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Science of the Total Environment 408 (2010) 1245–1255

Contents lists available at ScienceDirect

Science of the Total Environment

journal homepage: www.elsevier.com/locate/scitotenv

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ARTICLE INFO

Article history:
Received 3 May 2009
Received in revised form 22 November 2009
Accepted 17 December 2009
Available online 13 January 2010

Keywords:
Nutrient reduction
Eutrophication
Seine
Somme
Scheldt
Seneque–Riverstrahler model

ABSTRACT

The Seine, Somme, and Scheldt Rivers (France, Belgium, and Netherlands) are the major delivering rivers flowing into the continental coastal zone of the Southern Bight of the North Sea, an area regularly affected by eutrophication problems. In the present work, the Seneque–Riverstrahler model was implemented in a multi-regional case study in order to test several planned mitigation measures aimed at limiting stream nutrient contamination and restoring balanced nutrient ratios at the coastal zone.

This modeling approach, which is spatially distributed at the basin scale, allows assessing the impact of any change in human activities, which widely differ over the three basins. Here, we define realistic scenarios based on currently proposed measures to reduce point and non-point sources, such as the upgrading of wastewater treatment, the introduction of catch crops, and the development of extensive farming. An analysis of the current situation showed that a 47–72% reduction in P point-source emissions within the three basins could be reached if the intended P treatment was generalized to the largest treatment plants. However, only an overall 14–23% reduction in N could be achieved at the outlet of the three basins, by combining improved wastewater treatment and land use with management measures aimed at regulating agricultural practices. Nonetheless, in spite of these efforts, N will still be exported in large excess with respect to the equilibrium defined by the Redfield ratios, even in the most optimistic hypothesis describing the long-term response of groundwater nitrate concentrations.

A comprehensive assessment of these mitigation measures supports the need for additional reductions of nutrient losses from agriculture to control harmful algae development. It also stresses the relevance of this mechanistic approach, in which nutrient transfers from land to sea can be calculated, as an integrated strategy to test policy recommendations.

1. Introduction

The second half of the 20th century, was characterized by unprecedented contamination of surface waters by anthropogenic sources in Western European countries. While a general stabilization or reduction of phosphorus (P) emissions from land to coastal zone is now observed (Behrendt et al., 1999), nitrogen (N) trends have not been similarly attenuated and are still increasing in many places. Primary production and phytoplankton growth are among the ecological functions of aquatic systems that have been mostly affected by increasing human pressure, as they are controlled by nutrient availability (Billen et al., 2007). As a consequence, there has been a general eutrophication of both continental surface waters and marine coastal areas in Western Europe (Cugier et al., 2005; Lancelot et al., 2005). Moreover, the imbalance in nutrient loading has induced shifts in algal communities that are undesirable or even toxic (Justic et al., 1995; Rousseau et al., 2000; Turner et al., 1998).

At the same time, public awareness of the resulting environmental deterioration has led to attempts at decreasing river pollution (Iversen et al., 1998). Attempt at reducing nitrogen and phosphorus were considered in turns independently until the adoption of the EU Water Framework Directive (WFD, 2000/60/EC) that settled targets for each nutrient to achieve a good ecological status in fresh and coastal waters.

Here, the terrestrial continuum that includes the Seine, Somme, and Scheldt River basins, the three major delivery systems to the Southern Bight of the North Sea, offers an interesting example of a transborder territory with high-intensity anthropogenic pressures but characterized by considerable disparities regarding their spatial distribution. The need of quantitative approaches able to formalize the link between nutrient sources and nutrient loads at the outlet of the river was demonstrated at the scale of these three river basins (Billen et al., 2009; Thieu et al., 2009). Based on the Riverstrahler model, these works provided accurate descriptions of the past and

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doi:10.1016/j.scitotenv.2009.12.031

present status of nutrient transfers and a detailed census of current human pressures for these three basins. It thus offers a realistic framework for further testing the effects of regulations that aim at reducing nutrient emissions.

In this paper, we assess the results that can be expected from currently planned management measures in terms of river water contamination and nutrient delivery to the coastal zone. Based on a the mechanistic and multi-element Riverstrahler model, changes in riverine fluxes and balance between nitrogen, phosphorus and silica are jointly analysed with respect to their potential to support harmful algae development. In the final section of this paper we quantify and discuss the additional long-term efforts required to further reduce nutrient pollution.

2. Methodology

2.1. Study area

Driven by the circulation of Atlantic waters, the Seine River plume travels upward along the French Eastern Channel, gathering contributions from the Somme River before reaching the North Sea and the Belgian continental coastal zone until the Scheldt estuary at the Netherlands border (Fig. 1). The area is currently described as suffering from eutrophication and has been designated as "a problem area" according to the OSPAR strategy (2005), as both French and Belgian coastal zones undergo regular undesirable developments of dinoflagellates. The most spectacular expression of this eutrophication is the massive mucilaginous algal blooms of *Phaeocystis* that occur every spring in the North Sea (Lancelot et al., 2005), although other harmful toxic species (as *Dinophysis*) are often observed at the Seine bight (Cugier et al., 2005). The present work focuses on the Seine, Somme, and Scheldt (Table 1) Rivers, since they represent the major river systems; indeed the southern extent of the Rhine River plume is of much less influence, as shown by Lacroix et al. (2007), and has therefore been disregarded in our approach.

For these three watersheds, the early 1950s marked the beginning of a period of mass domestic consumption as well as a shift to modern agricultural practices based on the use of synthetic fertilizers. As a consequence, urban loading and diffuse sources increased, reaching a plateau in the late 1980s, before decreasing due to a stabilization of the population, to improved wastewater treatment, and to the ban on polyphosphates in laundry detergents. However, nitrogen contamination of surface water and groundwater is still important. In addition, phosphorus levels, a major source of eutrophication of freshwater systems, continue to remain an important issue in drinking water production. Past reconstruction of the Seine (Billen et al., 2007) and the Scheldt (Billen et al., 2005) have pointed out the major changes that have occurred in the rural and urban environments due to a major affluence of the population to urban centers. One of the more striking change is the concentration of point sources of pollutants (e.g. the 10 million inhabitants of Paris conurbation), linked to the increased volume of urban wastewater and its discharge into surface water, sometimes without effective treatment (e.g. the case for wastewaters from Brussels with its 1 million inhabitants) directly discharged, without any treatment, until the recent implementation of two

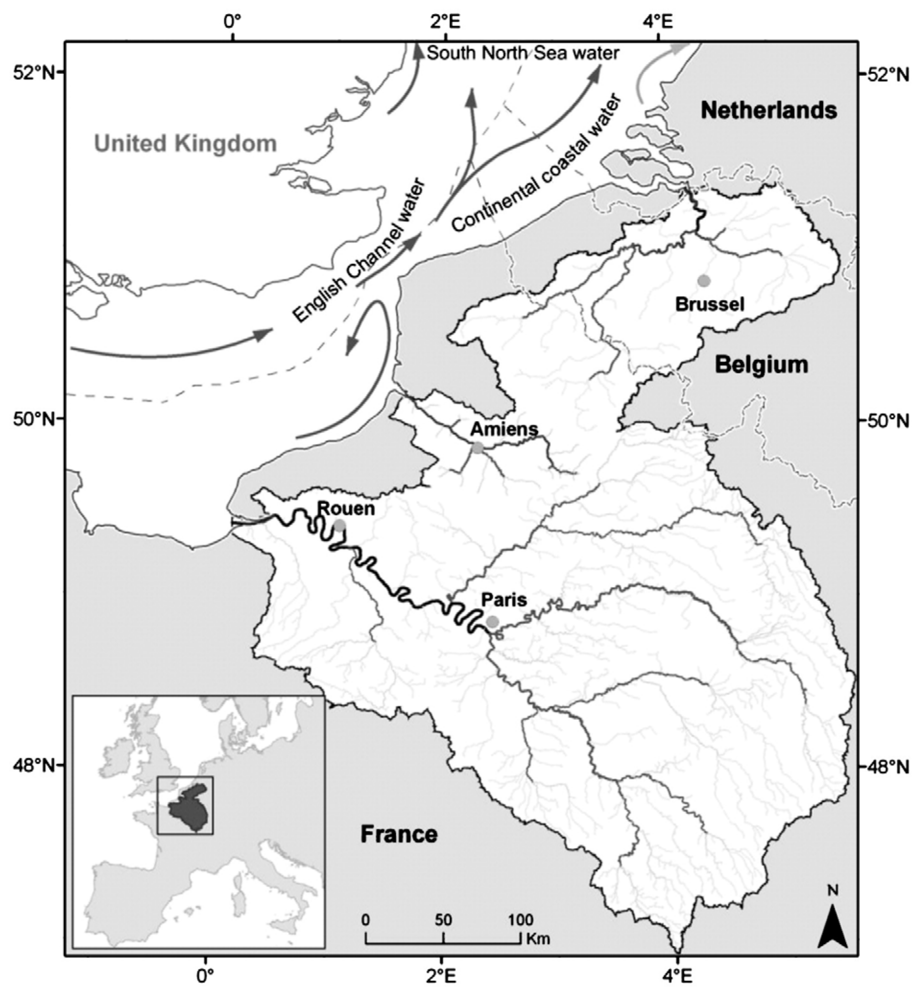

Fig. 1. Location of the Seine, Somme and Scheldt Rivers (3S) case study. Boundaries of adjacent coastal waters shown for each country are those of the EEZ (Exclusive Economic Zone) limit areas. The schematic circulation of water masses in the southern North Sea is derived from Turrell et al. (1992).

Table 1
General information on the basins of the Seine, Somme and Scheldt Rivers. Land-use values are derived from the Corine Land Cover (CLC: Bossard et al., 2000) and agronomical results are those of the French (Agreste, 2000) and Belgian (NIS, 2000) agricultural census.

	Seine	Somme	Scheldt
Population density, inhab km^{-2}	202	101	496
Total area, km^2	76,370	6190	19,860
Grassland, %	10	4	8
Arable land, %	53	77	39
Fodder corn crop, % arable area	2	4	14
Cattle density, LCU km^{-2}	15	23	70

purification plants, in 2000 and 2006. Other changes have relied on intensification and specialisation of agricultural activities. Such territory like the rural Somme basin (101 inhabitants per km^2) or the central part of the Seine are mostly dedicated to cereal and industrial crops, while the Scheldt (496 inhabitants per km^2) and the western fringes of the Seine support intensive animal farming activity.

2.2. The Seneque–Riverstrahler basin-scale approach

The first vocation of quantitative approaches of nutrient transport is to link riverine exports at the outlet of a (sub-)catchment to data on nutrient surplus in soil or net anthropogenic inputs (Boyer et al., 2002). Statistical regressions have been widely used for this purpose, from the simple lumped representations of aquatic systems (Howarth et al., 1996; MONERIS: Behrendt et al., 2002) to fully distributed approaches (GREEN: Grizzetti et al., 2005; SPARROW: Smith et al., 1997). An additional perspective lies in the understanding of the complex functioning of these systems, and requires process-based approaches of nutrient dynamics. The Riverstrahler model (Billen et al., 1994; Garnier et al., 1995) is one of the few models that describe all biogeochemical processes occurring within a drainage network and acting on the transformation, elimination and/or immobilization of nutrients (including all inorganic, organic, dissolved and particulate N, P and Si forms; see Garnier et al., 2002a for a complete description). The essence of the model is to couple water flows routed through the defined structure of basins and branches with a model describing biological, microbiological, and physicochemical processes occurring within the water bodies (in both the planktonic and the benthic phases). The calculation is managed by the Seneque software (Ruelland et al., 2007), which allows all the input files required for running the Riverstrahler model to be derived from a general GIS data base covering the watershed, at any location in the drainage area. It also enable a scalable representation of the drainage network, supporting the delineation of: i) sub-basins, each idealized as a regular scheme of confluent tributaries of increasing stream orders (Strahler, 1957) and characterized according to morphological properties, and ii) river branches represented with high spatial resolution (1-km stretches). Whilst most mechanistic models are suitable for modeling small catchments (e.g. INCA model: Wade et al., 2005; SWAT: Neitsch et al., 2005), our approach includes a multi-basins system exceeding over 100.000 km^2, in which we consider a combination of 46 “sub-basins” (1592 km^2 on average) and 14 “axes” (i.e. a total of 1991 km of river branches). Also, in contrast to other approaches providing (multi-year) annual nutrient fluxes (e.g. POLFLOW model: De Wit, 2001) the Riverstrahler model enables a 10-day time step for the simulation of the seasonal variations of water fluxes. Although the Riverstrahler drainage network approach does not explicitly simulate the land-based processes affecting nutrient dynamics in soils, it assumes that the variability of the responses of river systems are directly related to the environmental constraints including the diffuse and point sources of nutrients.

Point sources include household and industry sewage and are calculated on the basis of i) the effective volume released (effective treatment capacity of the plant), ii) the type of treatment applied, and iii) their localization, specified either by stream-order for a basin or at the exact point of release into branches within the drainage network.

Non-point sources are defined according to land use and agricultural practices (see Thieu et al., 2009, for a detailed description). For each sector of the drainage basin considered, both sub-root water and groundwater concentrations of each nutrient are provided as a constant concentration and then ascribed to the corresponding, seasonally variable discharge components, namely surface runoff and base flow; both of which are used to calculate diffuse fluxes of nutrients transferred to the river network.

2.3. Definition of reference conditions

2.3.1. Hydrological reference

The Seneque–Riverstrahler model's architecture makes a clear distinction between hydrological and anthropogenic constraints, distinguishing the effects of year-to-year hydrological variability from changes in human pressure. A detailed analysis of observed discharges was carried out and the three contrasting years, 1996, 2000, and 2001, representative of, respectively dry, mean, and wet hydrological conditions, were selected (Fig. 2). Regarding the Somme, the apparent weak seasonal variability of its specific runoff compared with that of the two other rivers is directly related to the chalky nature of the watershed, with an important share of base flow contributing to the total discharge, especially during the low-water season. The mean specific runoff of the Scheldt, calculated here for the years 1996 and 2001, is higher (6.3–15.3 l/km^2/s respectively for the two years) than that of either the Seine (4.8–13.1 l/km^2/s) or the Somme (4–11.8 l/km^2/s), in agreement with the higher rainfall over the Scheldt basin.

2.3.2. Pristine state

A pristine state, although unrealistic in Western European countries, is defined as the insignificant impact of pressures on ecosystem functioning and thus approximates the “natural environment.” The latter is defined as the non-impacted pristine state (which differs from the “good ecological state,” the aim of application of the WFD).

In our modeling approach, the pristine state of the watersheds has been established by erasing all human alterations of the river, including any point-source releases from industry or urban sewers and all hydraulic works (canalized sections, diversion of water, dams and ponds). Regarding diffuse inputs, within the latitudes of the three basins, a completely forest covered landscape is assumed such that leaf litter along the river banks represents the only, albeit significant, input of allochthonous organic matter that directly feeds the river system. This input has been estimated to 0.25–4 kg C km^{-2} $year^{-1}$ (e.g. from 25 to 400 g C m^{-2} $year^{-1}$, Bell et al., 1978; Chauvet and Jean-Louis, 1988; Cummins et al., 1980; Dawson, 1976; Herbst, 1980) with a C:N:P molar ratio of 1250:16:1 (Meybeck, 1982). This assumption led us to propose an average N and P inputs amounting to 0.06 kg N km^{-1} day^{-1} and 0.01 kg P km^{-1} day^{-1}, brought about by leaf litter along the river bank.

2.3.3. The present impacted state

The pristine state does not constitute a target for achievement of the WFD; rather, it is an end-point situation to be compared along a trajectory ranging from the present impact of land-based anthropogenic sources to effective prospective scenarios. This implies that the present level of pressure has to be quantified as another reference.

The year 2000 was selected as representative of present conditions because it is the year for which the most complete input database was available, including various field measurements and experiments that allow the implementation and validation of the Seneque–Riverstrahler model. For point sources, more recent improvements of sanitation were disregarded; for example, the important (1,100,000 inhabitant-equivalents) Brussels North sewage plant that came on line in 2006 within the Scheldt. For diffuse sources, updated land-use information (CLC2000: Bossard et al., 2000) and agricultural statistics (Agreste, 2000;

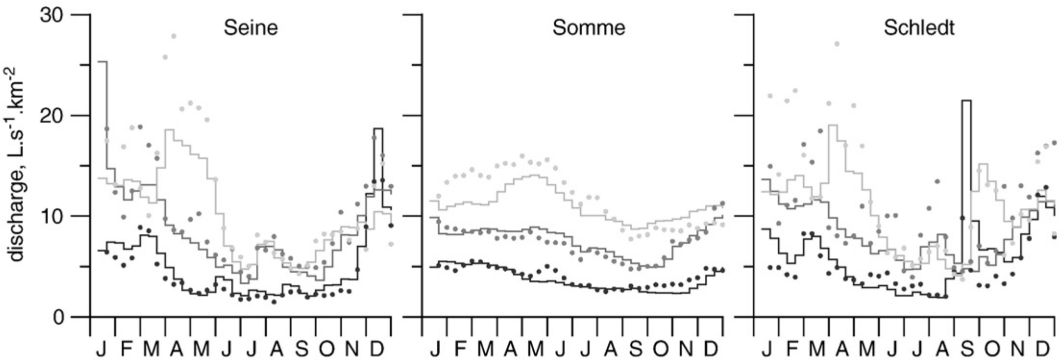

Fig. 2. Specific runoff calculated at downstream stations of the Seine (Caudebec), Somme (Cambron), and Scheldt (Doel) Rivers for the years 1996 (dark grey lines), 2000 (grey line), and 2001 (light grey line). Observations are given for comparison.

NIS, 2000) are available, both of which can be applied to define sub-root concentrations of nutrients. For groundwater, nutrient concentrations were derived either from an inventory of aquifer composition (ADES, 2003) or on the basis of empirical relationships issued from field measurements or model results (Gomez, 2002) for the same period (2000). These data revealed considerable differences, especially for nitrate, which showed recent and rapid increasing sub-root concentrations (Fig. 3). These differences can be interpreted as reflecting either a time lag in groundwater contamination from the root zone or a retention/elimination process occurring within the aquifer or the vadose zone. A more detailed description of sub-root and groundwater concentrations defined for the three basins is found in Thieu et al. (2009).

2.4. Scenarios and regulation options

A decade before the EU Water Framework Directive, several regulation were established to mitigate nutrient emission from land to river and coastal sea, as the EU Council Directive 91/676/EEC, known as the "Nitrate Directive," or the EU directive on urban wastewater treatment (91/271/EEC). The mitigation measures prescribed by these directives are still on the current political agenda, and are here retrieved to build realistic scenarios described in the following sections.

2.4.1. Improvement of wastewater treatments

Improvement in wastewater treatment efficiencies for both nitrogen and phosphorus is a serious issue in the present management of water resources in France and Belgium. In response, new water-treatment plants have been built and treatment facilities have been added or upgraded in order to reach 70% and 90% of nitrogen and phosphorus abatements, respectively. These levels were designated on the basis of the implantation of nitrification, denitrification, and dephosphatation processes with respect to the EU directive on urban wastewater treatment (91/271/EEC) recommendations. Starting from the reference situation (2000), this scenario was progressively calculated by first considering the largest plants (with a treatment capacity of over 500,000 inhabitant-equivalents), following by those with treatment capacities higher than 100,000 and 20,000 inhabitant-equivalents, and, finally, all identified urban releases, regardless of size. These scenarios were established taking into account an accurate description of the current sanitation situation of the three basins, thus ensuring a realistic assessment of the results to be expected from wastewater treatment policies aimed at improving river quality and nutrient fluxes exported at the mouth of the basins.

2.4.2. Decreasing nitrogen leaching from agricultural land

The list of recommendations issued from the Oslo-Paris convention (OSPAR, 2005) includes the reduction of N and P leaching and runoff from agricultural land, in particular the reduction of fertilizers used per hectare and the introduction of winter catch crops (also called nitrogen-trapping winter crops, NTWCs). These measures were also mentioned by the Nitrate Directive (91/676/EEC) in the code of good agricultural practices addressed to member states of the EU Community. The efficiency of catch crops like perennial ryegrass (Hansen and Djurhuus, 1997; Thomsen, 2005) is the result of their ability to absorb nutrients left over in the soil during the winter months, hence preventing them from leaching. Within the boundaries of the three basins, the findings of several specialized agronomical models support the benefit of NTWCs, as reported by Sohier and Dautrebande (2005) in the Walloon part of

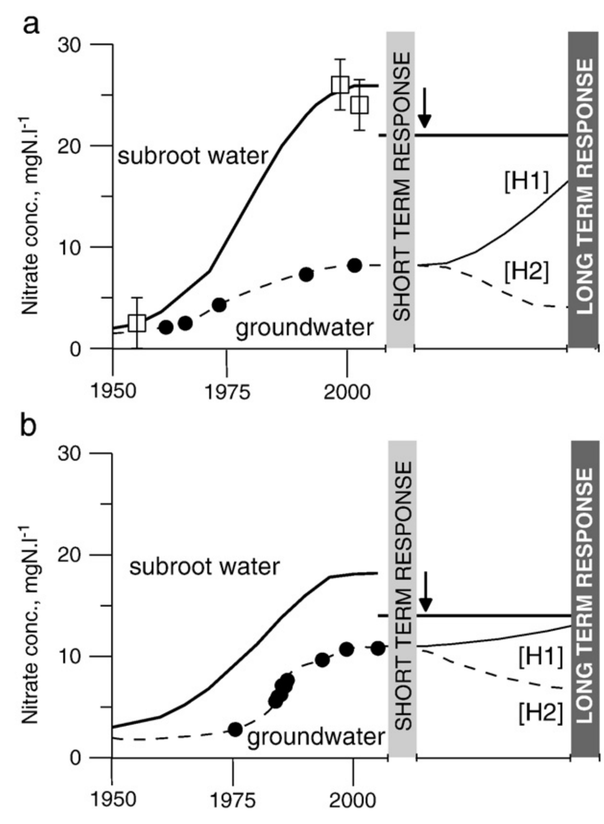

Fig. 3. Evolution of sub-root (modeled) and groundwater (literature data) nitrate concentrations from 1950 to the present, and hypothetical responses to a simulated change in short-term and long-term land use as assumed in the modeling approach. a) Case of the chalky Champagne district, sub-root nitrate concentrations are reflected in the lysimetric values collected by Ballif et al. (1996). The 1950–2000 trend for groundwater was extracted from Strebel et al. (1989). b) Case of the Brusselian sand aquifer present within the silty Belgium agricultural district. Groundwater concentrations are those extracted from De Becker et al. (1985). The arrow indicates the sub-root nitrate concentrations according to the short-term assumption whereas the two hypothetical long-term trends in groundwater concentration (H1, H2) are intentionally without time annotation as they represent a theoretical (no additional change in land-based anthropogenic pressure) evolution.

the Scheldt, and Beaudoin et al. (2005) in northern France. The latter authors suggested a mean value for nitrate reduction of 23%, without a significant influence of soil properties. A similar scenario was also offered for the entire Seine basin (Viennot et al., 2007), with nitrate sub-root reduction ranging from 5 to 35%. It included regionalization of the crop model STICS (Brisson et al., 2003) at the scale of homogeneous agricultural areas, similarly considered by the Seneque–Riverstrahler approach. Beaudoin et al. (2004) also suggested that the introduction of intermediary catch crops is generally followed by a decrease in the use of nitrogen fertilizers.

Based on these findings, consistent sub-root nutrient concentrations resulting from NTWCs implementation were defined and are synthesized in Table 2. The common representation of agricultural landscape proposed by Mignolet et al. (2007) for the Seine basin allows the assessment of NTWCs efficiency to be derived, as related by Viennot et al. (2007). Here, these values have been extended to the Somme and Scheldt basins with respect to the characterization of agricultural practices and compared with local/experimental results when available.

By limiting the period of uncovered soil, the introduction of catch crops might also limit the amount of suspended solids (SSs) and associated particulate phosphorus (TIP) exported to the river from erosion of crop areas (Némery et al., 2005). Since the effect of winter crops on TIP and SS is poorly documented, we have arbitrarily reduced both SS surface runoff concentrations and associated TIP content by the same factor as for nitrate.

2.4.3. Limiting fodder crops to support extensive cattle farming

The return to more extensive cattle-farming practices is another agricultural incentive measure envisaged in the framework of agro-environmental policies (regulation 2078/92/EEC), as this approach represents less yield-oriented and more environmentally concerned management. Our third scenario introduces a modification of the agricultural landscape by the substitution of fodder corn crops with grazed meadows.

Vertes et al. (2002) showed that nitrogen leaching from grazed meadows can be estimated from cattle density. Here, we used their empirical relationship and increased the area of grazed meadows at the expense of fodder crops, keeping cattle number unchanged. The regionalization of this measure lies in the downscaling of the agronomical district scale applied by the French and the Belgium agricultural census to the elementary watershed units of the Riverstrahler model. The calculated corresponding changes in nitrate sub-root concentration are shown in Table 2.

2.4.4. Short- and long-term assessment of agro-environmental measures

Due to the inertia of groundwater reservoirs, the effect of any changes in agricultural practices affecting sub-root water concentrations on the quality of the surface runoff component of river discharge will be observed rapidly, while the quality of base flow will be only affected after a significant delay, as illustrated in Fig. 3. The short-term assessment of agro-environmental measures is therefore based on the change of surface runoff composition only. For assessing the effect over longer periods, a hypothesis describing the response of the aquifers is required. The Riverstrahler model is not able to simulate the infiltration and circulation of phreatic water fluxes and their responses to any change in sub-root water concentrations. Therefore, we considered two hypotheses (H1 and H2) to explain the present substantial difference between sub-root water and groundwater nitrate concentrations. The first hypothesis (H1) assumes that the difference between surface water and groundwater concentrations is only due to the infiltration time, which induces a delay in the response of aquifers. Thus, regardless of the forthcoming evolution of aquifers in response to recent changes in agricultural practices, it will result in a long-term equilibrium between surface water and groundwater concentrations. The second hypothesis (H2) assumes that retention/elimination processes occur during nitrate transfer from the sub-root zone to the aquifer, as suggested by Billy et al. (in press). Assuming that these processes remain identical, it can be estimated that the long-term base-flow component of a river discharge will be reduced similarly to the surface runoff, by keeping a constant retention/elimination rate. The truth probably lies somewhere between these two hypothetical long-term trends, which therefore describe the possible long-term river water quality that can be expected (Fig. 3).

3. Results and discussion

In this section, we first calculate the effect on nutrient transfer of the different measures considered above, when implemented separately, compared with the 'present' reference situation. We then combine these

Table 2
The implementation of agronomical scenarios. The typology of agricultural district was derived from Mignolet et al. (2007) and extended to the Seine, Somme and Scheldt boundaries (Thieu et al., 2009). Nitrate concentrations refer to superficial water under the sub-root zone and do not include riparian retention/elimination.

	Reference	Extensive farming		Nitrogen catch crop		
Arable land	NO3, mg N/l (sub-root)	NO3, mg N/l (sub-root)	Fodder, % (arable converted)	NO3, mg N/l (sub-root)	Fertilizers, % (reduction)	NTWCs, % (reduction)
Agriculture parisienne	21	21	<1	12	−10	−35
Argonne bassigny	14	13	6	10	−5	−25
Brie beauce	18	18	<1	12	−10	−25
Champagne crayeuse	26	26	<1	18	−5	−27
Champagne humide	14	14	3	8	−5	−35
Depression yonne	14	14	1	10	−10	−20
Limon riche	18	18	2	11	−10	−35
Morvan	14	14	1	8	−10	−35
Perche auge bray	26	25	10	17	−20	−20
Plateau jurassique	14	14	2	10	−5	−20
Plateau normand	26	25	7	19	−10	−20
Vignoble	4	4	1	4	−0	−5
Belgium agri.regions	21	21	9 to 20	14	−10	−25
Grass land	NO3, mg N/l (sub-root)	NO3, mg N/l (sub-root)	Fodder, % (arable converted)	NO3, mg N/l (sub-root)	Fertilizers, % (reduction)	NTWCs, % (reduction)
Sablo-limoneuse	3.9	1.7	−30	3.9	–	–
Limoneuse	2.8	2.0	−30	2.8	–	–
Sableuse	4.9	1.6	−33	4.9	–	–
Argonnne	2.8	0.5	−13	2.8	–	–
Morvan	2.1	0.3	−3	2.1	–	–
Normande	4.2	0.7	−20	4.2	–	–

different measures. The results in terms of nutrient inputs to the river system and export to the coastal zone are summarized in Fig. 4.

3.1. Nutrient budgets in the present state

3.1.1. Nitrogen sources

Regarding nitrogen emission, the Scheldt—with the highest population density and the lowest wastewater treatment efficiency—proportionally has the highest point-source inputs (1013 kg N km^{-2} $year^{-1}$ *vs.* only 553 kg N km^{-2} $year^{-1}$ for the Seine River). Within the Somme basin, urban emissions of nitrogen are comparatively low (186 kg N km^{-2} $year^{-1}$). The differences in agricultural diffuse sources of nitrogen are much smaller; those of the Scheldt are equivalent to those calculated for the Seine and Somme, ranging from 1583 to 5571 kg N km^{-2} $year^{-1}$ depending on wet or dry hydrological conditions, although the spatial distribution of these sources within the basins widely differs too.

3.1.2. Phosphorus sources

Point sources follow the population-density gradient, with mean emission values of 24, 72, and 101 kg P km^{-2} $year^{-1}$, respectively, for the Somme, Seine and Scheldt Rivers (100, 200, and 496 inhabitants per km^2, respectively). The contributions of point and non-point P sources differ according to hydrology, with, logically, a higher share of urban emissions during dry years (60, 54, and 61%, for the Seine, Somme, and Scheldt) than during wet years (44, 19, and 46% respectively).

3.1.3. N, P retention and transfer

Overall nitrogen retention (fraction of total inputs to the river system that are not delivered at the outlet) varies from 46 to 67% and is directly linked to the contribution of nitrogen diffuse sources and their elimination within the riparian area. Other in-stream or groundwater processes are of lower significance for the nitrogen budget. Phosphorus retention is lower, with an average of 25–46% for the three basins, and is essentially related to the deposition of phosphorus adsorbed on suspended matter or taken up by planktonic algae.

3.1.4. Nutrient export

Exported fluxes were calculated at the outlet of the three basins: at Caudebec for the Seine, Cambron for the Somme, and Doel for the Scheldt. With respect to dry and wet hydrological conditions, nitrogen fluxes were 1378–2311 kg N km^{-2} $year^{-1}$ for the Seine, 639–1829 kg N km^{-2} $year^{-1}$ for the Somme, and 985–1587 kg N km^{-2} $year^{-1}$ for the Scheldt. Phosphorus fluxes were in the range of 87–100 kg P km^{-2} $year^{-1}$ for the Seine, 31–61 kg P km^{-2} $year^{-1}$ for the Somme, and 74–96 kg P km^{-2} $year^{-1}$ for the Scheldt.

These figures highlight the importance of hydrological conditions that control the contributions of diffuse sources and at the same time alter the dynamic of in-stream processes, subsequently changing the ratios between nutrient sources and exports. The advantage of the Riverstrahler mechanistic approach lies in its ability to assess the responses of a river system to changes in point and non-point sources while taking into account the variability introduced by the hydrology.

3.2. Reduction of point sources by improved sanitation

3.2.1. Phosphorus inputs

In the Seine River, because of the large population concentrated within the huge conurbation of Paris, P point sources could be reduced by as much as 51% simply by implementing P treatment in the sewage

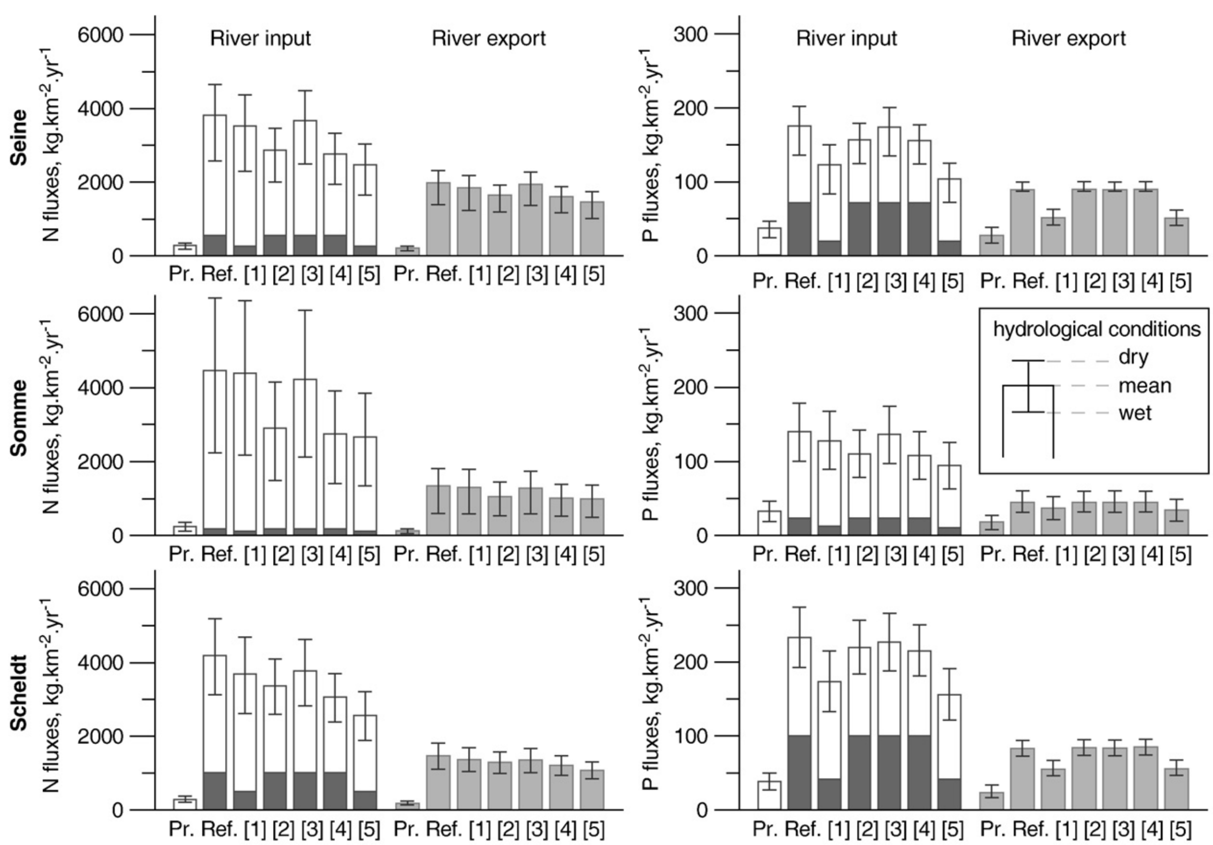

Fig. 4. Modification of nitrogen and phosphorus fluxes in response to several mitigation measures, as simulated by the Seneque-Riverstrahler model for the Seine, Somme, and Scheldt Rivers. River input fluxes are characterized by the contributions of point sources (dark grey) and diffuse sources (white), the latter being affected by hydrological variations, i.e., dry (negative error bar) and wet (positive error bar). River export (light grey) is the fluxes exported to the coastal zone. Pr: Pristine state; Ref: present situation; [1]: upgrading of wastewater treatment plants with a capacity greater than 100,000 inhabitant-equivalents for the Somme and 20,000 inhabitant-equivalents for the Seine and Scheldt Rivers; [2]: introduction of NTWCs and reduction of fertilizer use; [3]: extensive farming (substitution of fodder crops by grazed meadow); and [4]: [2] + [3] and [5]: [1] + [2] + [3].

plants with a treatment capacity of over 500,000 inhabitant-equivalents. In the Scheldt a similar scenario will reduce P from point sources by only 22%, while a 59% reduction could be achieved by applying measures addressed at effluent releases higher than 20,000 inhabitant-equivalents (Fig. 5). The Scheldt response is more linear, as the organization of population (and the corresponding release from wastewater treatment plants) within the basin is more homogeneously distributed.

In agreement with the European quality-standard level for P, it does not seem necessary to extend this improvement in P reduction to small treatment plants. Fig. 6 shows that the mean critically high summer concentrations of phosphate in the rivers could be maintained between “medium” (0.5–1 mg $PO_4 l^{-1}$) and “good” (<0.5 mg $PO_4 l^{-1}$) by limiting the emissions of WWTPs with treatment capacities greater than 100,000 inhabitant-equivalents. The same target could be reached downstream of the main urban areas of the Seine and Scheldt Rivers by implementing thresholds of 20,000 inhabitant-equivalents for improved P treatment in WWTPs. Indeed, beyond that threshold, the additional reduction of P fluxes would be hardly appreciable and the number of WWTPs that should be involved would be very high, necessarily making the cost-efficiency of this scenario quite low.

At total, the corresponding fluxes of phosphorus exported to the coastal zone could be decreased to 41–62, 21–52, and 46–67 kg P km^{-2} $year^{-1}$ for the Seine, Somme, and Scheldt Rivers, respectively. Such a scenario allows, according to hydrological conditions, to reach or at least to approach the 50% reduction of phosphorus loads targeted by OSPAR commission for the North Sea. Similar conclusions have been brought for the Elbe River (MONERIS) with the use of mean hydrological conditions (Hofmann et al., 2005).

3.2.2. Nitrogen inputs

Improved N removal by WWTPs over the same thresholds as defined for phosphorus lead to a decrease of N point sources of 59, 54, and 55%, respectively, for the Seine, Somme, and Scheldt drainage networks. However, because of the larger share of diffuse sources and the lower retention/elimination processes in the drainage network at lower nitrogen loading, the overall reduction of total nitrogen ranges from 2 to 7% at the outlets of the three basins. Similar prospective investigations carried out by De Wit and Bendoricchio (2001) highlighted that agriculture has became the major source of nitrogen contamination in many European rivers, when N point sources have

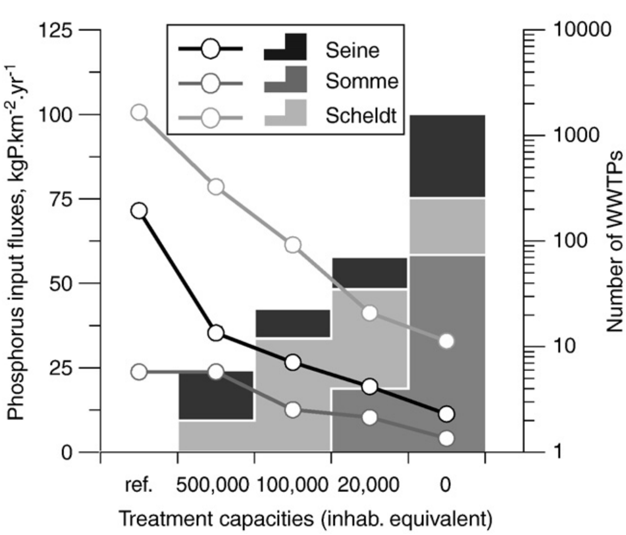

Fig. 5. Decrease in phosphorus emission levels within the three basins in response to progressive improvements in P treatments by wastewater treatment plants (WWTPs) with capacities of 500,000; 100,000; 20,000 inhabitant-equivalents and less. The numbers of WWTPs are indicated on the second *Y* axis (log scale). Ref: present situation.

been reduced. Our results obtained for the Seine, Somme and Scheldt rivers, confirm the limited impact on overall nitrogen loading of reducing point sources and emphasize the need to instead target diffuse sources to significantly reduce nitrogen emissions and exports.

3.3. Limiting N leaching through agro-environmental measures

3.3.1. Converting fodder corn crop areas

Extending cattle farming by converting fodder corn crop areas into grass land is mainly efficient within the Scheldt basin, where it leads to a reduction of nitrogen fluxes by 7.7–8.6% at the river outlet. This result reflects the importance of cattle farming and the proportion of arable land devoted to fodder corn crop, presently reaching up to 14% of arable land in the Scheldt basin. By contrast, in the Seine and the Somme basins only 2–4% of the arable land is likewise concerned, such that a nitrogen reduction at their outlets of only 1.9–3.7% could be expected following a conversion to cattle farming.

3.3.2. Implementation of NTWCs and reduction of nitrogen-based fertilizer use

The more widespread use of catch crops in cultural successions appears to be the most efficient option to regulate nitrogen export fluxes, with a nitrogen reduction of at least 10 % at the Scheldt outlet and as much as 17% at the Seine and 21% at the Somme outlet, the latter being largely oriented toward intensive cereal crops.

3.3.3. Combined effect

The increase in pasture and concomitant decrease in arable land is often presented as an efficient but not sufficient land-use management option (Volk et al., 2008), which must therefore be associated with the cultivation of intercrops (Krause et al., 2008). We have therefore combined the two previous scenarios as a single “land use and agricultural practices management measure,” which led to a nitrogen reduction ranging of 15 and 19% for the Seine, 14 and 23% for the Scheldt, and 16 and 23% for the Somme, during, respectively, dry and wet hydrological conditions.

3.3.4. Long-term impacts of the scenarios

As a simple approach to the long-term groundwater response to these scenarios, the two hypotheses explained above, were considered to predict long-term N fluxes (Fig. 7). In accordance with the H1 hypothesis (balance of surface water and groundwater concentrations), N fluxes increase from 0.5 to 4% with respect to the short-term effect of the combined agricultural measures. In the scope of the H2 hypothesis (i.e., maintaining a constant apparent denitrification rate in groundwater), a decrease from 3.5 to 22.3% with respect to the short-term effect is predicted over the long term. The considerable divergence between the H1 and H2 responses for the Seine and the Somme Rivers is explained by the large difference between nitrogen surface water and groundwater concentrations (Fig. 3a) whereas within the Scheldt basin surface water and groundwater N concentrations are more similar (Fig. 3b), resulting in a smaller difference between the predictions of the H1 and H2 hypotheses and no significant difference between the long- and short-term effects of agro-environmental measures. For the three rivers, simulation of the long-term variability of nitrogen fluxes remains less important than the influence of hydrological conditions (Fig. 7).

3.4. Imbalance of riverine fluxes

3.4.1. Pristine state and present references

Between pristine and the present conditions, the amount of silica delivered to surface waters has remained relatively unchanged across the three basins. Indeed, the dissolved form of SiO_2 is mainly related to rock weathering whereas point sources of silica arising from urban runoff, with the consumption of food and the use of detergent (Sferratore et al., 2006), account for only 1–8% of total silica input

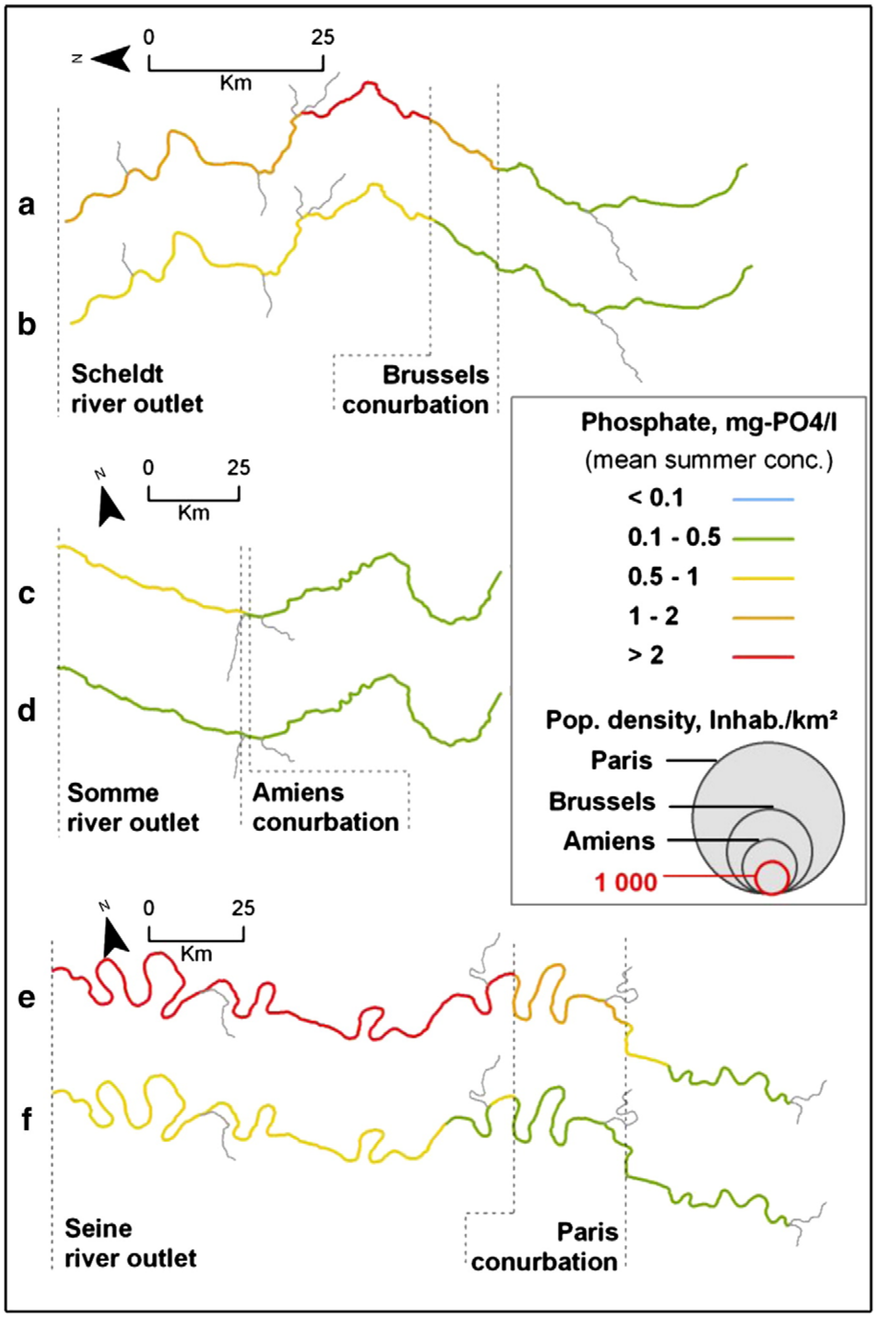

Fig. 6. Distribution of mean summer concentrations of phosphate as calculated by the Riverstrahler model for sanitation state and hydrological conditions defined for the year 2000 for the downstream portions of the Seine (e), Somme (c) and Scheldt (a) rivers. Improved sanitation situations refer to the upgrading of P treatment by WWTPs with a treatment capacity of above 100,000 inhabitant-equivalents for the Somme River (d) and 20,000 inhabitant-equivalents for the Seine (f) and Scheldt (b) rivers. River sections directly impacted by urban releases are delineated with dotted lines. Acceptable quality range from 0.5 to 1 mg $P0_4 l^{-1}$ according to the French Water Agency classification, whereas 0.1–0.5 mg $P0_4 l^{-1}$ is required for good quality status by both French and European regulation (circular DCE 2005/12).

(Thieu et al., 2009). On the other hand, artificial regulation and damming have contributed to increased silica trapping. The modeled slight effect has been small increases in the amounts of silica exported at the outlets of the three basins compared to pristine conditions, ranging from 1 to 7% according to wet or dry conditions. Among the scenarios, the impact on silica emission is limited to a decrease in the particulate biogenic form of silica (phytoliths) concomitant with a decrease of uncovered soil erosion. However, as particulate silica was estimated by Garnier et al. (2002b) to comprise less than 10% of silica input fluxes, silica emissions from watersheds remain almost unchanged and could be used as a reference with respect to the N and P fluxes, which are altered to a greater extent.

The Redfield molar C:N:P:Si ratio 106:16:1:20 (Redfield et al., 1963) is commonly used to characterize the growth of marine algae, including diatoms. The pristine state simulated by the Seneque–Riverstrahler model shows well-balanced nutrient riverine fluxes, with the N:P ratio ranging from 14 to 17 (Table 3), while high Si:N and Si:P ratios indicate nitrogen and phosphorus limitation of diatoms. By contrast, in the present situation, the availability of silica limits the production of siliceous algae, while N and P are in large excess. This promotes the development of non-siliceous algae. The elevated N:P ratio (here ranging from 34 to 66) indicates P limitation and is known to favour shifts in phytoplankton communities and/or the development of harmful algal blooms (Skogen et al., 2004; Cugier et al., 2005).

3.4.2. Combined nutrient mitigation scenario

In view of the present imbalance of nutrients delivered to the coastal zone, it is clear that measures affecting both phosphorus and nitrogen point and non-point sources have to be combined in order to improve the overall health of the coastal area. In the following, we therefore consider a coupled scenario that combines agro-environmental measures (decreased use of fertilizers, introduction of nitrogen catch crops, extension of grazed meadows) with improved nitrogen and phosphorus removal in WWTPs (down to a capacity of 100,000 inhabitant-equivalents for the Somme and 20,000 inhabitant-equivalents for the Seine and Scheldt basins). The results are presented in Table 3 for both the short and the long term (H1 and H2).

Except for the Somme system, in which phosphorus *vs.* silica fluxes are already well-balanced, this coupled scenario allows the return of the Si:P ratio at the outlets of the three river basins to a value close to that of the Redfield ratio (20), thus limiting an excess of phosphorus over silica. This drastic reduction in phosphorus is also required for limiting PO_4 contamination downstream of the main urban areas during the summer (Fig. 6). However, it contributes to dramatically increasing the already too high N:P ratio, despite the implementation of agro-environmental measures aimed at mitigating nitrogen fluxes.

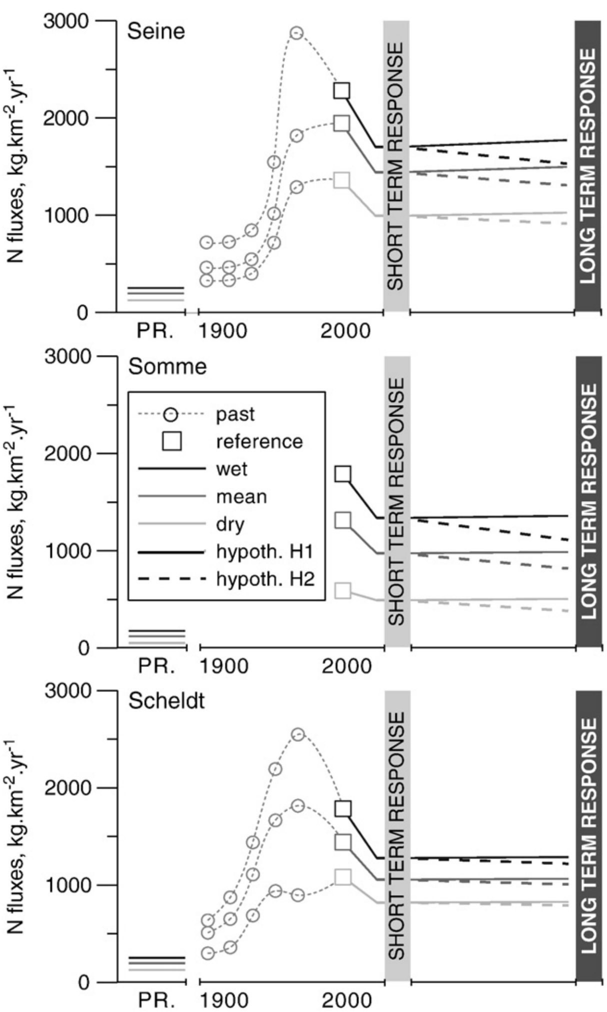

Fig. 7. Impact of combined agricultural mitigation measures on nitrogen export fluxes for dry, mean, and wet years (light grey to dark grey). Historical (1950–1990) data were extracted from Billen et al. (2007) for the Seine, and Billen et al. (2005) for the Scheldt. Theoretical long-term responses follow two hypothetical trends (see text). H1 (solid line) supports the long-term balance of groundwater concentration with sub-root concentration, while H2 (dotted line) supports a constant denitrification term between sub-root and groundwater concentrations.

With silica fluxes as a reference, the required amount of nitrogen delivered should not exceed 200–450 kg N^{-2} $year^{-1}$ according to the hydrology, while the present range is an order of magnitude higher. The combination of measures for reducing diffuse and point sources allows a short-term 32–39% reduction of the nitrogen excess over silica. Long-term assessment shows exported fluxes of 900–1200, 400–1500, and 850–1450 kg N km^{-2} $year^{-1}$ for, respectively, the Seine, Somme, and Scheldt (i.e., an 18–50% reduction of the nitrogen excess over silica). In any case, these reductions are clearly insufficient to restore a balanced N:P ratio, and a large excess of nitrogen with respect to the requirements of phytoplankton will continue to be delivered to the coastal sea.

Consequently, the substantial excess of nitrogen evidences a need for additional measures to control nitrogen sources from agriculture. The same conclusion is reached by Kersebaum et al. (2003) who applied a nutrient emission model (Stone model) to the Elbe watershed for assessing regional effect of a scenario of conversion of arable land into pasture scenarios. The limited impact on nitrogen contamination of agro-environmental measures promoted by the European Commission is also pointed out by the prospective work of nutrient transport models across Europe, as for example in the Po river (POLFLOW model: De Wit and Bendoricchio, 2001) where the required ‘optimal’ reduction of nitrogen is described as unrealistic. In the small Kervidy catchment (Brittany), the application of the INCA model (Durand, 2004) also showed the limited impact of moderate changes in management practices (combining reduced fertilization and catch crop introduction) and highlighted the need for alternative scenarios. These may combine management of nutrient sources with rehabilitation of retention areas in the river system to achieve higher attenuation of nutrient as suggested for the Danish Gjern river (TRANS model: Kronvang et al., 1999). It could also relies on a drastic change in agricultural practices going far beyond “rational agriculture” as the complete substitution of synthetic fertilizers by organic farming practices in a mixed farming context. The substantial decreases in nitrogen leaching obtained under less intensive farming practices (Drinkwater et al., 1998; Tilman, 1998) may be needed to reach a balanced N:P:Si ratio with respect to the expected reduction of phosphorus and the overall rather constant silica fluxes. However, a wide-ranging shift to organic farming over the entire regional watershed would imply deep modifications in the respective agricultural sectors and to our knowledge does not appear on the current political agenda.

The Seneque–Riverstrahler model does not construct alternative scenarios but instead provides a useful multi-element model, scalable from the local to the regional level, which is able to implement and test these scenarios once relevant information is provided. It thus emphasizes the need to up-scale both the results of field experiments and the generalized results from agronomical models to enable their transposition to process-based modeling of the drainage network.

4. Conclusions

The objective of this study was to investigate the contribution of three adjacent and highly contrasting drainage networks to the enrichment of the Southern Bight of the North Sea. In ignoring administrative boundaries, this study proposes an integrative framework for basin-scale management, and contributes to erect a description of aquatic system health possibly applicable to a European scale. While targets set for phosphorus by the EU-WFD and OSPAR recommendation are about to be achieved in the short term, EU directives (Nitrate and Urban Wastewater Treatment) for nitrogen will not be sufficient neither to avoid nitrate contamination of surface water nor to restore the balance of nitrogen export fluxes with respect to phosphorus and silica fluxes. This study also highlights the influence of hydrological constraints with a variability that may lead to minimize the influence of long-term groundwater changes (as for nitrogen) and emphasizes the need to incorporate the on-going work on climate change into the forecasting investigations on nutrient mobility within river basins.

Table 3
Pristine, present, and future nutrient fluxes calculated by the Riverstrahler model at the outlets of the Seine, Somme, and Scheldt Rivers, with the Redfield molar C:N:P:Si ratios (106:16:1:20), for wet and dry hydrological conditions.

	Seine		Somme		Scheldt	
	Dry	Wet	Dry	Wet	Dry	Wet
Pristine state						
N fluxes (kg km^{-2} year^{-1})	127	253	52	176	123	235
P fluxes (kg km^{-2} year^{-1})	17	38	8	27	16	33
Si fluxes (kg km^{-2} year^{-1})	598	1276	348	1151	751	1425
N:P:Si (molar ratio)	17:1:39	15:1:37	15:1:49	14:1:48	17:1:51	16:1:48
N surplus over Si (kg km^{-2} year^{-1})	−112	−257	−88	−285	−177	−335
P surplus over Si (kg km^{-2} year^{-1})	−16	−32	−11	−37	−25	−46
Present condition						
N fluxes (kg km^{-2} year^{-1})	1378	2311	639	1829	985	1568
P fluxes (kg km^{-2} year^{-1})	87	100	31	61	74	96
Si fluxes (kgkm^{-2} year^{-1})	525	1143	363	1140	639	1187
N:P:Si (molar ratio)	35:1:7	51:1:13	43:1:13	66:1:21	34:1:10	43:1:14
N surplus over Si (kg km^{-2} year^{-1})	1156	1833	448	1342	833	1331
P surplus over Si (kg km^{-2} year^{-1})	57	36	11	−2	37	29
Short-term coupled scenario						
N fluxes (kg km^{-2} year^{-1})	994	1702	492	1338	822	1278
P fluxes (kg km^{-2} year^{-1})	41	61	19	49	46	66
Si fluxes (kg km^{-2} year^{-1})	521	1123	355	1112	640	1158
N:P:Si (molar ratio)	54:1:14	62:1:20	57:1:21	61:1:25	40:1:15	43:1:19
N surplus over Si (kg km^{-2} year^{-1})	785	1253	350	893	565	815
P surplus over Si (kg km^{-2} year^{-1})	12	−1	−1	−13	10	2
Long-term coupled scenario [H1]	1041	1798	514	1379	839	1306
N fluxes (kg km^{-2} year^{-1})	41	62	19	49	46	67
P fluxes (kg km^{-2} year^{-1})	529	1139	360	1127	649	1174
Si fluxes (kg km^{-2} year^{-1})	1041	1798	514	1379	839	1306
N:P:Si (molar ratio)	56:1:14	65:1:20	59:1:21	62:1:25	40:1:15	43:1:19
N surplus over Si (kg km^{-2} year^{-1})	830	1342	370	928	579	837
P surplus over Si (kg km^{-2} year^{-1})	12	−1	−1	−13	11	2
Long-term coupled scenario [H2]						
N fluxes (kg km^{-2} year^{-1})	928	1552	387	1128	804	1237
P fluxes (kg km^{-2} year^{-1})	41	63	19	51	46	67
Si fluxes (kg km^{-2} year^{-1})	528	1141	361	1133	650	1175
N:P:Si (molar ratio)	50:1:14	55:1:20	45:1:21	49:1:25	39:1:16	41:1:19
N surplus over Si (kg km^{-2} year^{-1})	717	1095	243	675	545	767
P surplus over Si (kg km^{-2} year^{-1})	12	−1	−1	−13	11	2

Acknowledgements

This research was carried out as part of the Thresholds IP program which supported the Vincent Thieu's Ph-D. The work was also carried out in the framework of the French PIREN-Seine (CNRS, UMR Sisyphe) and TIMOTHY (Belgian Science Policy) programs. Thanks are due to Christiane Lancelot (ESA, Univ. of Brussels) and Marc Benoît (INRA-Mirecourt, France) for constructive discussions on the scenarios.

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