sigma layers	5 sigma layers
Nutrients	N, P, Si	N, P	N, P, Si	N, P, Si
Dissolved oxygen	Yes	Yes	Yes	No
Number of river outlets	NA	47	130	15
Atmospheric forcings	NA	MM5 forecast model	Meteo-France (Arpege)	UK Met Office
Atmospheric N deposition	NA	EMEP (monthly)	EMEP (monthly)	EMEP (monthly)
Phytoplankton	Diatoms, Chlorophyceae, Cyanobacteria	Generic species	Diatoms, dinoflagellates, nanoflagellates, *Phaeocystis globosa*	Diatoms, nanoflagellates, *Phaeocystis globosa*
Zooplankton	Microcrusteaceans, Rotifers, Ciliates	Generic species	Microzooplankton, mesozooplankton	Microzooplankton, copepods
References	Garnier and Billen, 1993, Billen et al., 1994, Garnier et al., 1995, Ruelland et al., 2007, Thieu et al., 2009	Mateus et al., 2012, Campuzano, 2018	Lazure and Dumas, 2008	Lancelot et al., 2005, Lacroix et al., 2007b, Dulière et al., 2017

results, all model data have been projected onto a larger grid encompassing the three model domains. There is a complete overlap between ECO-MARS3D and MIRO&CO in the English Channel and the Southern Bight of the North Sea. After comparison of model performances (not shown), a decision was made to keep ECO-MARS3D results in the English Channel, which includes the Bay of Seine and the Bay of Somme, and MIRO&CO results in the Southern Bight of the North Sea. The reason is that these areas respectively correspond to where each model has been historically developed, calibrated and validated (see references on the validation in Sections 2.2.3 and 2.2.4). To smooth the final image, a gradient was drawn along a few cells between ECO-MARS3D and MIRO&CO at the junction between the English Channel and the North Sea. Another gradient was drawn between ECO-MARS3D and PCOMS in the Atlantic between the Bay of Biscay and the Portuguese Shelf.

The variables studied in the marine ecological models are the winter (Jan-Feb) nutrients, i.e., Dissolved Inorganic Nitrogen (DIN) and Dissolved Inorganic Phosphorus (DIP) and the winter DIN:DIP ratio. The Chl P90 is calculated over the vegetative period (Mar-Oct).

2.2.1. PyNuts-Riverstrahler

The newly developed biogeochemical pyNuts-Riverstrahler model (Desmit et al., 2015b, Thieu et al. unpublished, www.fire.upmc.fr/rive) rises the challenge of a large scale (1 million km^2) modelling approach, based on a mechanistic representation of microbial-driven processes occurring within the water column and involved in the transfer and retention of nutrients in the entire drainage network. The pyNuts-Riverstrahler model considers the physiology of algae (three functional groups, one diatoms and two non-diatoms) and zooplankton (two functional groups) for the calculation of their biomass (Billen et al., 1994; Garnier et al., 1995). The model simulates the biogeochemical cycles of carbon, oxygen, nitrogen, phosphorus and silica in the drainage network, taking also explicitly into account heterotrophic (including denitrifying) and nitrifying bacterial communities (see Garnier et al., 2002). Most of the important processes in the transformation, elimination, and/or immobilization of nutrients during their transfer in-streams are explicitly calculated at the seasonal scale (at a ten-day resolution). The Riverstrahler model considers the invariance of physiology for organisms along the river network from up to downstream (Garnier et al., 2002), assuming that the singular ecological functioning of aquatic system depends only on the hydro-morphological and anthropogenic constraints to which streams are subjected. The Riverstrahler model thus considers that an accurate description of natural and anthropogenic constraints is the key to understand the biogeochemical functioning of any hydrosystems, raising the issue of harmonising the necessary inputs information available under various formats with different spatial and temporal resolutions across the NEA domain. Additionally, a process-based modelling of aquatic processes operating at large scale also required appropriate solution to address the problem of calculation time and data management. The pyNuts modelling environment is a python framework (with the "Nuts" suffix standing for NUTrientS) designed specifically for such large-scale challenges. It aims at setting-up the Riverstrahler model simulations from individual watersheds to a multitude of small and large watershed covering a regional area of several thousands to millions of km^2, pre-processing all the model inputs and ensuring its propagation following a regular scheme of confluence between all modelling units from up to downstream. Here, 174 river basins were modelled using the pyNuts-Riverstrahler model, and among them 17 major river basins were subdivided and specifically analysed. That selection of river watersheds picks up all important nutrient inputs that enter the marine domain, i.e., the domain covered by the marine models in this study. Fig. 2 provides the comparison of nutrient fluxes calculated and observed at the outlets of the main rivers basins. More details and validation of the pyNuts-Riverstrahler modelling environment are provided in the final report of the EMoSEM project (www.fire.upmc.fr/rive; Desmit et al., 2015b). Nevertheless, in view of the generic nature of the pyNuts-Riverstrahler model, which is not calibrated on any particular basin, the agreement is rather satisfactory.

2.2.2. PCOMS

The Portuguese Coast Operational Modelling System (hereafter referred to as PCOMS; Mateus et al., 2012, Campuzano, 2018) model application consists in two nested domains West Iberia (2D) and Portugal (3D) that run the Mohid Water Modelling System (Neves, 2013; http://www.mohid.com) covering the Iberian Atlantic coast and its contiguous ocean. The PCOMS is 3D full baroclinic hydrodynamic and ecological regional application with a horizontal resolution of 5.6 km and with 50 vertical levels with a resolution of down to 1 m near the surface. The application scales down the Mercator-Océan PSY2V4 North Atlantic solution (Drillet et al., 2005) that provides daily values for water levels and 3D values for horizontal currents, temperature and salinity. Tides are included in the PCOMS application by forcing the West Iberia domain open ocean boundary with tidal components obtained from the FES2004 global tide solution (Lyard et al., 2006). A MM5 model application (Meteorological Model 5; Grell et al., 1994) for Western Iberia implemented by the IST meteorological group (http://meteo.tecnico.ulisboa.pt) is used to provide atmospheric boundary conditions. On the ocean side, the Portugal domain was initialised and forced at their boundary conditions by monthly climatological profiles of oxygen, nitrate and phosphate obtained at the coordinates Longitude: 12.5°W and 38.5°N from the World Ocean Atlas 2009 (Garcia et al., 2010a, 2010b). To simulate the primary production in the coastal area of Western Iberia, the Water Quality module coupled in the MOHID model has been configured. This module is basically a

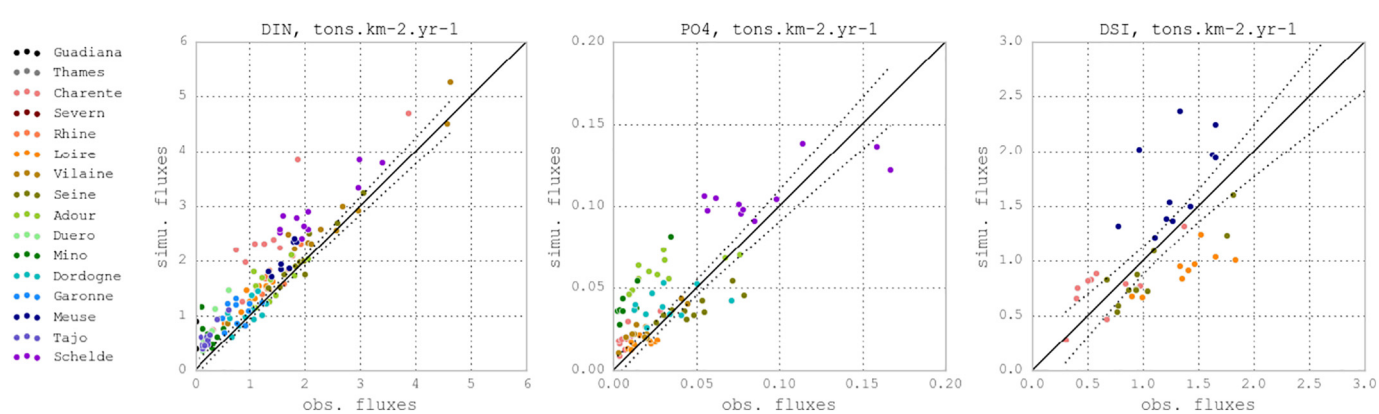

Fig. 2. Comparison between the pyNuts-Riverstrahler model simulations and observed yearly nutrients fluxes calculated from series of observed daily discharge and available concentration data at the outlet of the main river systems during the time frame 2000–2010. Dotted lines indicate the 90% confidence interval. The observed yearly nutrient fluxes are defined as the product of the annual discharge with the discharge weighted mean concentration, as recommended by Moatar and Meybeck (2007).

nutrient- phytoplankton-zooplankton (NPZ) model adapted from a model initially developed at USEPA (U.S. Environmental Protection Agency) (Bowie et al., 1985). The model simulates inorganic and organic forms of N and P as nutrients in its dissolved and particulate forms. It also calculates oxygen, phytoplankton and zooplankton concentrations. In this application the phytoplankton growth has not been limited by temperature considering the maximum optimum interval for phytoplankton growth between 10 and 25 °C.

2.2.3. ECO-MARS3D

The French ECO-MARS3D model is based on the IFREMER's MARS3D hydrodynamical code (Lazure and Dumas, 2008). The current application to the French Atlantic shelf uses a regular grid with 4 × 4 km meshes and 30 sigma levels covering the Bay of Biscay, the English Channel and the Southern Bight of the North Sea, up to the Rhine estuary (Fig. 1). To improve the oceanic currents and the biogeochemical fluxes at the western boundary of this domain, and to avoid local artificial enrichment through zero derivative boundary conditions, this grid has been embedded in a wider one, extending from 41°N to 55°N and 18°W to 9°30′E, with also 4 × 4 km meshes and 30 sigma levels. This large domain has been used once for computing at the boundaries of the working domain the instantaneous hydrodynamic forcing (sea surface elevation and 3D currents) during the 2000–2010 decade, and a climatological year (average over the decade) for biogeochemical fluxes. Wind, atmospheric pressure and thermal fluxes at the sea surface are provided by the Arpege model of Météo-France with a 30 km and 6 h space-time resolution. Suspended particulate matter is set to the maximum of the ambient climatological fortnightly mean distribution derived from satellite data (Gohin, 2011), and the suspended matter brought by the rivers, which is simulated as a particulate conservative tracer with uniform and constant settling velocity. The basic biogeochemical model contains 17 state variables, describing the N (with nitrate and ammonium separately), P and silicon cycles and the dissolved oxygen in the pelagic ecosystem. Phytoplankton is divided into three groups: diatoms, dinoflagellates and nanoflagellates, with biomass expressed in nitrogen concentration in the equations (instead of carbon). To assess specific consequences of eutrophication, a module simulating the single-cells, colonial forms and mucus of the haptophyte *Phaeocystis globosa* (inspired from the MIRO&CO model) has been added in competition with the bulk phytoplanktonic variables. Total Chl is deduced from the nitrogenous state variables of the model by an empirical Chl:N ratio. There are two zooplanktonic components, with biomass expressed in N concentration: the microzooplankton, which feeds on nanoflagellates, dinoflagellates and detrital particulate matter everywhere, along with diatoms in oceanic regions (depth > 200 m), and the mesozooplankton, which grazes diatoms, dinoflagellates and microzooplankton. In this model, diatoms do sink, whereas nanoflagellates and dinoflagellates do not. Three particulate detrital variables (detrital N, detrital P, detrital Si) close the biogeochemical cycles, and settle in the water column to the sediment layer. A detailed description and validation of this ECO-MARS3D model is provided in Ménesguen et al. (in rev.). Initial conditions for Pristine and Reference (current) simulations were obtained with a two-years spin up run of 1999 conditions. For the scenarios (see further), initial conditions were obtained after a 2-years spin up run of Pristine scenario initial conditions.

2.2.4. MIRO&CO

MIRO&CO results from the coupling of the 3D hydrodynamic COHERENS model (Luyten, 2011) with the biogeochemical MIRO model (Lancelot et al., 2005). COHERENS is a three-dimensional numerical model, designed for application in coastal and shelf seas, estuaries, lakes, reservoirs. MIRO is a biogeochemical model that has been designed for *Phaeocystis*-dominated ecosystems. It describes the dynamics of phytoplankton (three functional groups), zooplankton (two functional groups), bacteria, organic matter degradation (dissolved and particulate) and nutrient cycles (N, P, Si) in the water column and the sediment. The current setup has been obtained by coupling MIRO with COHERENS v2 (MIRO&CO v2): details and validation are shown in Dulière et al. (2017). The MIRO&CO domain has a horizontal grid resolution of about 5 km × 5 km and 5 vertical sigma coordinate layers. The surface wind and atmospheric pressure fields from the UK Meteorological Office have been used to force the model in addition to 6-hourly precipitation rate, cloud cover, specific humidity and air temperature fields. The sea surface temperature was derived from the weekly sea surface gridded temperature (20 × 20 km^2) obtained from the BSH (Bundesamt fuer Seeschiffahrt und Hydrographie) (Loewe, 2003). A 2D implementation of COHERENS v2 over the North Sea continental shelf provides the water currents at the open sea boundaries. At the English Channel open sea boundary (4°W), nutrient concentrations for the Reference conditions were derived from climatological databases of the European Union NOWESP and ERSEM projects (Radach and Lenhart, 1995). Phytoplankton concentrations were imposed from SOMLIT data (station Roscoff ASTAN; somlit.epoc.u-bordeaux1.fr/fr). For the Pristine conditions and the prospective scenarios, the western ocean boundary conditions were adapted to include the nutrient fluxes generated by ECO-MARS3D. At the northern boundary of the Southern North Sea (52.5°N), nutrient concentrations from NOWESP, ERSEM and ICES combined data were imposed in the west part, while in the east part of this boundary a zero-horizontal cross-boundary gradient of nutrients has been implemented (i.e., the nutrient gradient between the two sides of the boundary is equal to zero – in practice, we imposed that the nutrient concentration that enters the domain is equal to the one inside the domain). The DINEOF methodology (Data Interpolating Empirical Orthogonal Functions) has been applied to the MODIS-Aqua images of Total Suspended Matter (TSM) to reconstruct a daily climatology of TSM (Sirjacobs et al., 2011). Initial conditions were obtained with a 6-years spin up run of 1999 conditions.

2.3. Coastal domain definition and spatial distribution of coastal values

An asset of the present study is that the spatial distribution of winter nutrients and Chl P90 can be estimated with combined marine models across the NEA, for instance along the coastal domain (versus the offshore). Such a coastal domain across the NEA should be somehow defined. Previous works of the OSPAR Commission have defined the coastal areas as a function of salinity because in some eutrophicated coastal zones of the North Sea there is a linear relationship between salinity and marine nutrient concentrations (Claussen et al., 2009). However, this relationship is not the same everywhere and is less convenient in upwelling areas like the Portuguese coast. In this study, it was chosen to define the coastal area in direct relation to the eutrophication indicator Chl P90 using the satellite observations of Fig. 1. The long-term average (2000–2010) of remotely sensed (RS) Chl P90 delivers an accurate view of the eutrophicated areas showing a systematic coastal-offshore gradient in Chl P90 (Fig. 1). The statistical distribution of RS Chl P90 values was used to design the limits of the coastal domain. The cumulative sum of the distribution (not shown) displayed a curve with three inflexion points at values 2 μg Chl L^{-1}, 4 μg Chl L^{-1} and 8 μg Chl L^{-1} (also drawn as isolines in Fig. 1). The coastal domain across the NEA was chosen as the sub-area where RS Chl P90 is higher than 4 μg Chl L^{-1}. For each modelled variable, a probability distribution function (PDF) across the coastal domain (spatial distribution) has been calculated assuming a "generalised extreme value" distribution model. The PDFs summarise the spatial distribution of eutrophication indicators across the NEA coastal zone and the effect of each scenario thereof.

2.4. Past and present situations, prospective scenarios

As a first step in the process of reducing coastal eutrophication, the assessment of the current status of the NEA ecosystems was made in comparison with their pristine status. This is a request of European

Table 2
Indicative changes of nutrient levels in a past situation (Pristine) and in the prospective scenarios with respect to the current situation (namely Reference). The hypotheses taken for each prospective scenario (application of the Urban Waste Water Treatment Directive, UWWTD, use of good agricultural practices, GAP, and deep changes in the agro-food system, LocOrgDem) are split in terms of population, wastewater treatment and agriculture.

	No human impact	Current situation	Prospective scenarios		
	"Pristine"	"Reference"	"UWWTD"	"GAP"	"LocOrgDem"
Population	None	Current	Current	Current	Current
Waste water	–	Current	UWWTD	UWWTD	UWWTD
Agriculture	–	Current	Current	GAP	LocOrgDem

policies, such as the WFD, in which the status of the system (high, good, moderate, poor) is defined with respect to a system-specific 'background' situation that can reflect a pre-industrial situation (Serna et al., 2010) or a genuinely pristine situation (this study). There is, however, limited knowledge of the pristine status of coastal ecosystems across Europe. In the present study, the pristine situation in the riverine and coastal systems results from a model in which Western Europe is covered with primitive forests and is not subject to any human influence. The assessment of both the current and the pristine status of eutrophication across the NEA allows scaling the effects of human activities on the riverine and marine ecosystems. The prospective scenarios, in turn, describe some steps of the future effort in reducing the cause and effects of the current eutrophication. They either simulate specific demands or advices in existing policies through common water and crop management ("UWWTD" and Good Agricultural practices, "GAP") or they go further and propose integrated structural changes (Local Organic Demitarian, "LocOrgDem") (see Table 2 and below for hypotheses and details).

The prospective scenarios (Table 2) propose nutrient reductions that are not simply drastic a priori percentage abatements in the inputs but are instead quantified reductions based on the capacity of human society to change, more or less deeply, their practices. The scenarios are built on proposed changes in land use and wastewater management, which results in reducing river nutrient inputs to the sea with different impacts to the marine ecosystem. Three levers can be moved for defining possible scenarios: (i) The first one is related to the point sources of nutrient through urban waste water (hereafter UWWTD); (ii) in addition to the UWWTD scenario the second lever involves improving farming practices in the scope of the current agricultural systems (hereafter GAP); (iii) in addition to UWWTD the third lever implies a profound reshaping of the structure of the agro-food system: hereafter called LocOrgDem (the demitarian diet is a shift in diet towards more vegetal and less animal protein consumption, http://www.nine-esf.org/node/281/index.html, organic farming replace dominant conventional one, local means abandoning territorial specialisation of agriculture reconnecting crops and animals, and food and people). These three levers are ordered following their degree of feasibility: the first involves the ongoing European legislation on point sources supported by the Urban Waste Water Treatment Directive, the second corresponds to the more recent definition of Good Agricultural Practices (see the Nitrate Directive), and finally the third one implies in-depth changes in agricultural landscape, trade exchanges and human diet and follows a number of current "weak signals", e.g., an increasing demand for local and healthy food.

2.4.1. Current situation, hereafter called "Reference"

The Reference situation has been built using the most recent level of pressures reported by national and European databases and was thus considered as a starting point for scenario building. It basically mixes information recently published, describing both natural and anthropogenic pressures (Table 3). The "Reference" is thus a composite state assuming a permanent validity of the different constraints applied to the hydrosystems. Only hydrological constraints are dynamically described over the period 2000–2010 to better simulate the year-to-year variability of riverine fluxes.

2.4.2. Pristine situation: scenario "Pristine"

To estimate what could be the functioning of coastal marine systems in the absence of any anthropogenic nutrient inputs, a hypothetic "pristine" scenario was established to calculate the corresponding nutrient loadings from all rivers of the North-East Atlantic watershed. Pristine conditions were defined as natural land cover from the dominant climax vegetal formations in the different climatic and lithological zones of the NEA watershed under current climate conditions, without hydrological management. Based on the climatic stratification of Europe proposed by Metzger et al. (2005), and on the delimitation of European Ecoregions by Wasson et al. (2010), we classified each individual watershed as either Mediterranean or temperate on the one hand, and crystalline or sedimentary on the other hand. The pristine level of nutrient

Table 3
Construction of the "Reference" situation: data sources and time periods for the different inputs required by the pyNuts-Riverstrahler model.

Model inputs	Data sources and references	Timeframe
Morphology		
Hydrographic network	River catchment characteristics (CCM2 version 2.1; Vogt et al., 2007)	2007 and later
Point sources		
Loads and treatments type	Waterbase UWWTD (EEA 2012)	Reporting period 2009
Per capita emissions	National census and other published values (Desmit et al., 2015b)	–
Diffuse sources		
Landuse	Corine Land Cover 2006 (Bossard et al., 2000; EEA, 2007)	2006
Agricultural statistics	Eurostat statistics by NUTS3 regions (Eurostat 2010)	2010
Soil erosion	European Soil Data Center: PESERA map (Kirkby et al., 2004)	2002
Soil content	Land use/land cover Frame Survey LUCAS (Toth et al., 2013)	2001–2009
Litho-morphology	Hydro-Ecoregions (Wasson et al., 2010)	2007
Hydrology		
National monitoring networks	United Kingdoms: Centre of Ecology and Hydrology Ireland: Hydronet EPA; Office of Public work Netherlands: Waterbase of the Ministry of infrastructure and Environment Germany: Federal Institute of Hydrology and the Global Runoff Data Centre Belgium: Hydronet, Aqualim and Voies hydrauliques France: Banque Hydro Switzerland: Federal Office for the Environment Spain: Centro de Estudios y Experimentacion de Obras Publicas Portugal: SNIRH and INAG	2000–2010

inputs to river systems corresponds to background concentration in leaching and runoff water from unperturbed forested areas, plus the input of litter from riparian trees. The former was evaluated based on a compilation of literature data for N, P and Si concentrations in pristine forested rivers (Kirchner and Dillon, 1975, Likens and Bormann, 1975, Meybeck, 1982, Meybeck, 1986, Lewis, 2002, Perakis and Hedin, 2002, Garnier et al., 2005, Némery et al., 2005, Sferratore et al., 2006), considering climate and lithology of the watershed. The input of litter fall from riparian vegetation to rivers was estimated based on a review of the literature on the subject (King et al., 1987; Chauvet and Jean-Louis, 1988; Stewart and Davies, 1990; Benfield, 1997; Pozo et al., 1997; Sabater et al., 2001; Bernal et al., 2003; Acuna et al., 2007; Gonzalez, 2012). This review revealed no significant differences between Mediterranean and temperate riparian systems, but showed a strong stream order dependence of the annual litter input per km river stretch, with average values increasing from 400 kg C km^{-1} yr^{-1} for 1st order rivers to 2500–3000 kg C km^{-1} yr^{-1} at 4th order, then levelling off. The elemental composition in terms of C:N:P of the material constituting litter inputs from the most frequently occurring natural species (*Alnus* sp., *Salix* sp., *Corylus* sp., …) is about 500:17:1 (by weight).

2.4.3. Improvement of urban waste water treatment according to EU directive: scenario "UWWTD"

Mitigation of point sources could be an efficient way to reduce the amount of nutrients transferred to the aquatic system, especially for P. Despite their recent improvement, conurbations and associated waste water treatment plants of the NEA domain do not still fully comply with European requirements (UWWT Directive) concerning the adverse effects of urban waste water discharge. The UWWTD database (Directive 91/271/EEC) has been used to characterise point sources emission within the NEA domain. The 2009 data-reporting was used to reconstruct the waste water loads generated, collected and finally treated (or not) in waste water treatment plants. Per-capita emissions based on national statistics have been estimated for each treatment type (namely primary, secondary and more stringent treatment). Overall, it represents 4636 points discharging >200 million inhabitants, and among them about 25 million are located in small coastal watersheds which are not taken into account in the pyNuts-Riverstrahler modelling approach, thus are considered directly discharging into the coastal sea. Mapping of the NEA sanitation capacity indicates a higher density of waste water treatment plants in its northern part, while a more scattered sanitation capacity is observed in the Iberian Peninsula. In addition, the Southern part of the NEA domain does not comply with the requirement of the UWWT Directive yet. Indeed, each waste water discharge is explicitly connected to a specific receiving area which "sensitivity" will determine the type of waste water treatment that should be put in place. The UWWTD scenario has been built on this basis and assumes a full implementation of the UWWT Directive, considering that all NEA catchments were classified either as "Normal" or "Sensitive" areas which implies according to the directive:

- The collection and treatment of waste water in all conurbations of >2000 population equivalents (p.e.)
- Secondary treatment of all discharges from conurbations of >2000 p.e.
- More advanced treatment with 70% N and 90% P reduction for conurbations >10,000 population equivalents in designated areas and their catchments.

This scenario also includes an additional reduction of the per capita human emissions of P, considering a complete ban of P in both laundry powders and dishwasher detergents, in line with the Commission proposal (COM) 2010-597 amending Regulation 648/2004/EEC on detergent. Despite that the recent revision of the Regulation (EC) No 648/2004 only sets limits on the content of phosphates and others phosphorus compounds in detergents, the use of P-free detergent in Europe is here assumed as a plausible option in the near future.

2.4.4. Agricultural scenarios

2.4.4.1. Reasoned and good agricultural practices: scenario "GAP". For more realism in the timeframe of implementation, the GAP scenario is considered as combined with concomitant effort in the mitigation of nutrient point sources following the narrative of the UWWTD scenario. At least in most vulnerable areas, considerable efforts have been devoted in the last decades to adapt the level of fertilisation to the needs of crop growth. The current situation described in the reference scenario by the N soil balance of the different agricultural regions of the NEA thus probably reflects the lower limit of the level of N soil balance (defined as the difference between total N inputs to the soil and export of N with the harvest) which can be achieved without major change in the current cropping systems and yield objectives. The main lever that could lead to a further reduction of N leaching consists of the introduction of catch crops preventing bare soils during the autumn and winter period of intense drainage before spring crops. Such a measure has indeed been demonstrated to significantly reduce the fraction of N soil balance of croplands being leached during the drainage period, on the condition that the presence of catch crop is taken into account for the adjustment of the succeeding crop fertilisation (Justes et al., 2012). We estimated the N soil balance reduction made possible in each region by the application of this measure, based on the frequency of spring crops in the dominant rotations in use in each region: we considered a flat rate abatement of 10 kg N/ha/yr for fertilisation of spring crops and used the leaching reduction coefficients proposed by Anglade (2015) and Anglade et al. (2017). As a mean over all regions of the Western European domain, the N leaching reduction which can be expected from this measure of generalised catch crop implementation is <30%.

Regarding diffuse phosphorus loads, the effect of these agricultural practices, mainly addressing N issues, is difficult to assess. While winter cover is likely to protect otherwise bare soils from erosion, hence to reduce the associated phosphorus inputs to the river network, the uncertainty on the magnitude of this effect is large, and we preferred neglecting it for a conservative estimation of the nutrient loading in the scenario. Moreover, soil P balance in European agriculture has been severely reduced during the last decades, up to the point of being often negative, without affecting either agricultural yield nor P losses because of the high pools of soil P accumulated during the 1960–1980 period (Bouwman et al., 2006; Garnier et al., 2015; Bouwman et al., 2017; Le Noë et al., 2017). The P reduction of the GAP scenario with respect to the Reference one is therefore considered the same as that of the UWWTD scenario (Table 2). Because of the important proportion of point sources in the P budget of river systems, this conservative assumption is acceptable.

2.4.4.2. Structural change in the agro-food system: scenario "LocOrgDem". N losses to hydrosystems can be viewed as the direct consequence of the specialisation and disconnection of territorial agro-food systems resulting in the opening of nutrient cycles (Lassaletta et al., 2014; Le Noë et al., 2017). Reconnecting crop production and livestock farming, as well as agriculture and local human food consumption is a major option for reducing N contamination of water resources and fluxes to the sea (Thieu et al., 2011, Billen et al., 2012, Billen et al., 2015, Garnier et al., 2016, Billen et al., 2018, in press). The construction of this scenario, called Local-Organic-Demitarian, for the Western European domain involved the following steps:

- A healthy human diet, the 'demitarian diet', in line with the Barsac declaration (http://www.nine-esf.org/Barsac-Declaration) is considered, with a shift towards more vegetal and less animal protein ingestion (inverting the current 35% vegetal and 65% animal protein proportions) and a reduction of wastage. In the Mediterranean countries this option is well represented by the so-called Mediterranean diet.

- Livestock Units (LU) in each agricultural region are adjusted to the possibilities of feeding it locally on grass and forage crops, without feed import from outside the region, also considering the local human requirements of animal proteins. Europe livestock systems today are very dependent on feed imports (Leip et al., 2015). Because of the strong reduction of meat and milk consumption in this scenario, livestock density and production are decreased by about half as an average over the domain, but are redistributed much more evenly among the regions (Fig. 3), in line with the objective of the scenario to re-connect crop and animal farming in over-specialised regions (Garnier et al., 2016; Anglade et al., 2017).
- Organic cropping systems with long and diversified rotations involving legumes are established everywhere, keeping unchanged the total agricultural area. N inputs to the soil are controlled by the local availability of manure and symbiotic N fixation, with no application of synthetic fertilisers. The resulting N yield in each region is calculated assuming the same yield vs. fertilisation relationship as the one currently observed (Anglade et al., 2015). The scenario implies a significant decrease of crop production, by 30 and 40% in France and Spain respectively.
- The analysis of the fluxes of vegetal and animal protein produced in this scenario (Billen et al., 2018, in press) shows that, despite its considerable de-intensification, the food production remains able to meet the 'demitarian' requirements of the local population, with much less long-distance transport. While no more export of meat and milk are possible, surplus production of vegetal proteins remains significant (750 kt N/yr in France, 70 kt N/yr in Spain) and is available for either export to international market, or for energetic use, e.g. through methane generation.

The N soil balance of arable land is largely reduced in this scenario with respect to the reference situation (Fig. 4), leading to a strong reduction of diffuse N load to the hydrosystems.

Again, regarding phosphorus, the effect on diffuse P loading is difficult to assess and has been neglected. The total P loading of the LocOrgDem scenario is therefore conservatively considered identical to that of the UWWTD and GAP scenario.

3. Results

3.1. Pristine versus reference situations

Nutrient concentrations along the land-ocean continuum of the NEA show clear differences between the Pristine and the Reference situations (Fig. 5). Within the drainage network of the NEA, the calculated mean levels of nitrate concentrations under Pristine conditions show everywhere values below 0.45 mg N L^{-1} (32 μmol L^{-1}). This is to be compared to the riverine nitrate concentrations in the Reference (current) situation, where values vary from 0.45 to 12.9 mg N L^{-1} (32 to 921 μmol L^{-1}) in some rivers of the NEA, especially in Brittany, in the north of France and in Belgium, thus showing an increase by a factor 25 between the Pristine and the Reference situations. Regarding orthophosphate concentration in the drainage network, the values in our Pristine situation remain below 0.03 mg P L^{-1} (1 μmol L^{-1}). In the Reference situation, most rivers across the NEA exhibit orthophosphate concentration values around 0.16 mg P L^{-1} (5 μmol L^{-1}), except for some Portuguese rivers and for the Scheldt and Thames estuaries where values may reach 0.65 mg P L^{-1} (21 μmol L^{-1}). That is an increase by a factor ~20 compared to the Pristine situation. In addition to quantifying the size of human-induced eutrophication, the model results obtained from Pristine conditions also provide some estimate of the "natural background value" of each indicator (N, P, Chl; see Discussion), a concept close to the one used in the WFD policy.

Silica drained by continental waters to the coastal zone slightly varies between scenarios as it originates from rock dissolution or weathering (not shown). However dissolved silica (DSi) loads to the sea vary depending on diatom production in the river system. A reduction in P inputs to the drainage network will reduce phytoplankton production and hence the diatom consumption of DSi in the river, making

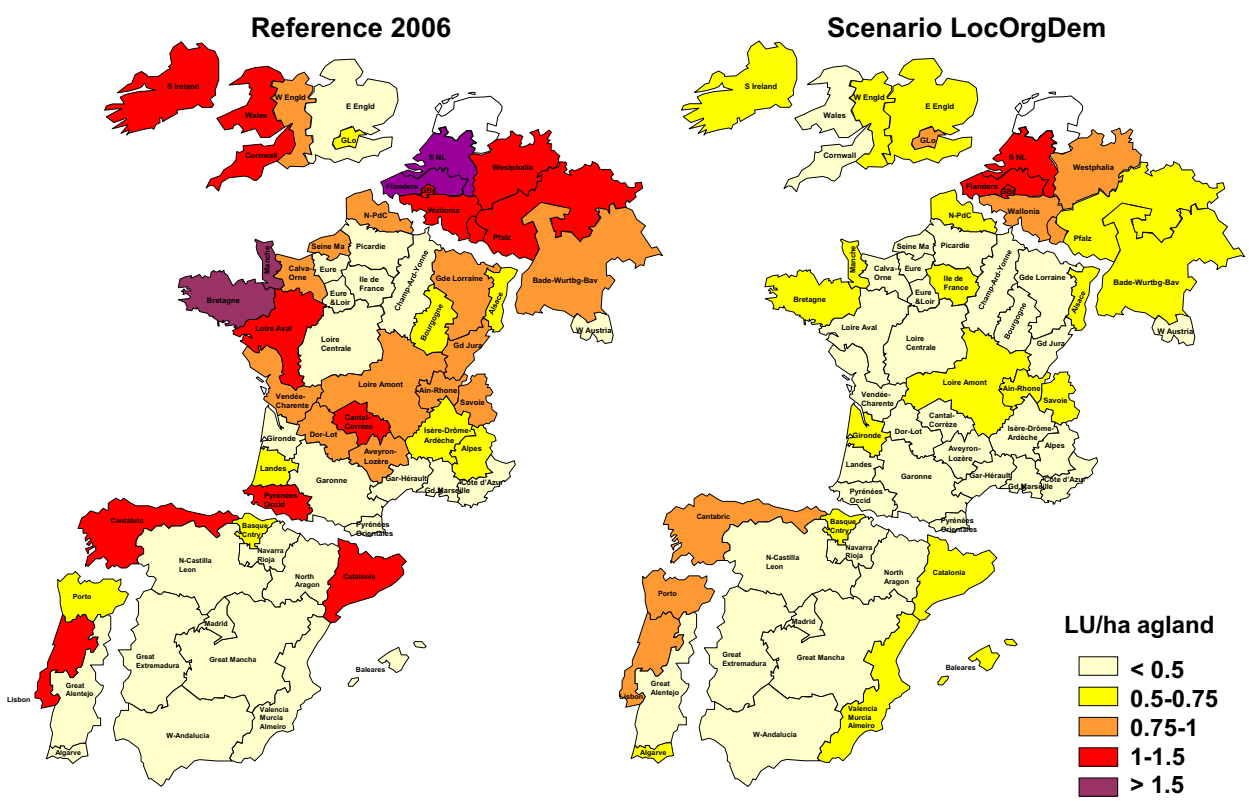

Fig. 3. Distribution of livestock density in the Reference situation and in the LocOrgDem scenario.

more DSi available for coastal diatoms. This increase in DSi coastal delivery due to reduction in P loads has already been observed in the Rhine (Hartmann, 2011), the Seine (Passy et al., 2013) and in the Dutch rivers and the Dutch Continental Shelf (Prins et al., 2012). Such a link between nutrients has already been modelled in the Southern North Sea (Lancelot et al., 2014) and the Bay of Seine (Passy et al., 2016).

In the Pristine situation, the highest nutrient concentrations are found in the coastal zone of France (Bay of Biscay) where the winter DIN concentration may reach 14.7 μmol L^{-1}. The same is observed for winter DIP and the values may reach 0.79 μmol L^{-1}. These concentrations are considered relatively high under Pristine conditions and may probably be linked to an accumulation of nutrients in these areas due to longer residence times (Nixon et al., 1996). At the ocean margin of the Bay of Biscay, the relatively high nutrient concentrations modelled in Pristine conditions are partly the result of modelling the natural phenomenon of deep water ascent at the shelf break and partly due to an artefact linked to numerical diffusion at the margin when sigma layers are elongated. This is especially visible in winter DIP figures. In the Pristine situation, the ocean margin appears as an additional source of winter DIP to the shelf sea while riverine inputs are relatively low. Outside of these areas and especially in the coastal zones, Pristine N and P nutrient concentrations are remarkably low.

The terrestrial export of nutrients to the sea has an impact on phytoplankton dynamics in the coastal zones (Fig. 6). The Pristine situation reflects the natural state of the marine domain characterised by relatively low Chl P90 in most coastal zones, although Chl P90 may reach 7 μg L^{-1} in some French bays (Biscay, Seine and Somme). In comparison, the Reference situation reflects the anthropogenic enrichment in nutrients and exhibits much larger values of Chl P90 along the NEA coasts with maxima around 40 μg L^{-1} and an extension of the eutrophicated domain offshore in the sea. Only the coasts of the Iberian Peninsula feature typical marine values for Chl P90, except at the mouths of the rivers Tagus and Douro. The Pristine situation shows the lowest winter N:P ratios in this study compared to the Reference situation and the other scenarios. Still, high values of winter N:P may be found in some coastal zones, like in France (37 mol N mol P^{-1}). This suggests that the coastal winter N:P ratio in Pristine conditions is not necessarily identical to the oceanic background value of 16 mol N mol P^{-1} as coastal values will depend on riverine non-anthropogenic inputs and local features (line of coast, hydrology, biogeochemical processes, …). The results show a clear N enrichment in all NEA coastal zones (compared to P) under Reference conditions as highlighted by Romero et al. (2013).

As the Pristine conditions in the oceanic end-member (beyond the boundaries of marine models) remain unknown, it was assumed they were the same as the current ones. In the northern part of the domain, i.e., North Sea and Irish Sea, such an assumption neither considers the expected reduction in nutrient loads coming from northern UK rivers outside of the river model domain nor the processes occurring in the shelf seas north from the boundary.

3.2. Prospective scenarios versus Reference situation

When calculating the N fluxes delivered at the coastal zone by the main rivers, one can notice the importance of the contribution of the largest ones (Rhine, Seine, Loire, especially; Fig. 7). However, the sum of the specific fluxes of the small rivers (small watersheds) cannot be neglected. Results show that only the LocOrgDem scenario induces a notable decrease in riverine N fluxes to the coastal zones: −30% for the Portuguese Spanish coast and up to −67% for Loire-Brittany coastal area (Table 4). Regarding the point sources of nutrients, Fig. 7 clearly shows that a reduction of N deliveries cannot be expected from waste water treatments alone (scenario UWWTD) as N is mainly originated from diffuse agricultural sources. The same is true for P already well treated, except for the small watersheds in the southern UK and Ireland. The N-ICEP indicator (not shown, excess N over Si with respect to the

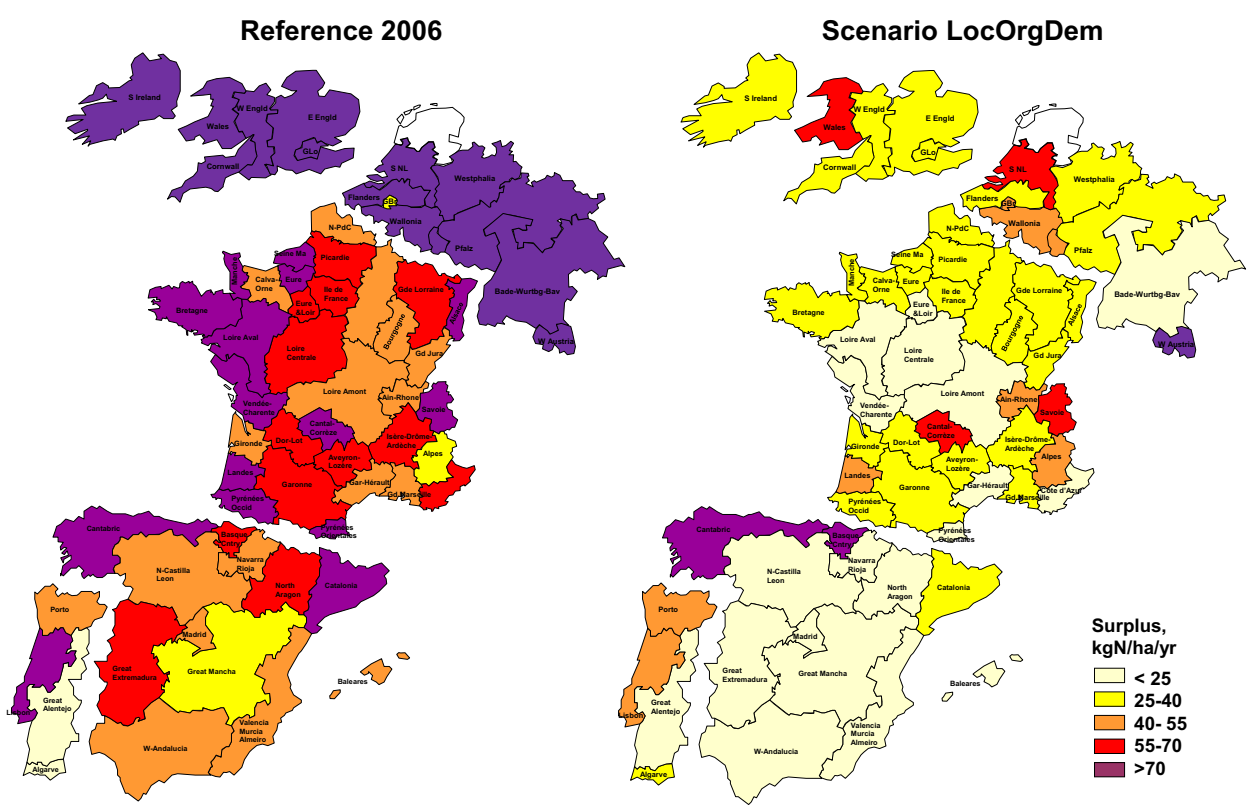

Fig. 4. Distribution of the average N soil balances on arable soils over the model domain in the Reference situation and in the LocOrgDem scenario.

requirements of diatom growth, representing a potential eutrophication when higher than zero, Billen and Garnier, 2007), showed that, whereas the values are logically negative in the Pristine situation (i.e., Si in excess to N), N-ICEP is positive for the Reference situation and largely decreases for the two agricultural scenarios GAP and LocOrgDem. Although still positive, the potential for N eutrophication is reduced from 30 to 90%, approaching the nutrient balance (zero N-ICEP) in the LocOrgDem scenario. Regarding P fluxes, a small decrease is observed in UWWTD compared to the Reference situation in most rivers (especially the large rivers). For P-ICEP (not shown), P being rather well balanced already with respect to Si, an overall decrease by 20% for the UWWTD scenario was estimated compared to the Reference situation.

3.2.1. "UWWTD" scenario

The calculated mean level of nitrate concentration within the drainage network in the UWWTD scenario is shown in Fig. 5. Water quality is slightly improved in specific parts of the drainage network: main stems of the Tagus river basin, of the Guadiana and of the eastern part of the Great Ouse river basin move from medium to good quality as established in EU environmental regulations (see Romero et al., 2016, Table 3). The downstream part of the Seine River (between Paris and Poses) is also improved. For P, the combined effect of improving waste water treatment and reducing P emissions from laundry powders and dishwasher detergents leads to a significant reduction of P concentrations in northern river namely the Rhine, the Meuse, the Scheldt and all British Island rivers and in the Tagus river. The resulting decrease in nutrient export to the coastal zone is limited compared to the Reference situation, except for P in northern countries. Though winter DIN and winter DIP marine concentrations decrease by <5% in most coastal zones, winter DIP decreases by 10% in limited geographical areas at the Loire and the Seine mouths and at the Thames and the Severn mouths (the colour scale does not allow to see it on the map). The

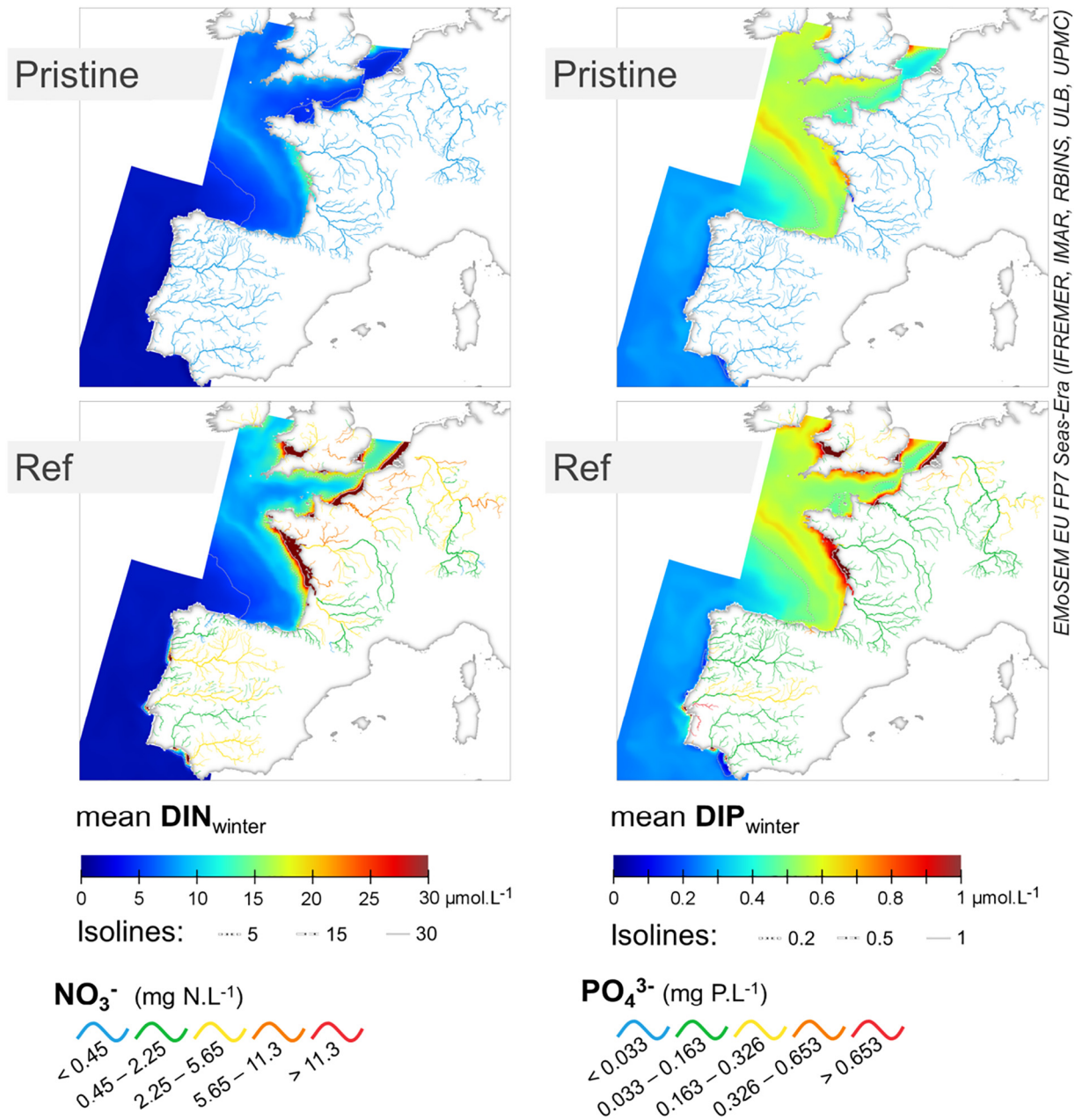

Fig. 5. Model results for nitrogen (left) and phosphorus (right). NEA-integrated contour maps of multiyear-averaged marine winter DIN and DIP (Jan-Feb; µmol L^{-1}). River colours show nitrate and orthophosphate concentrations (mg L^{-1}) across the domain. Each graph represents a scenario. The GAP scenario was not tested with PCOMS and therefore results are not available for this scenario along the Portuguese Continental Shelf. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

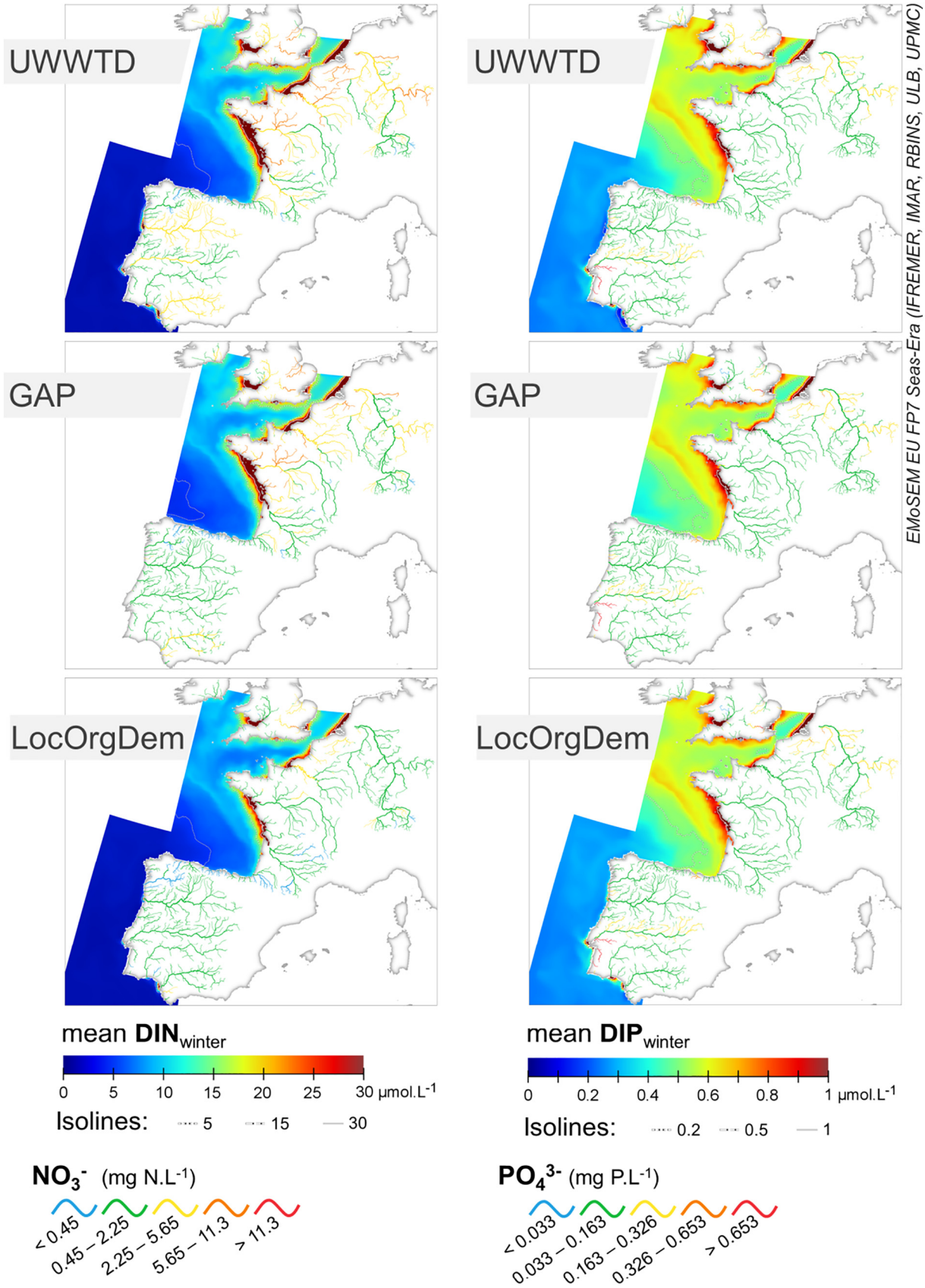

Fig. 5 (*continued*).

overall limited change in nutrient export to the sea indicates that many improvements required by the UWWT Directive have already been achieved in Western Europe. The marine Chl P90 concentration follows a partial decline in coastal areas indicating a slight improvement of coastal water quality (Fig. 6).

3.2.2. "GAP" (+"UWWTD") scenario

The improvement of nitrate concentrations in the drainage network with the GAP scenario is visible, though some rivers keep high nitrate concentrations in their downstream part, especially in the north of France and Belgium (Fig. 5). In the coastal zones, the effects of the GAP scenario on winter DIN compared to the Reference situation are difficult to notice on the map: there is a decrease of winter DIN marine concentrations of 5 to 10% in most coastal zones close to the river outlets (a higher decrease of 20% is observed at the Seine mouth). Regarding winter DIP, marine concentrations in coastal zones remain almost unchanged in the GAP scenario compared to the UWWTD scenario (Fig. 5) because of similar P fluxes to the sea (Fig. 7). The Chl P90 shows a similar decrease in the GAP scenario as in the UWWTD scenario, following the P decrease from waste water treatment operations (Fig. 6). This scenario was not applied to PCOMS marine model in the Portuguese Continental Shelf area.

3.2.3. LocOrgDem (+UWWTD) scenario

The mean nitrate concentrations in the watersheds have been lowered in the LocOrgDem scenario to the point where river waters in most areas return to a status without eutrophication (Fig. 5). In the marine system, the winter DIN is considerably reduced in the LocOrgDem scenario compared to the Reference situation (Fig. 5). The spatial extent of the high winter DIN concentrations is visibly reduced across the NEA. The winter DIP riverine concentrations and inputs to the coastal zones slightly decrease in the LocOrgDem scenario but not in the same proportion as the winter DIN. The decrease in winter DIP is mainly linked to the implementation of the UWWTD (Fig. 7).

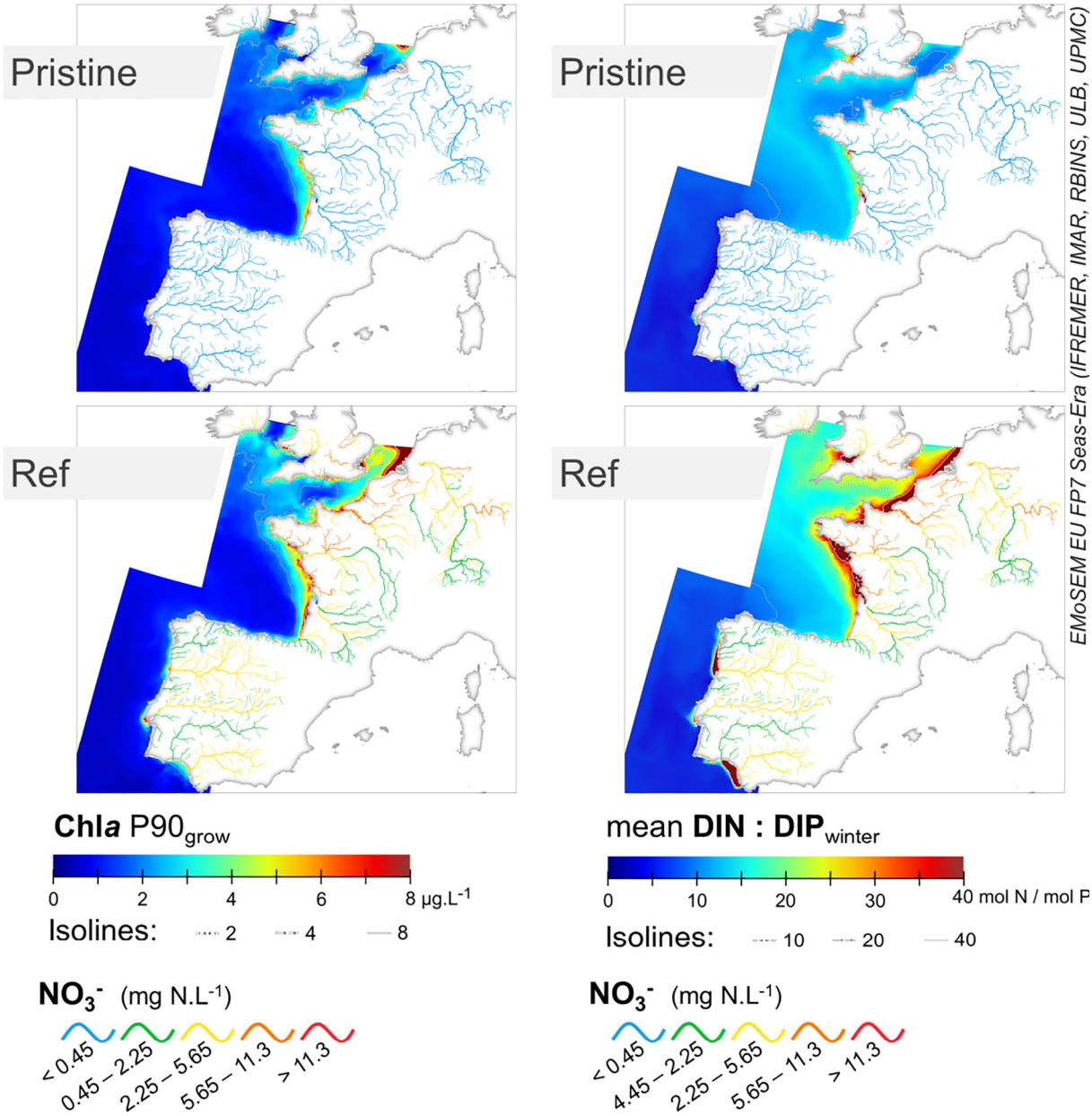

Fig. 6. Model results. Left – NEA-integrated contour maps of multiyear-averaged marine chlorophyll *a* P90 over the growing season (Mar-Oct; µg L^{-1}). River colours show nitrate concentrations across the domain. Each graph represents a scenario. Right – Same for marine winter DIN:DIP (Jan–Feb; mol N mol P^{-1}). The GAP scenario was not tested with PCOMS and therefore results are not available for this scenario along the Portuguese Continental Shelf.

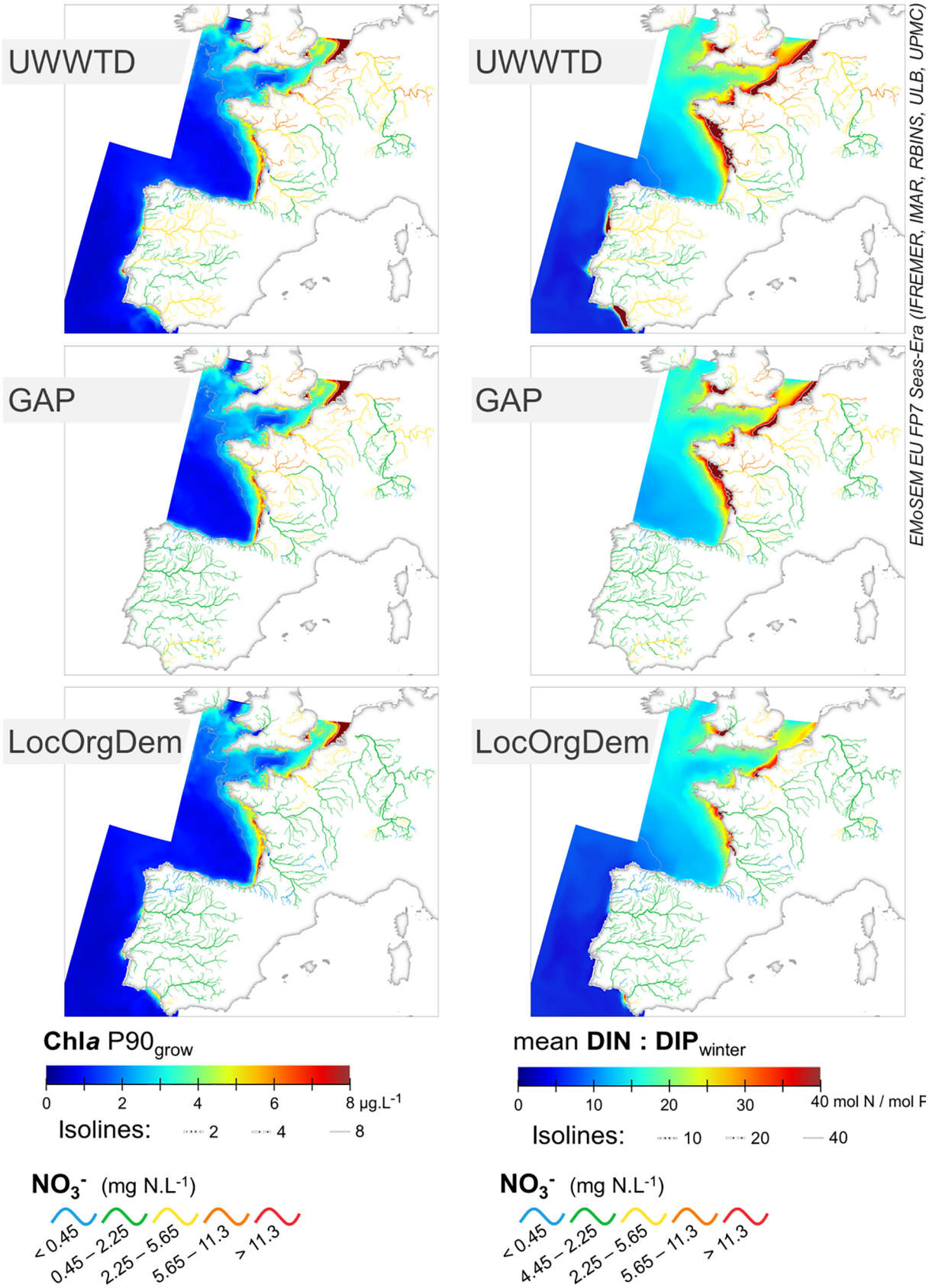

Fig. 6 (*continued*).

Despite a significant N reduction in the coastal zones, the Chl P90 remains relatively high (Fig. 6). The spring peak of phytoplankton is indeed controlled by the P availability in most coastal zones of the NEA (Billen et al., 2011), also including at least the Dutch continental shelf (Troost et al., 2014) and the Wadden Sea (Philippart et al., 2007). Still, the Chl P90 undergoes a visible decrease along the French coasts in

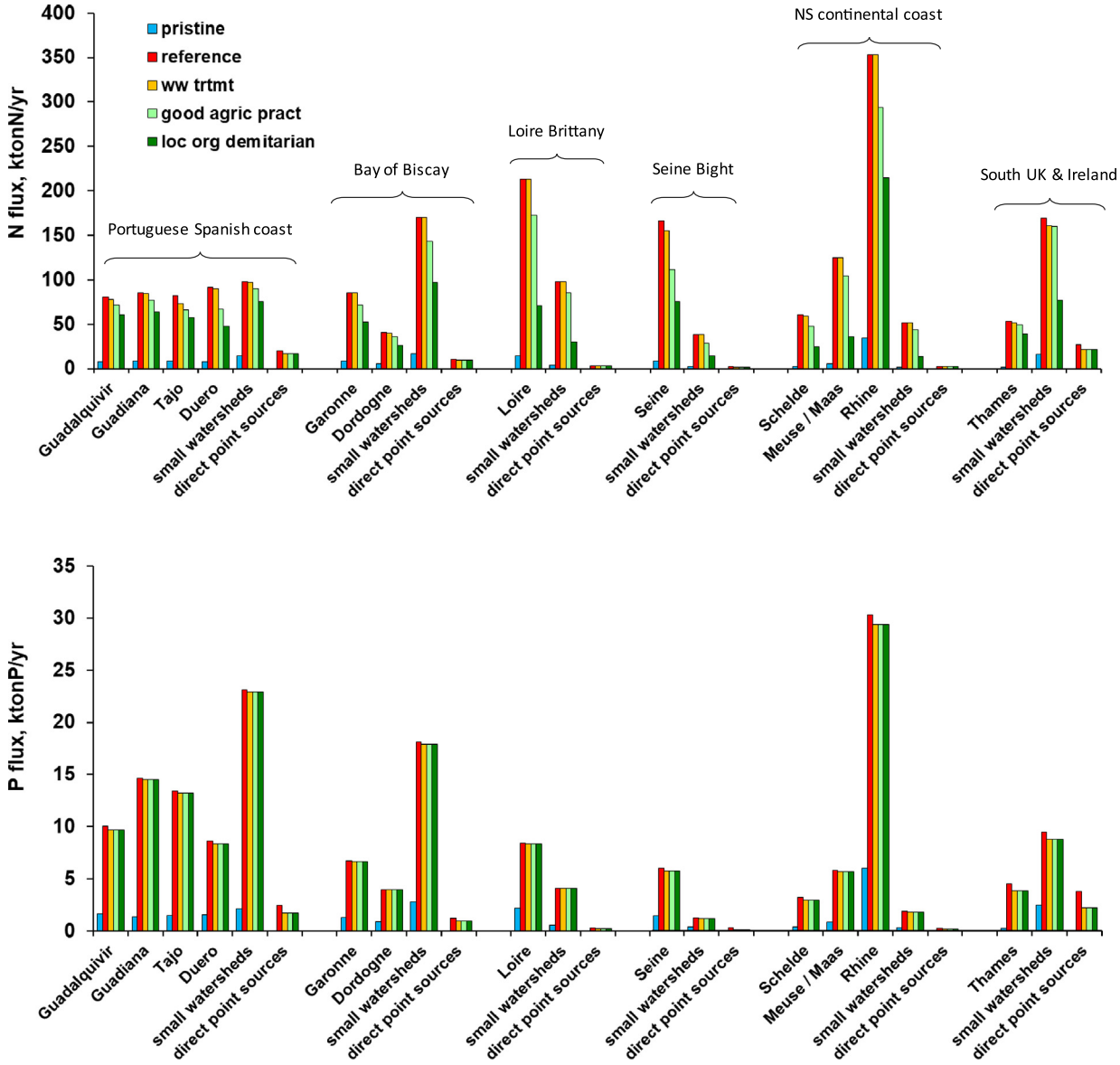

Fig. 7. Summary of the modelled fluxes of nitrogen (N, top panel) and phosphorus (P, bottom panel) delivered to the sea by the main rivers and the total of the small ones along large coastal areas of the NEA, for the Pristine and Reference situations and three prospective scenarios. The direct point sources of nutrient are also indicated (equaling zero for the Pristine situation).

the LocOrgDem scenario, especially in the Bays of Seine and Somme and in the northern and southern shores of Brittany. In addition, high Chl P90 values are not observed anymore at the mouths of Portuguese rivers. The Chl P90 is one of the possible indicators of eutrophication because it represents the height of the annual peak (generally occurring in spring and controlled by P). However, Chl P90 does not reflect the

Table 4
Fluxes of N and P from land to sea distributed between the different coastal sectors in the reference situation, and % reduction in the different scenarios.

Sector	N fluxes					P fluxes		
	Reference	UWWTD	GAP	LocOrgDem	Pristine	Reference	UWWTD GAP LocOrgDem	Pristine
	Gg d^{-1}	%	%	%	%	Gg d^{-1}	%	%
Spain - Portugal	1.21	−3.1	−14.9	−30.1	−89.4	0.20	−2.7	−88.5
Bay of Biscay	0.82	−0.1	−15.2	−40.7	−89.9	0.08	−2.0	−83.1
Loire - Brittany	0.86	0.0	−16.8	−67.5	−94.1	0.04	−1.3	−77.7
Seine - Somme	0.57	−5.8	−31.2	−56.1	−94.5	0.02	−6.7	−75.1
Rhine - Scheldt	1.64	−0.3	−17.1	−51.2	−92.6	0.12	−3.6	−82.0
British Isles	0.62	−4.4	−6.2	−47.7	−93.1	0.04	−10.0	−80.9

Table 5
Statistics of the PDFs shown in Fig. 8: 'mu' is the central tendency of the PDF (in unit μg Chl L^{-1} or μmol nutrient L^{-1}, resp.), 'xmax' is the maximum value of modelled Chl P90, winter DIN or winter DIP in the NEA coastal domain, and 'area' is the percentage of the NEA coastal area where values exceed the Pristine 'xmax' (vertical dotted line in Fig. 8).

	Chl P90			Winter DIN			Winter DIP		
	mu	xmax	Area (%)	mu	xmax	Area (%)	mu	xmax	Area (%)
Pristine	2.1	7.1	0.0	4.6	15	0.0	0.40	0.79	0.0
Reference	4.3	41	27	22	569	81	0.63	3.2	43
LocOrgDem	3.9	36	20	15	132	62	0.61	3.0	36

length of the bloom or the succession of phytoplankton species, which are both strongly influenced by N inputs. A more detailed analysis of these aspects is shown in a test case below in Section 3.4.

3.3. Probability distribution functions of coastal values

At the higher range of human impacts, the Reference situation exhibits nutrient concentrations in the coastal domain above the ones recommended by the MSFD: winter DIN concentration shows a spatial mean of 22.3 μmol L^{-1} and a maximum value of 569 μmol L^{-1} while winter DIP shows a spatial mean of 0.63 μmol L^{-1} and a maximum value of 3.20 μmol L^{-1} (Table 5). Indeed, the threshold values for winter DIN and DIP concentrations mentioned in the MSFD are respectively 8 μmol N L^{-1} and 0.5 μmol P L^{-1} for Portuguese coastal areas, 29 μmol N L^{-1} and 1.8 μmol P L^{-1} for France and 15 μmol N L^{-1} and 0.8 μmol P L^{-1} for Belgium. Under Pristine conditions, nutrient concentrations show values below 14.7 μmol L^{-1} for winter DIN and 0.79 μmol L^{-1} for winter DIP; and winter DIN concentration across the NEA coastal domain shows a spatial mean of 4.60 μmol L^{-1} and the spatial mean of winter DIP is 0.40 μmol L^{-1}. In the LocOrgDem scenario, winter DIN concentrations are considerably lower than in the Reference situation: the spatial mean value decreases by >33% and the maximum value decreases by almost 80%. Such a considerable decrease is not observed for winter DIP: its spatial mean drops by only 3% while its maximum value drops by 7% (this drop is due to UWWTD scenario mainly).

The spatial extent of coastal eutrophication is further illustrated by the frequency distribution of nutrient concentrations across the coastal domain of the NEA (Fig. 8). The highest nutrient concentrations under Pristine conditions could arbitrarily be chosen as a natural landmark for nutrients (vertical dotted line in Fig. 8). In that case, the percentage area of the coastal domain showing values above that limit would then depict the area subjected to human-induced eutrophication. For instance, under the Reference conditions, winter DIN exhibits values above its Pristine maximum (14.7 μmol L^{-1}, Table 5) in 81% of the NEA coastal domain. This is reduced to 62% in the LocOrgDem scenario. When winter DIP is considered, the coastal area under eutrophication drops from 43% to 36% between the Reference conditions and the LocOrgDem scenario.

Regarding Chl P90, under Pristine conditions, a maximum "natural" value would be 7.1 μg L^{-1}. This maximum Chl P90 value increases to 41 μg L^{-1} under the Reference conditions and 27% of the coastal domain is above the Pristine maximum value. In the LocOrgDem scenario, the Chl P90 maximum value drops to 36 μg L^{-1} (a drop of 12%) and a bit <20% of the coastal domain exhibits Chl P90 values above 7.1 μg L^{-1}. Despite a considerable reduction in DIN and a return to nutrient balance, this limited effect of the LocOrgDem scenario on the Chl P90, illustrates the non-linearity of the biological process. The main effect of the LocOrgDem scenario compared with the other two scenarios (UWWTD and GAP) is an important reduction of N loading. This does not reflect in a strong reduction of Chl P90, because of P rather than N limitation in most of the coastal domain. However, although this is only partially dealt with in the marine ecological models, a bundle of evidences shows that high N:P conditions in coastal waters favour the growth of opportunistic species, toxic dinoflagellates, or the production of toxins by some diatom species like *Pseudo-nitzschia*. For instance, by balancing nutrients through N reduction, the LocOrgDem scenario positively affects the planktonic structure as shown below in a case study (Section 3.4).

3.4. Phytoplankton dynamics: case study in Belgian waters

To illustrate the relative effect of the GAP and the LocOrgDem scenarios (moderate to strong N reduction and slight P reduction) on the modelled plankton dynamics, a test case is shown at the Belgian station

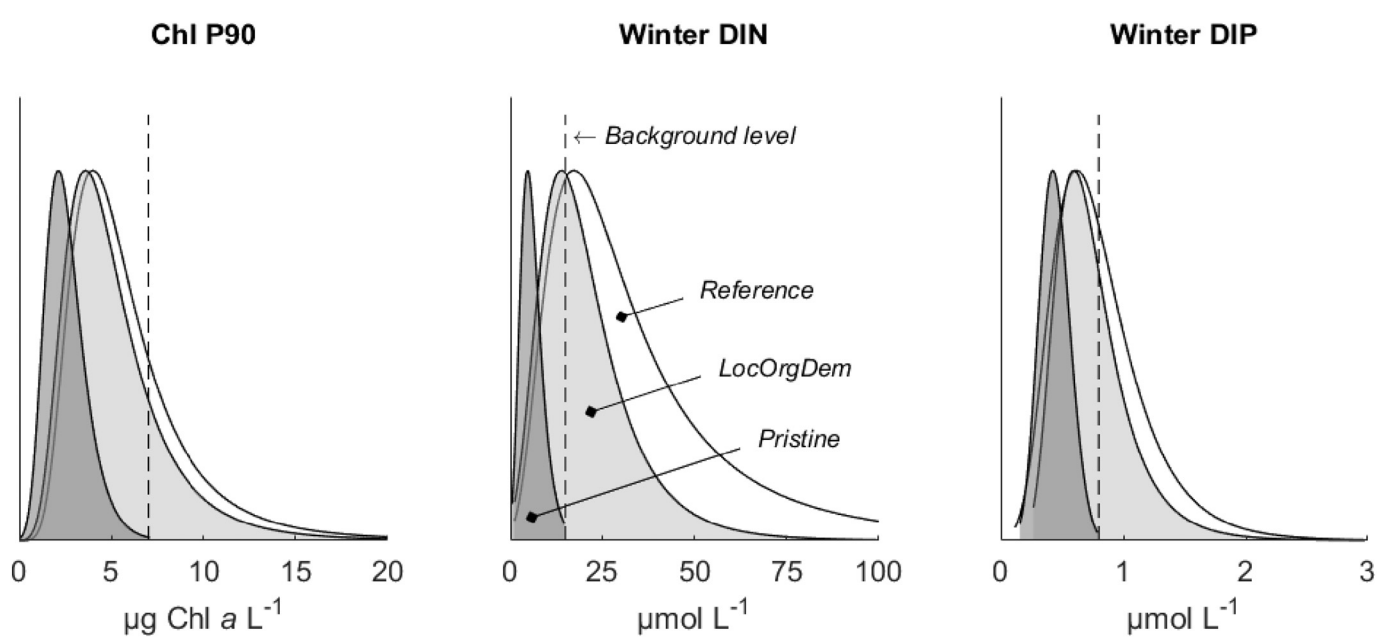

Fig. 8. Probability distribution functions (PDFs) of Chl P90, winter DIN and winter DIP concentrations in the coastal domain of the NEA for the Reference and Pristine situations and the LocOrgDem scenario. The PDF of coastal values were computed from the data shown in Figs. 5 and 6. The vertical dotted lines indicate the background levels chosen as the superior limit value of the corresponding Pristine PDFs (7 μg L^{-1} for Chl P90, 15 μmol L^{-1} for winter DIN and 0.8 μmol L^{-1} for winter DIP; see Table 5).

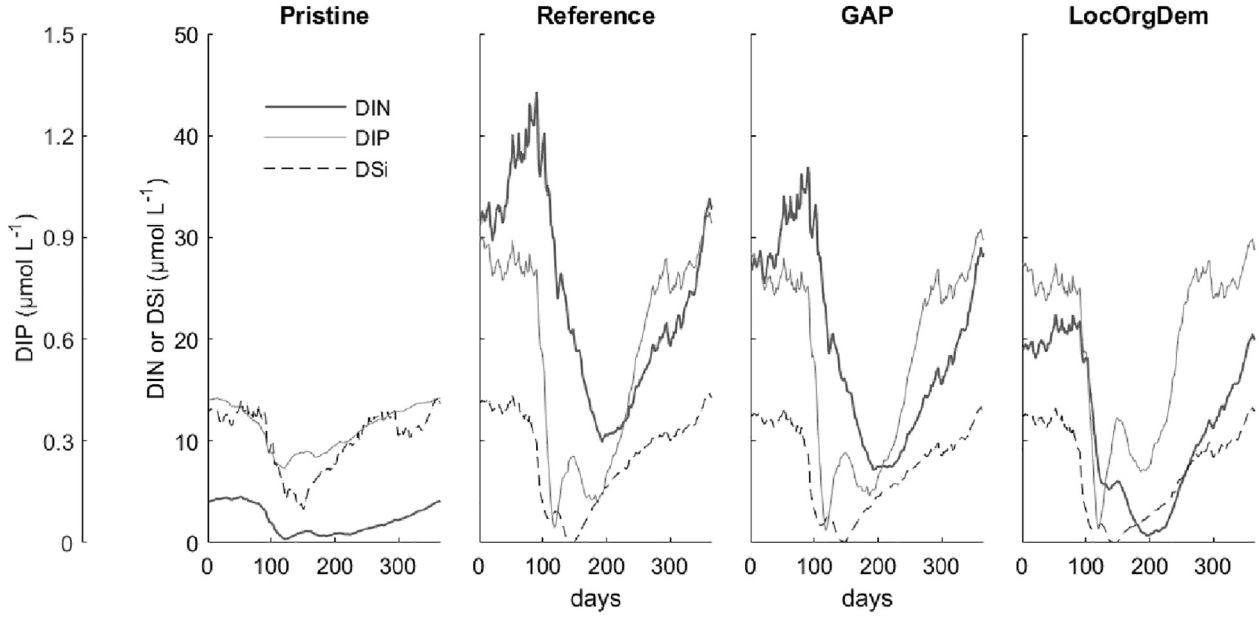

Fig. 9. Nutrient time series at station 330 in the Southern North Sea ([51.4°N 2.83°W]; results from MIRO&CO, multiyear mean 2000–2010). The Pristine and Reference situations are compared to the scenarios GAP and LocOrgDem.

330 ([51.4°N 2.83°W], Southern North Sea). Figs. 9 and 10 respectively illustrate the seasonal change in nutrient concentrations and the seasonal phytoplankton succession. In the Pristine situation, the winter N is depleted by the production of early spring diatoms before Si or P reach limiting concentrations in the model. *P. globosa* colonies are only found in very low concentrations as the low amount of nutrients

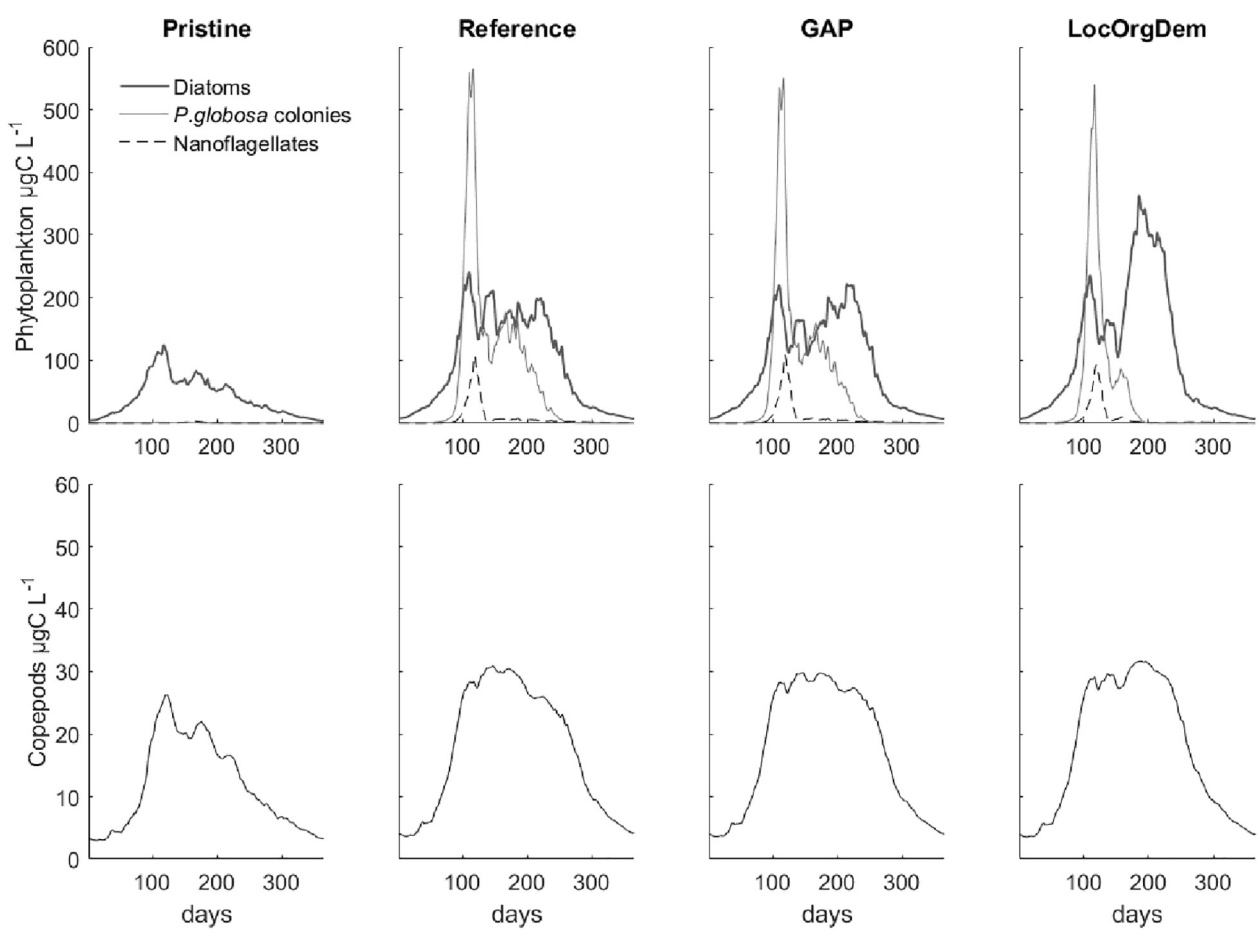

Fig. 10. Phytoplankton and zooplankton biomass time series at station 330 in the Southern North Sea ([51.4°N 2.83°W]; results from MIRO&CO, multiyear mean 2000–2010). The Pristine and Reference situations are compared to the scenarios GAP and LocOrgDem.

inhibits their net growth (i.e., modelled concentrations < 1 μg C L^{-1} in the Belgian and Dutch areas). In summer, diatom production is ensured by local nutrient regeneration and inputs from the ocean and the land.

In the Reference situation, the winter DIN and DIP (Jan–Feb) increase by a factor 8 and by a factor 2 respectively, compared to the Pristine situation (Fig. 9). The early spring diatoms deplete the dissolved silica much before depleting N or P content in the water column. The considerable amount of nutrients remaining fosters the growth of *P. globosa* colonies (and nanoflagellates to a lower extent) until DIP is depleted and DIN concentration reaches 25 μmol L^{-1}. After the spring bloom collapses, modelled diatoms and *P. globosa* maintain a significant production that mainly depends on the inputs of P (and Si for diatoms), either from recycling or from riverine inputs. The large spring colonies of *P. globosa* are inedible by copepods and fuel the microbial loop, which results in a significant increase in the mean biomass of bacteria (Table 6). According to the model, the annual net phytoplankton production (NPP) increases by a factor 4.5 between the Pristine and the Reference situation but the annual copepod grazing flux increases only by a factor 2.2 (Table 6). The annual mean diatom biomass increases by a factor 1.6 compared to the Pristine situation and the copepod biomass increases by a factor 1.5, which constitutes a moderate change compared to the concomitant increase in winter nutrients (factor 8 for N and 2 for P). These figures suggest that the current eutrophication mostly sustains a high phytoplankton production without increasing the copepod grazing to the same levels, with the trophic efficiency reduced by half. This supports Daro et al. (2006) and Lancelot et al. (2009), who already suggested that eutrophication in that area merely enhances undesirable blooms of inedible *P. globosa* colonies instead of fueling higher levels of the trophic web.

To mitigate the current eutrophication, the nutrient reduction in the GAP scenario (which includes the reductions of the UWWTD scenario) results in a measurable decrease in N and P winter concentrations at sea (in winter, N is reduced by 15% and P by 6% at station 330 compared to Reference). However, that effort in nutrient reduction does not result in a change in the phytoplankton or zooplankton dynamics. Both diatoms and *P. globosa* annual biomasses are slightly reduced following a reduction in NPP. However, the planktonic structure does not change considerably and the trophic efficiency only increases by 2% compared to Reference.

In contrast, the LocOrgDem scenario (which includes the reduction of UWWTD) imposes a more drastic nutrient reduction (in winter, N is reduced by 41% and P by 8% at station 330) and induces a significant change in phytoplankton dynamics. While *P. globosa* annual biomass decreases by 41% (especially in late spring), the annual diatom biomass concomitantly increases by a factor 1.4 (mainly large summer diatoms, *Guinardia*). The nutrient reduction lowers the annual NPP and the total phytoplankton biomass but remarkably it imposes a shift in phytoplankton species favouring diatoms. It may seem counter intuitive that diatom biomass increases when nutrient concentrations decrease but this new assemblage of species is only the result of rebalancing winter N and P levels with DSi (almost unaffected by scenarios).

In spring, phytoplankton production consumes the winter DIN in the water column and in summer DIN concentration is five times lower than in the Reference situation (~5 μmol L^{-1} instead of ~25 μmol L^{-1}). When summer diatoms start to grow, they efficiently use up the remaining N, which leaves no room for *P. globosa* to grow. Though, the annual flux of NPP is lowered by 17% in the LocOrgDem scenario compared to the Reference situation, the annual flux of copepod grazing slightly increases and the trophic efficiency increases by 24%. The Chl P90 (spring peak of Chl, mainly *P. globosa*) only slightly decreases in the LocOrgDem scenario despite the large N reduction as the spring bloom peak is limited by P (Billen et al., 2011; Desmit et al., 2015a). However, the duration of *P. globosa* bloom is shortened in the LocOrgDem scenario compared to the GAP scenario and the Reference situation. This case study shows that only a drastic decrease in nutrient inputs resulting from profound changes in the watersheds can positively alter the dynamics of marine phytoplankton and zooplankton in that area.

Table 6
Multiyear mean modelled biomass of plankton and bacteria at station 330 [51.4°N 2.83°E] in the Southern North Sea (period 2000–2010; modelled colonial *P. globosa* expressed in μg C L^{-1} includes mucus carbon). Annual modelled fluxes of net phytoplankton production (i.e. NPP = photosynthesis − respiration) and zooplankton grazing at same site (period 2000–2010). The trophic efficiency is the ratio between grazing and NPP.

	Units	Pristine	Reference	GAP	LocOrgDem
Yearly mean biomass					
Diatoms	μg C L^{-1}	41.0	93.9	90.7	109
P. globosa	μg C L^{-1}	0.472	66.4	57.7	39.3
Nanoflagellates	μg C L^{-1}	0.484	7.52	7.47	6.15
Total phytoplankton	μg C L^{-1}	42.0	168	156	155
Copepods	μg C L^{-1}	11.7	17.5	17.4	17.4
Bacteria	μg C L^{-1}	5.90	16.4	15.7	15.0
Annual flux					
Phytoplankton NPP	g C m^{-2} yr^{-1}	73.6	334	309	278
Copepod grazing	g C m^{-2} yr^{-1}	44.0	95.2	94.8	98.0
Trophic efficiency	%	59.8	28.5	30.7	35.3

4. Discussion

4.1. Model hypotheses

The choice of boundary conditions may possibly affect the results of our study. Thus, in the Pristine situation, an input of winter DIN and winter DIP along the UK is visible east of the Irish Sea and west of the central North Sea, which is very likely overestimated (Fig. 5). There, the nutrient concentrations at the boundary were imposed from existing datasets (see Section 2.2.4), which include the nutrient plumes of rivers north from the model boundary and not reduced by the Pristine scenario. Also, the northern boundary of the North Sea (eastern part of the boundary) shows an accumulation of Chl that is especially visible in the Pristine situation (Fig. 6). This is an artefact due to the zero-gradient boundary applied there (see Section 2.2.4). This artefact has a very limited influence on the processes occurring in the Southern North Sea. A transboundary nutrient transport study, i.e., the identification of the relative contributions of different sources in N to the marine ecosystem, was carried out with the model tagging technique (i.e., additional state variables are added to the model; they are transported and processed like nutrients along the same pathways but are coming from unique sources, whether river or boundary; all sources of nutrients may be covered with specific simulations and their contribution analysed in any chosen area; Dulière et al., 2017). According to the results, it is unlikely that the Chl accumulation at the northern boundary affects model estimates in the Southern North Sea due to the northward residual water transport.

The strength of the pyNuts-Riverstrahler model is to be able to reproduce seasonal variations of N and P fluxes to the sea for the entire catchment area in Western Europe with a fixed set of parameters. The risk is that some river loads may be over- or underestimated locally. Yet, Fig. 2 illustrates that the agreement between model and data is rather satisfactory considering the model is not calibrated for any particular basin. Additionally, such large-scale validation of the pyNuts-Riverstrahler model raised-up the challenge of collecting a sufficient amount of water quality (and quantity) measurements all along the NEA domain. A total of eleven water authorities were requested to obtain daily records of nutrient and discharge, with a clear imbalance in the availability and frequency of monitoring between the main NEA rivers and between nutrients (with an obvious deficiency for silica). The marine models receive all their river loads from pyNuts-Riverstrahler in the Pristine and Reference situations and in every prospective scenario. The focus of the study being to compare the effects of different scenarios with respect to a Reference, any deviation of the simulated fluxes from reality in the Reference situation loses weight in the relative comparison.

Estuarine processes, which we did not consider in the present study, should probably be considered when their nutrient filtering role is significant with respect to the nutrient budget of the receiving coastal basin. Estuarine filtering functions may likely change because of scenarios and so would the nutrient fluxes to the sea. However, the heavy management of estuaries worldwide has impaired their ecosystem functioning and often strongly reduced their filtering capacity (Lotze et al., 2006 and Garnier et al., 2010 for the Seine River). A refined study of estuaries and transitional waters would require ad-hoc models, developed specifically in the studied areas, and the scale of our approach was unfit for this challenge.

4.2. Assessing background values for indicators and distance to GES target

The baseline of eutrophication assessments in EU policies (WFD, MSFD) is the estimate of a "pristine" status, a "natural background" in the marine ecosystem, to which the current level of eutrophication is compared. This generates many questions such as whether all ecosystems have the same pristine conditions or not. No real measurement is available to answer these questions or to describe the coastal ecosystem in its pristine stage. Estimates of pre-industrial N enrichment in the German Bight based on sediment core analyses were proposed by Serna et al. (2010) and tend to show that pre-industrial levels would be reached locally with 10% of current N river loads. The model outputs in our study give additional clues about the pristine conditions at the scale of the NEA. The nutrient fluxes from land to sea in the Pristine scenario (−89% to −95% for N and −75% to −89% for P compared to Reference, Table 4) support the estimate of Serna et al. (2010). With Pristine nutrient fluxes, the receiving coastal basins across the NEA show very low nutrient concentrations (spatial mean of winter DIN is 4.6 μmol L^{-1} and winter DIP 0.4 μmol L^{-1}; Table 5), the indicator Chl P90 has a spatial mean of 2 μg L^{-1} across the NEA coastal zones and, in our test case, the trophic efficiency is double than in the Reference situation (Table 6). These model outputs are indicative of what a Pristine stage could be in the NEA waters and we propose to extrapolate on these results to estimate a "natural background value" for the indicators of eutrophication. The "natural background value" of an indicator could be chosen as its maximum value under Pristine conditions in the NEA coastal domain, independent of national boundaries (the tail of its PDF in Fig. 8, i.e., xmax in Table 5). For instance, the background value for Chl P90 would be 7 μg L^{-1} across the NEA coastal waters according to our results. The background values for winter N and winter P in the NEA coastal domain would be respectively 15 μmol N L^{-1} and 0.8 μmol P L^{-1}. With these natural background values, the present study might also contribute to confirm or re-assess the boundary values used in EU policies for nutrients and Chl indicators. In the frame of EU policies, when assessing the Good Environmental Status of a system, the natural background value of an indicator may be used to recalculate its "very good-to-good" and "good-to-moderate" boundaries. A conventional method to perform the calculation is to add 50% of a boundary value to reach the next boundary value, starting with the "natural background". For instance, if the background value for Chl P90 was assumed to be 7 μg L^{-1} in the whole NEA coastal zones (Table 5) then its "very good-to-good" boundary could be 10.5 μg L^{-1} (+50%) and its "good-to-moderate" boundary could be 15.8 μg L^{-1} (+50%), a value close to the one proposed by France (in the English Channel) and Belgium in the frame of the EU WFD (15 μg L^{-1}).

How far are EU Member States from reaching the "Good" status in their Atlantic coastal waters? This question was addressed by Ménesguen et al. (2018, this issue) with the so-called 'distance-to-target' approach (DTT). Based on the hypothesis that high phytoplankton biomass occurs where nutrient concentrations are also high, the DTT has been defined as the nutrient reduction necessary in the rivers to reach the eutrophication GES in a marine region (Lenhart et al., 2013; Los et al., 2014). The strong N load reductions, which the LocOrgDem scenario focuses on, can be compared to the optimal N load reductions estimated by Ménesguen et al. (2018, this issue). Looking at the MSFD area within the English Channel and the Southern North Sea, the optimal N load reductions according to these authors should be −95% in the Seine, −98% in the Scheldt and −77% in the Rhine/Meuse compared to the current situation (decade 2000–2010; rivers N loads are forced by observations). In our study, the LocOrgDem scenario shows a N load reduction of −56% in the Seine and −51% in the Scheldt + Rhine/Meuse compared to the Reference situation (decade 2000–2010; rivers N loads are forced with pyNuts-Riverstrahler outputs; Table 4). The LocOrgDem scenario results in lesser N load reductions than required according to Ménesguen et al. (2018, this issue) and, indeed, the LocOrgDem scenario is not sufficient to reach the GES everywhere in the NEA coastal zones. Actually, the optimal N load reductions according to Ménesguen et al. (2018, this issue) are closer to the Pristine conditions in our study (Table 4). This confirms that reaching the WFD and MSFD N targets everywhere will be difficult and will probably require deep and structural transformations of the order of magnitude as proposed in LocOrgDem.

4.3. Geographical extension of marine eutrophication

The spatial analysis across the NEA coastal zone, summarised by the PDFs (Fig. 8, Table 5), emphasises the geographical extent of eutrophication at sea and shows how scenarios reduce its surface. Until recently, most assessments of marine eutrophication have focused on an averaged perception of water quality within an area: the studied area is on average "very good", "good", "moderate" etc. (WFD); it is a "problem area" or a "non-problem area" (OSPAR). With the MSFD assessments, more attention is given to the geographical extension of eutrophication in EU waters. In this context, observing the evolution of the area subject to eutrophication could become a criterion for assessing the effects of any measure taken upstream. For instance, with the LocOrgDem scenario, the area having winter DIN concentrations above 15 μmol L^{-1} (maximum value in Pristine) is reduced by 23% [(80.8–62.4)/80.8] relatively to the Reference situation (Table 5, Fig. 8). It means that, regarding winter DIN, an absolute percentage of the NEA coastal zone (+18%) returns to its "natural background" conditions as defined above. The area where Chl P90 remains higher than 7 μg L^{-1} is reduced by 26% [(26.9–19.8)/26.9] across the NEA coastal zone. The impact of restoring part of the NEA coastal zone into healthy conditions may be of importance from the ecological perspective as healthy ecosystems may positively influence surrounding waters. The relative significance of these improvements from a policy point of view is still a matter of discussion, also linked with the choice of appropriate indicators. As shown in Section 3.4, the Chl P90 is not the only indicator to be considered when measuring the positive effects of a nutrient reduction.

4.4. Options for the future

In the last decades, EU Directives have made only a limited impact on the quality of European coastal waters, partly due to social and economic factors (Grimvall et al., 2000; Artioli et al., 2008; Bouraoui and Grizzetti, 2011). By estimating nutrient loads to a few vulnerable European coastal zones in three periods (before eutrophication, during high eutrophication and in the current situation), Artioli et al. (2008) showed that EU legislations had measurable impacts on reducing point source emissions (especially P) but less impact on reducing diffuse emissions (especially N). This conclusion is not only explained by the delay in the system response to nutrient reductions (Grimvall et al., 2000), but also by the infringement to the EU Nitrate Directives in several EU Member States (Bouraoui and Grizzetti, 2011). There is indeed some uncertainty about how the economic actors may react to environmental regulations due to the economic and social constraints they are already facing. In the context of fisheries, it has already been shown that understanding and anticipating actors behaviour is key to manage a

sustainable food production, and hence to maintain or restore healthy natural ecosystems (Paterson et al., 2010; Fulton et al., 2011). One can anticipate that any deep structural change in agriculture at the scale of European watersheds would probably affect economic activities in at least the sector of the food production and good transportation. Adaptive management regimes (Pahl-Wostl et al., 2007) and collaborative governance through stakeholder engagement may foster new ideas and pathways to support a large-scale endorsement of the future measures. A transition towards more sustainable land and water management options will probably result from a transdisciplinary reflection on agricultural, social, economic and cultural (diet) practices.

The present study tested the effect in the NEA coastal areas of a profound change in the agricultural paradigm with the LocOrgDem scenario applied in adjacent watersheds. The scenario included a reconnection of crop and livestock farming, of food production and consumption, and the hypothesis of a generalised demitarian diet in Western Europe. By including a transition to less impactful diets and reduced trade of animal products, the LocOrgDem scenario allows sustaining food security while minimizing environmental impacts (Davis et al., 2016). Among the explored scenarios, the LocOrgDem scenario is the only one that substantially improved the modelled water quality in European rivers. By rebalancing nutrient fluxes to the sea, it would reduce coastal undesirable phytoplankton blooms, an adverse effect of eutrophication. Reducing nutrient emissions from land to sea down to Pristine levels (or even to pre-industrial levels) has been considered from its outset as unthinkable (Diaz and Rosenberg, 2008): it would require a zero-level nutrient emission from anthropogenic activities, including atmospheric N deposition. Estimating pristine conditions is useful for measuring the amplitude of the human impacts and to possibly define background levels but these conditions are not a realistic management objective. Significant nutrient reductions can probably be achieved through a combination of agricultural, social and economic measures. The purpose of the study was not to present the LocOrgDem scenario as the best or unique solution to mitigate coastal eutrophication. The LocOrgDem scenario is not prescriptive but only an attempt to explore some trends already acting on the dynamics of the current system via a consistent and deep change that western societies could support. It offers an extreme vision of what could be the agro-food system, and can certainly be modulated with other options.

One of the conclusions of this study is that it is realistic to expect a large-scale improvement of water quality by transforming and adapting human activities in the watersheds: i.e., a strong improvement of freshwater quality in the rivers with implications for drinkable water, a decrease in coastal eutrophication symptoms including their geographical extension at sea, and a change in marine phytoplankton dynamics in favour of a higher trophic efficiency and healthy ecosystems. Another conclusion is that any significant achievement in that direction, especially regarding the marine ecosystem, would include profound, structural changes in the human agro-food system and land use.

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