Modélisation spatialisée des flux de nutriments (N, P, Si) des bassins de la Seine, de la Somme et de l'Escaut : impact sur l'eutrophisation de la Manche et de la Mer du Nord

Vincent Thieu

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Vincent THIEU

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Impact sur l'eutrophisation de la Manche et de la Mer du Nord

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Devant le jury composé de :

M. Gilles BILLEN	Directeur de thèse
Mme. Josette GARNIER	Directrice de thèse
Mme. Christiane LANCELOT	Examinatrice
M. Wolfgang LUDWIG	Rapporteur
M. Patrick MEIRE	Rapporteur
Mme. Florentina MOATAR	Examinatrice
M. Alain SALIOT	Président

Université Pierre & Marie Curie - Paris 6
Bureau d'accueil, inscription des doctorants et base de données
Esc G, 2ème étage
15 rue de l'école de médecine
75270-PARIS CEDEX 06

Tél. Secrétariat : 01 42 34 68 35
Fax : 01 42 34 68 40
Tél. pour les étudiants de A à EL : 01 42 34 69 54
Tél. pour les étudiants de EM à ME : 01 42 34 68 41
Tél. pour les étudiants de MF à Z : 01 42 34 68 51
E-mail : scolarite.doctorat@upmc.fr

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Vincent Thieu

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Résumé

Modélisation spatialisée des flux de nutriments (N, P, Si) des bassins de la Seine, de la Somme et de l'Escaut : impact sur l'eutrophisation de la Manche et de la Mer du Nord

Vincent THIEU

Les apports fluviaux en éléments nutritifs comme l'azote (N) le phosphore (P) et la Silice (Si) sont indispensables au fonctionnement des écosystèmes côtiers et en conditionnent la productivité. Cependant, l'intensité et le déséquilibre des flux de nutriments peuvent conduire à l'eutrophisation de ces zones de transition marines. En Manche – Mer du Nord cet enrichissement excessif se caractérise par des blooms phytoplanctoniques qui apparaissent sous l'influence des apports de la Seine, de la Somme et de l'Escaut. Ces trois hydrosystèmes, qui diffèrent largement par leur morphologie, leurs débits mais également les pressions anthropiques qui s'y exercent, offrent un cas d'étude exemplaire pour analyser les flux d'éléments nutritifs dans les bassins versants et leurs impacts à la zone côtière.

L'objectif de cette thèse a été d'implémenter, pour ce continuum aquatique drainant plus de 100 000 km2 de surface continentale, une modélisation déterministe du fonctionnement des hydrosystèmes, s'attachant à décrire l'ensemble des processus microscopiques impliqués dans le transport, la rétention et la transformation des éléments N, P et Si. Validé sur la période récente (1996-2001), le modèle Sénèque-Riverstrahler a permis un calcul détaillé des transferts de nutriments, nécessaire pour quantifier le déséquilibre des flux d'azote et de phosphore exportés à la zone côtière franco-belge, par rapport à la silice. Différents scenarios, incluant des mesures de gestion concrètes, du type de celles préconisées par la directive Cadre Européenne sur l'eau (DCE), mais également des prospectives plus théoriques sur long terme, ont été élaborées et simulées. L'évaluation de ces scénarios porte sur la possibilité de rééquilibrer les apports fluviaux, en se basant sur les rapports de Redfield, de limiter la contamination azotée et phosphatée dans les réseaux hydrographiques et de réduire l'ouverture des cycles biogéochimiques qui a prévalu depuis plus de 30 ans.

Dans le cadre d'une approche intégrée 'rivière – zone côtière', ces résultats ont été couplés avec un modèle du fonctionnement des écosystèmes marins côtiers, permettant ainsi de traduire l'impact des mesures implémentées à l'échelle des bassins versants, en terme de développement algal atteint dans la zone côtière.

Mots-clés : continuum aquatique, fonctionnement biogéochimique, scenarios, eutrophisation, Seine, Somme, Escaut, azote, phosphore, silice.

Abstract

Modeling nutrient fluxes (N, P, Si) in the Seine, Somme and Scheldt rivers basins: impact on the eutrophication of the Eastern Channel and Southern Bight of the North Sea

Vincent THIEU

The productivity of coastal ecosystems is linked to river nutrient sources, in particular nitrogen (N), phosphorous (P) and Silica (Si), three elements essential for their functioning. However, nutrient loading and their imbalance can result in the eutrophication of costal marine areas. In the continental coastal zone of the Channel and Southern Bight of the North Sea, enriched water inflows from the Seine, Somme and Scheldt rivers contribute to the massive development of harmful algae species. The adjacent and contrasted watersheds of these three rivers differ by their morphology, hydrology and anthropogenic activities, and offer a textbook example for in-depth analysis of nutrient fluxes in river basins and their impact at the coastal zone.

This PhD thesis aimed to implement, at the scale of the 100,000 km² watershed drained by the aquatic continuum of the three rivers, a determinist modeling approach describing all basic processes involved in the transport, the retention and the transformation of nutrient (N, P, Si). The Seneque-Riverstrahler model implemented for that purpose, has been first validated for the recent period (1996-2001) and then used to establish a detailed budget of nutrient transfers. This latter was required to quantify the fluxes of nitrogen and phosphorus delivered in excess over silica at the French and Belgian coastal zone. Several mitigation measures have been designed and tested, including operational management options requested by the European Water Framework Directive (Eu-WFD) as well as long term scenarios of nutrient reduction. Assessment of these scenarios explores the ability to balance riverine exports with respect to the Redfield ratios, to improve nitrogen and phosphorous contamination within the drainage networks, and to limit the opening of biogeochemical cycles that has prevailed for more than 30 years.

These results have been coupled with a marine model describing the ecological functioning coastal ecosystem, in the framework of an integrated "river – ocean" approach. The impact of the different watershed scenarios is thus finally expressed in terms of algal development at the costal zone

Key-words: aquatic continuum, biogeochemical functioning, scenarios, eutrophication, Seine, Somme, Scheldt, nitrogen, phosphorus, silica

Avant propos

Le mémoire de thèse présenté est composé de six articles rédigés en anglais et réunis en trois parties. Quatre d'entre eux sont d'ores et déjà acceptés, et les deux autres sont en cours de révision, dans des revues scientifiques internationales.

Après une introduction générale, et afin de mettre en perspective les travaux réalisés et de mieux appréhender mon cheminement scientifique au cours de ces trois années d'étude, chacune des trois parties est introduite par une synthèse rédigée en français.

Table des matières

Liste des figures xvii

Liste des tableaux xxi

Introduction générale 1

Sources et transfert des éléments nutritifs (N, P, Si) dans les réseaux hydrographiques 15

1 - Bilan des transferts à l'échelle des continuums fluviaux : Seine, Somme et Escaut 21

1.1. Introduction 23
1.2. Study site and modeling approach 24
1.2.1. The Riverstrahler approach 24
1.2.2. General description of the three basins 25
1.3. Towards a restitution of the 3S heterogeneity 26
1.3.1. Morphology of the drainage network and hydraulic works 26
1.3.2. Modeling the hydrology of the three basins 28
1.3.3. Land use and non-point sources of nutrients 29
1.3.4. Wastewater point sources 32
1.4. Simulated and observed surface water quality 34
1.5. Computation of N, P, and Si budgets 38
1.6. Conclusions 42

2 - Evaluation comparative du devenir de l'azote dans les basins versants 43

2.1. Introduction 45
2.2. Description of the three basins 45
2.3. The input-output budget of the watersheds 47
2.4. The SCOPE/NANI approach 51
2.5. The partitioned retention approach 53
2.6. Process-based models 54
2.7. Discussion 59
2.8. Conclusions 61

Analyse prospective d'une modification des usages dans les bassins versants 63

3 - Impact d'une réduction des apports fluviaux, sur les flux exportés à la zone côtière 69

3.1. Introduction 71
3.2. Methodology 72
3.2.1. Study area 72
3.2.2. The Seneque-Riverstrahler basin-scale approach 74
3.2.3. Definition of reference conditions 75
3.2.3.1. Hydrological reference 75
3.2.3.2. Pristine state 76
3.2.3.3. The present impacted state 76

3.2.4. Scenarios and regulation options 77
3.2.4.1. Improvement of wastewater treatments 77
3.2.4.2. Decreasing nitrogen leaching from agricultural land 78
3.2.4.3. Limiting fodder crops to support extensive cattle farming 79
3.2.4.4. Short- and long-term assessment of agro-environmental measures 80
3.3. Results and discussion 80
3.3.1. Nutrient budgets in the present state 80
3.3.1.1. Nitrogen sources 80
3.3.1.2. Phosphorus sources 81
3.3.1.3. N, P retention and transfer 82
3.3.1.4. Nutrient export 82
3.3.2. Reduction of point sources by improved sanitation 82
3.3.2.1. Phosphorus inputs 82
3.3.2.2. Nitrogen inputs 84
3.3.3. Limiting N leaching through agro-environmental measures 85
3.3.3.1. Converting fodder corn crop area 85
3.3.3.2. Implementation of NTWCs and reduction of nitrogen-based fertilizer use 85
3.3.3.3. Combined effect 85
3.3.3.4. Long-term impacts of the scenarios 85
3.3.4. Imbalance of riverine fluxes 87
3.3.4.1. Pristine state and present references 87
3.3.4.2. Combined nutrient mitigation scenario 87
3.4. Conclusions 90

4 - Evaluation théorique d'une généralisation de l'agriculture biologique 91

4.1. Introduction 93
4.2. Characterising the current N-status of the three watersheds 94
4.2.1. Auto-heterotrophy status of the watershed 94
4.2.2. Deterioration of freshwater quality 95
4.2.3. Balance of riverine fluxes: Redfield ratios and N-ICEP indicator 95
4.3. Construction of an idealised OA scenario 96
4.3.1. Agronomical constraints 96
4.3.2. N-leaching under OA 98
4.3.3. Other constraints to the OA scenario 100
4.4. Results 100
4.4.1. Reducing the gap between autotrophy and heterotrophy 101
4.4.2. Improving the quality of the drainage network 103
4.4.3. Limiting the potential for coastal eutrophication 105
4.5. Discussion 106

5 - Application des scénarios tendanciels du Millenium Ecosystem Assessment 109

5.1. Introduction 111
5.2. Study area 113
5.3. Comparison of the Global NEWS and Riverstrahler models 114
5.3.1. *NEWS* 2 watershed models 114
5.3.2. Riverstrahler drainage-network model 115
5.3.3. Upscaling the spatial resolution of the drainage network 116
5.4. Consistency of high- and low-resolution input data 118
5.5. Downscaling global input data 119
5.5.1. Hydrology and temperature 120
5.5.2. Diffuse sources to the drainage network 121
5.5.3. Point sources to the drainage network 121
5.6. Deterministic approach of nutrient transfer based on downscaled global data 124
5.6.1. Validation for the period 1970–2000 124
5.6.2. Sub-regional analysis of the Millennium Ecosystem Assessment scenarios 125
5.6.3. Integrating the potential impact of the inner basin dynamics 128
5.7. Discussion and conclusions 129

Approche intégrée du continuum rivière – zone côtière **131**

6 - Evaluation écologique et économique d'une réduction des développements algaux en Mer du Nord **135**

6.1. Introduction 137
6.2. Material and methods 139
6.2.1. The SENEQUE/RIVERSTRAHLER-MIRO (SR-MIRO) model 139
6.2.2. SR-MIRO simulations 140
6.2.2.1. References 140
6.2.2.2. Nutrient reduction short-term scenarios 140
6.2.2.3. Nutrient reduction long-term scenarios 141
6.2.3. Cost valuation of the nutrient reduction scenarios 142
6.2.3.1. WWT improvement 142
6.2.3.2. Agro-environmental measures 143
6.3. Results 144
6.3.1. SR-MIRO validation 144
6.3.2. Nutrient reduction at the 3S river outlets 145
6.3.3. Nutrient status of the eutrophied BCZ 147
6.3.4. *Phaeocystis* colony magnitude and duration 149
6.3.5. BCZ ecosystem 'health' 151
6.3.6. Cost assessment and effectiveness 153
6.4. Discussion 156
6.4.1. Sensitivity to hydro-climatic factors 156
6.4.2. Relevance of the interdisciplinary approach 157

Conclusion générale et perspectives **161**

Bibliographie **167**

Annexe **183**

Liste des figures

Figure 0.1 : Représentation schématique du fonctionnement auto- ou hétérotrophe d'un écosystème 2

Figure 0.2 : conséquence de l'enrichissement progressif d'un écosytème aquatique. D'après Prepas and Charette (2003).. 5

Figure 0.3 : Représentation schématique du modèle RIVE des processus biogéochimiques dans les réseaux hydrographiques (d'après Thouvenot et al., 2007). 11

Figure I.1 : Carte synthétique des usages dominants par bassins versant élémentaires, étendue aux bassins de la Seine, de la Somme et de Escaut. 17

Figure I.2 : Bilan qualitatif et quantitatif des apports d'azote et de phosphore en lien avec la distribution des territoires agricoles et les densités de population... 19

Figure. I.3 : Apports nets d'azote anthropique aux bassins versants pour l'année 2000..................................... 19

Figure 1-1: (a) General map of the Seine, Somme, and Scheldt Rivers, indicating the main tributaries (blue) and urban centers (red). Population density (inhabitants.km^{-2}) is symbolized in gray graduation levels. (b) Homogeneous agricultural zones used for a spatial segmentation of arable land and grassland. 24

Figure 1-2: Specific runoff calculated (gray line) by the Hydrostrahler model and the observed value (dark circle). The data covered the period between 1996 and 2002 and was obtained at key stations of the Seine (Caudebec), Somme (Cambron) and Scheldt (downstream part of the Dendre affluent). 29

Figure 1-3: Results of simulations (line) and observations (circle) at the downstream station at Caudebec (Seine), Cambron (Somme), and Doel (Scheldt) in the dry year of 1996 (black), the hydrologically medium year 2000 (dark gray) and the wet year 2001 (light gray)... 35

Figure 1-4: Comparison of the simulations (line) made by the Seneque/Riverstrahler model and the observations (circle) along longitudinal profiles in 1996 (black), 2000 (dark gray), and 2001 (light gray). Downstream part of the Seine (left), main axis from the upstream part to the outlet of the Somme (middle), and the Senne affluent of the Scheldt river (right) are shown. 36

Figure 1-5: Comparison between model results (calculated) and data-based estimated fluxes of nutrients (observed) at several downstream stations of the Seine (square), Somme (circle), and Scheldt (triangle) drainage networks, in the years 1996 (black), 2000 (dark gray), and 2001 (light gray). 37

Figure 1-6: Distribution of ammonium concentrations over the three drainage networks beginning in September 2000, as calculated by the Seneque/Riverstrahler model.. 38

Figure 2-1: (a.) General map of the Seine, Somme en Scheldt basins and drainage network.(b.) Population density (data by cantons (France) and communes (Belgium) from the national statistics census).(c.) Land use (Corine Land Cover 2000, available on http://dataservice.eea.europa.eu/dataservice/). (d.) Livestock density (data by cantons from Agreste 2000 for France and by communes from INS for Belgium)......... 46

Figure 2-2: Conceptual representation of nitrogen fluxes through a watershed.. 47

Figure 2-3: Distribution of total yearly atmospheric deposition of oxidized nitrogen ($kgN/km^2/yr$) in the Seine, Somme and Scheldt watershed, according to EMEP (2000). 49

Figure 2-4: Distribution of the autotrophy: heterotrophy ratio (both calculated in kgN.km^{-2}.yr^{-1} at the scale of the French cantons and Belgian communes) in the Seine, Somme and Scheldt watersheds. Autotrophic areas (local N crop production > organic N consumption) appears green, while heterotrophic areas are red or yellow. 49

Figure 2-5: Past trends of the autotrophic/heterotrophic character of the Seine, Somme and Scheldt watershed, showing the opposite historical evolution of the 3 basins since the XXth century (historical data from Billen et al., 2008 for the Seine; from Billen et al, 1985 and De Becker, 1986 for the Scheldt). 50

Figure 2-6: (a) Specific N flux delivered at the outlet of sub-basins of the Seine, Somme and Scheldt drainage network as a function of the net anthropogenic nitrogen inputs (NANI, kgN.km^{-2}.yr^{-1}) in their watershed. The solid line represents the relationship proposed by Howarth et al. (2006). The dotted lines respectively represent N retention of 30 and 80% of NANI. The best fit linear regression line gives the relationship: delivery = 834 $kgN.km^{-2}.yr^{-1}$ + 0.16 NANI (r^2=0.38)(circle: Seine; triangle: Somme, square: Scheldt).(b) Apparent retention of N in the watersheds, expressed as a fraction of NANI, in function of specific runoff ($mm.yr^{-1}$). The upper straight line is the linear relationship proposed by Boyer et al. (2006), while the lower curve is a best fit quadratic function through all the data (retention = 0.51 + 0.002. runoff - 0.000004. runoff ^{2} ; r^2=0.6) 52

Figure 2-7: In-stream retention calculated according to the Sparrow-like approach in a number of sub-basins of the Seine, Somme and Scheldt watersheds, as a function of the basin area. (circle: Seine; triangle: Somme, square: Scheldt) 54

Figure 2-8: Budget of nitrogen transfers and transformations in the Seine, Somme and Scheldt drainage networks as calculated by the Seneque/Riverstrahler model for recent typical wet (2001) and dry (1996) hydrological conditions (results from Thieu et al., 2009).The main nitrogen fluxes to the terrestrial watershed (see Table 2-1) are also shown for comparison. 59

Figure II-1 : Impacts des scénarios analysés dans les chapitres 3 à 5 sur les quantités d'azote exportées à la zone côtière. Situation de référence, voir chapitre 1. econstitution historique pour la Seine et l'Escaut, voir Billen et al., (Billen et al., 2007b; 2005). 65

Figure 3-1: 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).... 73

Figure 3-2: 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. 75

Figure 3-3: 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. 77

Figure 3-4: 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); [4]: [2]+ [3] and [5]: [1] + [2]+ [3]. 81

Figure 3-5: 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. 83

Figure 3-6: 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. Rivers sections directly impacted by urban releases are delineated with dotted lines. Acceptable quality range from 0.5 and 1 mg $P04.l^{-1}$ according tp the French Water Agency classification, whereas 0.1 to 0.5 mg $P04.l^{-1}$ is required for good quality status by both French and European regulation (circular DCE 2005/12).......... 84

Figure 3-7: 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 extract 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. 86

Figure 4-1: Circulation of nitrogen in an idealised organic agriculture scenario. 97

Figure 4-2: Relationship observed between mean cereal crop yield and total (synthetic and organic) N fertilisation observed in the three basins in 2000 and 1955 (grey circles, as calculated from French and Belgian agricultural statistics). The mean yields reported for organic farms in the Centre region of France (Glachant 2009) are also plotted for comparison (dark grey squares). 97

Figure 4-3: N yield of cereal crop (a), Production of cereal crop (b) and Nitrate sub-root concentration as a function of organic fertilsers, here represented by cattle density (manure) and the proportion of non-fodder area devoted to legume crops (green manure) with no change in the total agricultural area of the Seine, Somme and Scheldt basins. 99

Figure 4-4: Autotrophy and heterotrophy of the three basins based on the assumption of a shift to organic agriculture (see Table 4-1) compared with the present and historical situations (Billen et al., 2009). 101

Figure 4-5: Distribution of yearly mean concentration of nitrogen over the three drainage networks for different scenarios, as calculated by the Riverstrahler model. 104

Figure 4-6**:** Mitigation of the Seine, Somme and Scheldt export fluxes. Calculation made by the Seneque-Riverstrahler model for mean (central symbol), dry (negative bar plot) and wet (positive bar plot) hydrological conditions. PRIS: pristine scenario, REF: reference year 2000 scenario, GAP: good agricultural practices scenario, BAU: business-as-usual scenario. 105

Figure 5-1: Map of the Seine, Somme, and Scheldt continental aquatic systems, as viewed by the Riverstrahler model (drainage network) and NEWS 2 models (basin scale). The grid size shown (0.5° × 0.5°) represents the elemental unit of the NEWS 2 models. 113

Figure 5-2: DIN and DIP flux exports to the sea, observed (dots) and calculated (line) as determined by the Riverstrahler model according to several representations of the drainage network for the year 2000: i) high, with the representation of each order-2 sub-basins, ii) intermediate, with the representation of each order-4 sub-basins, and iii) low, with the representation of the entire drainage network as a single basin. 117

Figure 5-3: Seasonal distribution of the annual runoff, and partitioning between surface and groundwater flow, based on hydrological modeling (rainfall-discharge) simulation, calibrated for the period 1996–2002.. 120

Figure 5-4: Distribution of drainage area and population equivalents by strahler order, as two synthetic indicators to describe the distribution of diffuse source and point source nutrient loading. This corresponds to the assessment of high-resolution information available for the year 2000. 123

Figure 5-5: Nutrient (TN and TP) fluxes exported to the North Sea by the Seine, Somme, and Scheldt Rivers systems, as observed and simulated by i) the NEWS 2 models on the basis of global inputs (black line); ii) the Riverstrahler model on the basis of downscaled global inputs (gray line); iii) the Riverstrahler model on the basis of a high-resolution database for two extreme hydrological conditions (Billen et al., 2007b; Billen et al., 2005) (gray area). TP fluxes could not be calculated for the year 1970 for the Somme (see also Table 5-3) due to the lack of correct global scale point sources data for this period. 123

Figure 5-6: Total N and total P deliveries calculated by the Riverstrahler model according to the MEA global inputs (GO: Global Orchestration; OS: Order of Strength; TG: Techno Garden; AM: Adapting Mosaic) downscaled to the Seine, Somme, and Scheldt drainage network. 127

Figure III-1 : Succession des productions diatomées et dino-flagellées en lien avec la disponibilité des sels nutritifs. Données extraites de (Billen and Garnier, 1997) 132

Figure III.2 : approche intégrée du continuum rivière – zone côtière, établie en collaboration avec les équipes de l'ULB-ESA et –CEESE. 134

Figure 6-1 RS-MIRO implementation geographical domain: the 3S watersheds, the coastal eastern Channel and Southern North Sea. Dotted line indicates the boundaries of the Exclusive Economic Zone (EEZ)....... 138

Figure 6-2 SR-MIRO simulations and observations of NO_3 (a), DSi (b), PO_4(c), Chl *a* (d) diatoms (e) and *Phaeocystis* colonies (f) in the BCZ for year 2000 144

Figure 6-3 SR-MIRO simulations of N (a) and P (b) Atlantic (light grey block) and Scheldt (black block) loads to the BCZ; of winter DIN (c) and PO_4 (d) concentration in the BCZ and their N:P (e) and Si:P (f) ratios: reference and mitigation scenarios. PR: Pristine, RF: 2000, WWT1: WWTP>500,000 IE; WWT2: WWTP>100,000 IE; WWT3: WWTP>20,000 IE; WWT4: WWTP>0 IE; AGR1: NTWC; AGR2: Extensive farming; AGR3: AGR1+AGR2; MIX: AGR3+WWT3; BEST: full biological agriculture + WWT3 with NO_3 diffuse at 6 (BEST1) and 3 (BEST2) mg N L^{-1} 147

Figure 6-4 SR-MIRO simulations of *Phaeocystis* maximum cells (a) and duration (b) obtained for the reference and mitigation scenarios. PR: Pristine, RF: 2000, WWT1: WWTP>500,000 IE; WWT2: WWTP>100,000 IE; WWT3: WWTP>20,000 IE; WWT4: WWTP>0 IE; AGR1: NTWC; AGR2: Extensive farming; AGR3: AGR1+AGR2; MIX: AGR3+WWT3; BEST: full biological agriculture + WWT3 with NO_3 diffuse at 6 (BEST1) and 3 (BEST2) mg N L^{-1} 150

Figure 6-5 SR-MIRO simulations of diatom and *Phaeocystis* colony annual production. All obtained for the reference and mitigation scenarios. PR: Pristine, RF: 2000, WWT1: WWTP>500,000 IE; WWT2: WWTP>100,000 IE; WWT3: WWTP>20,000 IE; WWT4: WWTP>0 IE; AGR1: NTWC; AGR2: Extensive farming; AGR3: AGR1+AGR2; MIX: AGR3+WWT3; BEST: full biological agriculture + WWT3 with NO*3* diffuse at 6 (BEST1) and 3 (BEST2) mg N L^{-1}........ 150

Figure 6-6 Relationship between the contribution of *Phaeocystis* colony production to annual primary production in the BCZ and the winter N:P ratios , all obtained from SR-MIRO simulations under the reference and mitigation scenarios. 152

Figure 6-7 Relations ship between cost effectiveness of the nutrient mitigation measure and the achieved nutrient reduction for phosphorus (a) and nitrogen (b) 152

Liste des tableaux

Table 0.1 **:** Caractéristiques des bassins versants Seine, Somme et Escaut 9

Table 1-1: Comparison of total and mean (*) morphological, demographic, and land-cover characteristics of the Seine, Somme, and Scheldt drainage networks according to stream order. 27

Table 1-2: Land-use types and corresponding nutrient concentrations in surface and base flows. 31

Table 1-3: Identification of the main water treatments for the three basins, effective treatment capacity as integrated by the Riverstrahler model, and corresponding per-capita values for the main components of the effluents (SM: suspended matter; OM: organic matter; NO_3: nitrate; NH_4: ammonium; PO_4: phosphate; DSi: dissolved silica; BSi: particulate biogenic silica). 33

Table 1-4: N, P, Si transfer budgets calculated by the Riverstrahler model and ratios of the nutrients delivered yearly at the outlets of the Seine, Somme, and Scheldt Rivers, for wet and dry conditions. Redfield ratios are in parentheses. 39

Table 2-1: Input/output Nitrogen budget of the Seine, Somme and Scheldt watersheds for the medium hydrological year 2000, compared to that of Europe as a whole (excluding former Soviet Union, van Egmond et al, 2002; unpublished data for riverine delivery), of the Mississipi (USA, Boyer et al., 2006) and the Red River (China and Vietnam, Le et al., 2005). 48

Table 2-2: Morphological characteristics of a number of sub-basins of various sizes selected within the Seine, Somme and Scheldt watersheds and associated N cycle 56

Table 2-3: Independent estimates of the intensity of various retention processes in the Seine, Somme and Scheldt watersheds (in $kgN.km^{-2}.yr^{-1}$) according to the NANI, the partitioned retention and the Seneque/Riverstrahler approaches. (Figures in brackets refers to estimations obtained by difference) 62

Table 3-1: 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. 73

Table 3-2: 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. 79

Table 3-3: 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. 88

Table 4-1: Estimation of autotrophy and heterotrophy of the three basins within the scenario of shifting to organic agriculture of the entire present agricultural area. The corresponding figures for the present situation (as described in Billen et al., 2009) are indicated for comparison. 102

Table 4-2: Human food production and consumption of the three basins in the organic agriculture scenario. The corresponding figures for the present situation (as deduced from agricultural statistics) are indicated for comparison. 102

Table 4-3: Calculated values of the indicator of coastal eutrophication potential (N-ICEP) for the different scenarios, under dry and wet conditions. 106

Table 5-1: State variables and processes taken into account by the Riverstrahler model. 116

Table 5-2: Detail of the high-resolution database used for the sub-regional assessment of the Seine, Somme, and Scheldt basins. 117

Table 5-3: Comparison of nutrient input to the Seine, Somme, and Scheldt Rivers on the basis of the high-resolution database integrated into the Riverstrahler models, and as provided by the NEWS 2 models after landscape retention for the years 1970 and 2000. 1970 inputs are derived from Billen et al., (2007b) for the Seine and from Billen et al., [2005] for the Schledt. For these two rivers, computation of diffuse sources is based on hydrological conditions derived from NEW2 (Scheldt) or highly similar (Seine: 173 mm/yr for the dry scenario in Billen et al., (2007b)). 119

Table 5-4: Transposition of the NEWS model variables following the requirement of the Riverstrahler model. The downscaling methodology is primarily based on the distribution by order observed throughout the high-resolution database. For (2), the transition from nutrient loads (tons/yr) to nutrient concentration, the annual runoff value (1) was used. Point sources (3) depend of population densities connected to sewage system, this driver is commonly described by the both models. 121

Table 5-5: Synthesis of the evolution of the main land-based drivers and sources according to the four Millennium Ecosystem Assessment scenarios: percentage values assessing change between 2000 and 2050. (*Diffuse-source values consider anthropogenic areas only; the "export" term includes crop export and animal grazing.) 126

Table 6-1: The integrated SR-MIRO approach : SENEQUE-RIVERSTRAHLER simulations of nutrient loads at the 3S river outlets: references and mitigations scenarios as used for the MIRO model. PR: Pristine, RF: 2000, WWT1: WWTP>500,000 IE; WWT2: WWTP>100,000 IE; WWT3: WWTP>20,000 IE; WWT4: WWTP > 0 IE; AGR1: NTWC; AGR2: Extensive farming; AGR3: AGR1+AGR2; MIX: AGR3+WWT3; BEST: full biological agriculture + WWT3 with NO_3 diffuse at 6 (BEST1) and 3 (BEST2) mg N L^{-1}.. 146

Table 6-2: SR-MIRO simulations scenarios: total cost and effectiveness of N and P reduction at the 3S outlet 154

Table 6-3: SR-MIRO scenarios: total cost and effectiveness of *Phaecystis* maximum and duration reduction in the BCZ. RF *Phaecystis* maximum= 45.8 10^6 cell L^{-1}; RF 'undesirable' bloom duration: 33 days 156

Table 6-4: Sensitivity analysis of RS-MIRO nutrient mitigation scenarios to hydro-climatic conditions 157

Introduction générale

L'étude des écosystèmes aquatiques s'initie le plus souvent de façon réactive, suite au constat d'un dysfonctionnement ou face à des manifestations visibles exprimées par ces systèmes en réponse à leurs altérations. L'emprise spatiale de ces symptômes a longtemps conditionné l'échelle de leur étude, ce qui a conduit à des expertises très sectorisées des milieux naturels.

Dans le cas des cours d'eau, cette approche a dû évoluer vers une représentation à plus grande échelle, capable d'inclure la dimension des altérations causées par l'activité de l'homme dans le bassin d'alimentation, et leurs répercutions dans le gradient amont-aval du réseau hydrographique. Le bassin versant s'est ainsi imposé comme une entité fonctionnelle pour aborder les problématiques liées à la ressource en eau. En transposant cette logique aux eaux côtières, où ces dysfonctionnements se manifestent sous des formes parfois dommageables, on comprend alors la nécessité d'une représentation encore supérieure, qui rassemble cette fois-ci l'ensemble des bassins versants contributifs d'une aire côtière marine à une échelle plurirégionale.

Paradoxalement, à cette nécessité d'intégrer des espaces d'étude plus étendus (géographiquement), l'avancée des connaissances scientifiques incite à analyser le fonctionnement de ces systèmes aquatiques à une échelle microscopique, celle des processus biologiques élémentaires, tout en gardant le lien avec les contraintes macroscopique auxquelles ces systèmes sont soumis. Les facteurs agissant sur les écosystèmes ont ainsi des résolutions variables et c'est cet assemblage d'échelles qu'il faut considérer pour appréhender le fonctionnement d'ensemble des réseaux hydrographiques et leur impact à la zone côtière.

Fonctionnement biogéochimique (écologique) des écosystèmes aquatiques

Le terme de *fonctionnement biogéochimique (écologique) d'un écosystème*, réfère à un ensemble d'interactions qui impliquent les constituants abiotiques du milieu et les organismes qui y vivent. Le fonctionnement d'un écosystème peut ainsi être caractérisé par la circulation des éléments biogènes en lien avec l'activité des organismes qui le composent, donc par les transferts de matière et d'énergie en son sein. Quelles que soient la nature et l'échelle de l'écosystème considéré, ces interactions sont essentiellement trophiques et se distinguent par des capacités à produire (*P : production, ou autotrophie*) et à consommer (*R : respiration, ou hétérotrophie*) de la matière organique et de l'oxygène (Odum, 1971). La façon dont s'équilibrent ces deux activités (Figure 0.1) permet alors de caractériser le *fonctionnement* d'un écosystème comme autotrophe (P/R>1) ou hétérotrophe (P/R<1).

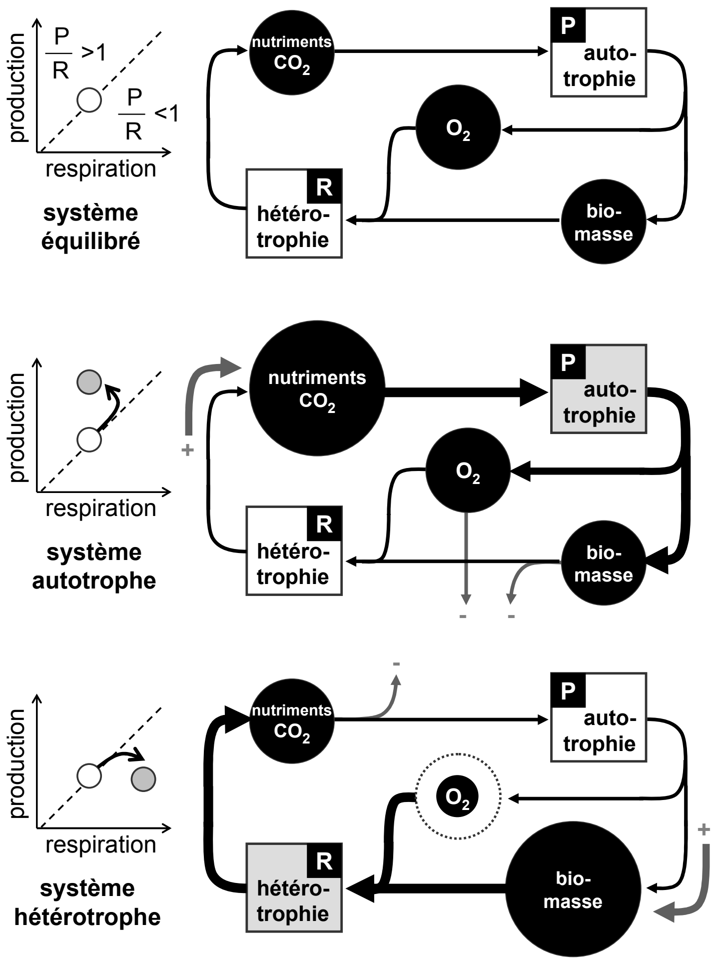

Figure 0.1 : Représentation schématique du fonctionnement auto- ou hétérotrophe d'un écosystème

Dans les écosystèmes aquatiques de surface, qu'ils soient lentiques (lacs, étang, ancien méandre) ou lotique (fleuves et rivière), la production primaire est assurée par l'activité photosynthétique du phytoplancton (microalgues) ou des macrophytes (macroalgues), qui élabore de la matière organique à partir d'énergie lumineuse et de CO_2. Néanmoins, leur croissance requiert d'autres éléments, comme le phosphore et l'azote qui interviennent dans la synthèse des protéines et acides nucléique, ou encore la silice indispensable pour les espèces à squelette siliceux comme les algues diatomées (Tréguer and Pondaven, 2000). Ce prélèvement élémentaire de nutriments dans le milieu, alimente le zooplancton (brouteur) puis les niveaux trophiques supérieurs, via la chaine trophique linéaire. La régénération de ces éléments nutritifs C, N, P, Si est assuré par l'activité microbienne, qui va dégrader certains organismes (lyse) et minéraliser la matière organique morte les rendant de nouveau disponible pour les producteurs primaires via la boucle microbienne (Azam et al., 1983). Ces interactions entre (micro)organismes producteurs, consommateurs, décomposeurs et minéralisateurs sont à la base de la circulation des éléments biogènes. On parle alors de *cycles biogéochimiques* pour décrire ces échanges de matière entre biotope et biocénose sous formes alternativement minérale et organique, dissoute et particulaire.

Le fonctionnement d'un écosystème aquatique, c'est-à-dire la manière dont s'équilibrent les activités autotrophes et hétérotrophes, est ainsi directement lié aux *cycles biogéochimiques* des nutriments.

Parmi les interactions complexes -biotiques et abiotiques- contrôlant l'intensité de la production primaire, la *disponibilité* des nutriments -azote, phosphore et silice- apparaît essentielle, en association avec les conditions d'éclairement, la stratification de la colonne d'eau et le temps de résidence (notion de susceptibilité NRC, 2000). L'azote est assimilable sous ses formes inorganiques, préférentiellement ammonium NH_4^+ et nitrate NO_3^-. Le phosphore est majoritairement disponible sous la forme d'ion phosphate (PO_4^{3-}), et la silice n'est prélevée que sous sa forme dissoute (H_4SiO_4). Le taux de consommation par le phytoplancton augmente alors linéairement avec la concentration des nutriments dans le milieu jusqu'à atteindre un plateau correspondant au taux maximum de croissance (cinétique de type Michaelis-Menten). Ainsi, des concentrations insuffisantes de nutriments peuvent contribuer à limiter la production primaire. Dans les cours d'eau, le phosphore est susceptible d'être limitant dans les secteurs amonts (cf. les travaux pionniers de Vollenweider, 1968) alors que la silice abondante dès les têtes de bassin pourra devenir limitante dans les parties plus aval (Garnier et al., 1995). L'azote, très abondant n'apparaît limitant que pour les hydrosystèmes marins (Menesguen, 1992). Le contrôle de la production côtière est nécessairement plus complexe, soumise à la fois aux apports terrigènes et marins.

Les besoins moyens en nutriments pour la croissance algale ont été établis il y a presque un demi-siècle par les travaux de Redfield et al., (1963), sur la base du rapport molaire des quantités de carbone, azote, phosphore et silice prélevées dans le milieu. Pour le phytoplancton d'eau douce, ce

rapport C:N:P:Si est de 106 :16 :1 :40, alors que dans les zones marines, les diatomées prélèvent moins de silice (Conley and Kilham, 1989) et portent ce rapport à 106 :16 :1 :20.
Au premier critère de *disponibilité* susceptible de conditionner l'intensité de la production algale, s'ajoute donc la notion d'*équilibre* entre nutriments qui va déterminer le type de développement, c'est-à-dire une production de diatomées ou non diatomées (Chlorophycées, flagellés, cyanobactéries) (Conley et al., 1993; Everbecq et al., 2001; Garnier et al., 1995; Turner et al., 1998b).

Déséquilibre et eutrophisation

Le terme d'*eutrophisation* fait initialement référence à l'enrichissement du milieu vers un état « eutrophe », c'est-à-dire riche en matière nutritive. La définition usuelle de ce terme intègre la notion d'excès et d'effets indésirables (OSPAR, 2005), désignant à la fois les causes et les conséquences de la fertilisation du milieu (Ménesguen et al., 2001). Les causes sont essentiellement liées à l'apport de nutriments dans des proportions excessives ou inadaptées à la croissance, ainsi qu'au déplacement de l'équilibre entre autotrophie et hétérotrophie traduisant alors un *dysfonctionnement* de l'écosystème.

Une des conséquences de cet enrichissement est l'accumulation de matière organique pouvant induire des déficits graves en oxygène, comme en 1982 où l'anoxie de la Baie de la Vilaine avait sévèrement détruit les communautés benthiques (Merceron, 1987), ou le Golfe du Mexique où les « dead zones », anoxiques, s'étendent d'années en années (Rabalais et al., 2002). L'augmentation de ces phénomènes d'hypoxie est observée partout dans le monde (Diaz and Rosenberg, 2008). Sans forcément atteindre des niveaux d'anoxie mortelle, l'eutrophisation se manifeste plus fréquemment par des proliférations algales phytoplanctoniques ou macrophytiques nuisibles et parfois toxiques pour les organismes supérieurs. Le développement de macroalgues est observé dans certaines zones côtières sous la forme de marées vertes (genre Ulva), et dans les cours d'eau amont où la dilution hydraulique ne permet que la croissance de végétaux fixés. Les proliférations massives de phytoplancton, également appelées *blooms* sont restreintes au secteur aval des grands fleuves (Garnier et al., 1995) ou aux bandes côtières. Dans ce dernier cas, les problèmes d'eutrophisation sont effectivement liés à un enrichissement excessif d'azote et de phosphore, mais également aux déséquilibres de ces apports par rapport à la silice, c'est-à-dire au besoin de la croissance diatomique (Billen and Garnier, 2007; Cugier et al., 2005; Officer and Ryther, 1980). Cette carence favorise alors l'abondance d'espèces non-siliceuses notamment les dinoflagellés, comme Dinophysis, espèce toxique, productrice de toxines diarrhéiques, et présente en été dans le panache de dilution de certains fleuves (Belin and Raffin, 1998; Cugier et al., 2005). Une autre manifestation du déséquilibre entre N, P, et Si, est l'accumulation importante de mousse sur le littoral nord atlantique (Manche et Mer du Nord), issue de la prolifération estivale de *Phaeocystis*, algue dinoflagellée non siliceuse (Lancelot, 1995; Lancelot et al., 1987).

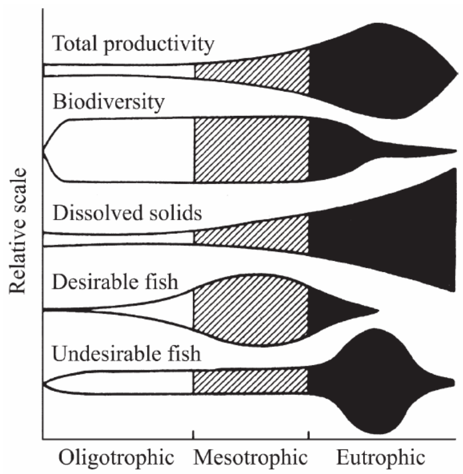

Figure 0.2 : conséquence de l'enrichissement progressif d'un écosytème aquatique. D'après Prepas and Charette (2003)

Les problèmes posés par l'eutrophisation marine côtière, sous des manifestations très diverses (Figure 0.2), apparaissent donc le plus souvent liés au développement d'algues non siliceuses soutenu par des apports terrigènes de nutriments contenant un excès d'azote et de phosphore par rapport à la silice au regard de la stœchiométrie des besoins des diatomées définis par les rapport de Redfield (Billen and Garnier, 2007).

L'hydrosystème, un continuum d'écosystèmes aquatiques

Dans les paragraphes précédents, il a été implicitement admis que la disponibilité naturelle des nutriments n'est pas la même pour tous les écosystèmes aquatiques, et que tous ne favorisent pas les mêmes types de développement algaux. Outre l'abondance de certains éléments nutritifs, la position amont-aval et les conditions hydrauliques, ont été évoquées comme discriminant le fonctionnement de ces écosystèmes et l'apparition d'épisodes eutrophes. Il est donc possible de considérer l'*hydrosystème* comme un *continuum d'écosystèmes aquatiques* en interaction les uns avec les autres, et dont la variabilité du fonctionnement est contrôlée par des contraintes environnementales qui évoluent dans l'espace et dans le temps. Le fonctionnement potentiel de chacun des écosystèmes de ce continuum est identique, mais la modulation de son expression dépend des conditions hydrauliques, climatiques et physico-chimiques auxquelles il est soumis.

Quatre dimensions sont évoquées pour décrire la dynamique des hydrosystèmes (Ward, 1989) et leurs différentiations spatiales. La dimension longitudinale souligne l'évolution des facteurs biotiques et abiotiques dans un gradient amont-aval. Initialement proposé par Vannote, (1980) le concept de continuum fluvial hiérarchise les réseaux hydrographiques et structure le fonctionnement des écosystèmes en accord avec les variations physiques et géomorphologiques s'établissant selon ce même gradient. La dimension latérale intègre les interactions avec les parties terrestres drainées et semi-terrestres comme les zones humides. La notion de connectivité hydrologique permet alors d'intégrer les échanges de matière et d'énergie entre le cours d'eau et ses parties riveraines (Amoros and Bornette, 2002). De même, les échanges s'opérant verticalement via les interstices saturés en eau des zones hypohréïques jusqu'aux aquifères (infiltration, exfiltration), jouent un rôle très important (Hynes, 1983). Enfin, la dimension temporelle intègre à la fois des cinétiques rapides entre biotopes et communautés vivantes, mais également des temps plus long en lien avec le transfert et la rétention des flux d'eau et de matière au travers les trois autres dimensions.

A l'échelle des hydrosystèmes, les écosystèmes aquatiques sont donc interconnectés, et interagissent avec des contraintes environnementales externes. La diversité des paysages terrestres qui constituent le bassin versant drainé par son hydrosystème conditionne largement les dynamiques qui se mettent en place dans ces quatre dimensions, et plus particulièrement l'approvisionnement en nutriments du réseau hydrographique.

Les cycles naturels N, P et Si

Ces changements d'échelle nous conduisent à prendre en compte l'activité des microorganismes assurant la circulation des éléments biogènes dans les écosystèmes aquatiques, et à caractériser un bassin versant et les activités dont il est le siège, et qui conditionnent les transferts de matière vers l'hydrosystème. Dans les systèmes naturels (ou faiblement anthropisés) ces nutriments proviennent principalement de la percolation de l'eau de pluie et de son interaction avec les sols du bassin ; on parle alors d'apports diffus.

La production de *silice* dans les bassins versants se fait essentiellement sous forme dissoute (H4SiO4), suite à l'altération de minéraux silicatés (feldspath) par l'acide carbonique présent dans l'eau de pluie. Cette altération est soumise d'une part à des facteurs climatiques (température, importance des précipitations), et potentiellement à la nature du couvert végétal (Humborg et al., 2004). Une fois dissoute, une partie de cette silice peut être utilisée par des végétaux terrestres (comme les graminées) puis réémis après décomposition sous forme de phytolithes. La proportion de ces apports particulaires biogéniques érodés reste cependant relativement faible (Garnier et al., 2005b; Sferratore, 2006). Les surfaces de bassins drainés et les apports diffus associés diminuent d'amont en aval et expliquent l'abondance naturelle de silice dans les premiers tributaires du réseau hydrographique.

Egalement issu de l'altération des roches, le *phosphore* s'absorbe rapidement au niveau des particules de sols. Il est soumis préférentiellement aux processus d'érosion, et les apports au cours d'eau sont essentiellement particulaires, sous forme de matières en suspension. Les organismes aquatiques incorporent uniquement le phosphore sous sa forme dissoute (orthophosphate), cette biodisponibilité dépend alors de l'équilibre absorption/désorption s'établissant avec les matières en suspension (Némery et al., 2005). En effet, le contenu en phosphore de la matière en suspension va s'équilibrer naturellement tout au long de son parcours d'amont en aval avec les concentrations en orthophosphate dans le milieu, et ce n'est qu'une fois en mer, milieu très appauvri en phosphore, que ce dernier se désorbera (Nemery and Garnier, 2007).

Contrairement au phosphore et à la silice, l'*azote* fait intervenir des processus biologiques, nombreux et complexes. La fixation de l'azote atmosphérique (N2) et la minéralisation de la matière organique (ammonification) vont produire de l'ammoniaque (NH3) et de l'ammonium (NH4+) en équilibre dans le milieu. Les activités nitrifiantes -nitrosation et nitratation- vont permettre la formation du de nitrate (NO3-). A ce stade, le retour à des formes d'azote réduites repose sur des processus i) d'assimilation aérobie, où l'azote est ensuite rapidement incorporé sous forme organique, ou ii) de dissimilation anaérobie avec la production d'ammonium. Les nitrates peuvent également être convertis sous formes gazeuses (N2O et N2) sous l'action de bactéries dénitrifiantes et dans des conditions favorables (anoxie, et présence de matière organique). L'ensemble de ces processus opère dans les parties superficielles du sol. Les quantités d'azote lessivé, principalement sous forme de nitrate, sont conditionnées par la nature du couvert végétal et par la présence de zones humides riveraines susceptibles de limiter ces « pertes » azotées sous forme dissoute.

L'activité humaine s'intègre au continuum aquatique

A cette circulation naturelle, étroitement fermée (encore présente dans les territoires forestiers), se sont superposées les activités humaines qui ont altéré les quantités et la nature des flux de nutriments parvenant aux réseaux hydrographiques (Billen et al., 1995). L'installation de l'homme sur les bassins versants s'est accompagnée d'une substitution des forêts, au profit de terres cultivées et d'une mise à nu régulière des sols, favorisant le lessivage de leur stock minéral entre chaque récolte.

Le retour à la terre des quantités produites et consommées localement dans un contexte d'agriculture traditionnelle, n'a pas résisté à la demande alimentaire croissante des pôles urbains (Billen et al., 2009a; Billen et al., 2007b). Il en résulte une perte de fertilité importante pour les sols de ces territoires ruraux, qui exportent des quantités importantes d'azote et de phosphore au travers de leur production agricole, sans pouvoir restituer les déchets de cette consommation. Ces derniers sont alors directement évacués vers les eaux de surface au niveau des zones de consommation urbaines, après traitement dans les stations d'épuration.

L'usage fait par l'homme des sols altère ainsi la circulation des éléments nutritifs, et on parle alors d'ouverture des cycles biogéochimiques. L'extraction minière de phosphore ou la découverte du procédé Haber-Bosch (1913) permettant de fixer industriellement l'azote atmosphérique, ont permis de compenser l'appauvrissement des sols par addition d'une fertilisation minérale. Cet apport de fertilisants de synthèse se substitue alors à la nécessité de maintenir le cycle des nutriments fermé pour les besoins même de la productivité agricole. Dés lors, l'augmentation considérable des rendements

agricoles s’est accompagnée d’une ouverture sans précédent du cycle des éléments biogènes, certains bassins exportant leur récolte bien au-delà de leurs limites topographiques, d’autres important en masse, des produits agricoles destinées à l’alimentation humaine ou animale, tous ayant recours à des fertilisants de synthèse produits industriellement.
L’urbanisation de certaines parties du territoire a par ailleurs conduit à la mise en place progressive d’un réseau de collecte des effluents, évacuant vers les cours d’eau les déchets domestiques et industriels sous forme concentrée (parfois après épuration). La directive cadre sur l’eau (EU-WFD, 2000), a largement contribué à l’amélioration des effluents urbains. Pour la silice, ces apports *ponctuels* sont faibles, de l’ordre de 0.8 gSi.hab-1.j-1 (Garnier et al., 2002b; Sferratore et al., 2006b). Pour l’azote, ces apports peuvent être nitriques, ammoniacaux ou organiques et s’établissent de façon assez stable autour de 10-15 gN.hab-1.j-1, pour l’Europe occidentale (Billen and Garnier, 1999; Servais et al., 1999). Ces émissions per capita sont moins constantes pour le phosphore, utilisé comme agent séquestrant pour piéger le calcium dans les poudres à lessiver, puis substitué par l’addition de zéolites. Après avoir atteint 4 gP. hab-1.j-1 dans les années 80-90, les apports spécifiques de phosphore domestique sont aujourd’hui stabilisés à une valeur proche des rejets physiologiques, autour de 1 gP. hab-1.j-1 (Garnier et al., 2006).

La recherche de généricité s’attachant aux concepts écologiques tels que le continuum fluvial, implique qu’on les établisse le plus souvent dans le cadre d’écosystèmes naturels et non perturbés. Cependant, en conclusion de son étude, Vannote et al (1980) précise l’adaptation du concept de continuum à des systèmes anthropisés, et souligne le principe d’équilibre dynamique régissant les communautés capables de s’adapter à des changements géomorphologique, physique et biologique. En s’accordant sur ce principe, l’hydrosystème est alors une unité intégratrice de l’ensemble des perturbations d’origine anthropiques altérant le fonctionnement des écosystèmes aquatiques qui se succèdent depuis les têtes de bassins jusqu’à la zone côtière (Billen et al., 1995).

Le nombre impressionnant d’études sur l’enrichissement des zones côtières comptabilisées depuis les années 50 et rappelé par Boesch (2002), n’est pas uniquement lié à un regain d’intérêt pour les environnements côtiers, mais également à une augmentation simultanée, très nette, des manifestations d’eutrophisation. L’augmentation des quantités de nutriments exportées vers les zones côtières est aujourd’hui reconnue comme la conséquence de l’ouverture des cycles d’éléments biogènes à l’échelle du monde (Green et al., 2004; Seitzinger et al., 2002a), en lien direct avec les activités supportées par les bassins versants (Galloway et al., 2004; Meybeck, 2003; Vitousek et al., 1997).

Le cas exemplaire des hydrosystèmes Seine-Somme-Escaut et de la zone côtière franco-belge

La zone côtière Manche Orientale-Sud Mer du Nord constitue un exemple caractéristique d'eutrophisation anthropique se manifestant sous forme de blooms printaniers de Phaeocystis qui affectent l'ensemble du littoral franco-belge (Lancelot et al., 1987). Bien qu'elles ne produisent pas de substance toxique, les efflorescences de Phaeocystis peuvent altérer le fonctionnement trophique des écosystèmes marin-côtiers, en limitant la croissance d'autres espèces phytoplanctoniques, et en formant des colonies qui atteignent rapidement des dimensions telles que le zooplancton, notamment les copépodes, ne peuvent les consommer (Lancelot, 1995; Rousseau et al., 1990).

La Convention pour la Protection du Milieu Marin de l'Atlantique du Nord-Est, dite, convention (OSPAR), classe cette partie du littoral comme « zone à problème » (OSPAR, 2005). L'abondance des proliférations d'algues nuisibles est un des critères communément retenus par la « Procédure » de classement OSPAR et les indicateurs « Phytoplancton » de la Directive Cadre sur l'Eau (DCE, 2000/60EC). Les observations analysées par Lancelot et al, (2009) montrent que la zone côtière franco-belge est bien au-delà du seuil de 10^6 cell.L^{-1} à partir duquel la prolifération est considérée comme nuisible. Par ailleurs les efflorescences de Phaeocystis ont été identifiées comme indicateur d'une détérioration des milieux aquatiques (Tett et al., 2007). L'enrichissement excessif de cette zone côtière résulte des apports terrigènes de trois hydrosystèmes anthropisés, la Seine et la Somme et l'Escaut. L'influence des systèmes Rhin/Meuse, plus au nord, et de la Tamise outre-Manche a par ailleurs été estimée minoritaire en raison des circulations hydrodynamiques (Lacroix et al., 2007).

Les trois hydrosystèmes diffèrent considérablement par leurs propriétés morphologiques, physiques mais également par la nature et l'intensité de leurs activités humaines (Tableau 0.1).

Table 0.1 **:** Caractéristiques des bassins versants Seine, Somme et Escaut

	Seine	**Somme**	**Escaut**
Surface, km²	76 370	6 190	19 860
Drainage, km	22 402	712	3265
Ordre, max.	7	4	6
Débit moyen, $m^3.s^{-1}$	500	40	180
Population, hab.km-2	202	101	496
Elevage, ugb.km-2	15	23	70
Terres arables, %	53	77	40
Prairies, %	10	4	8
Forets, %	26	8	8
Organisation intra-bassin	Périphérique, autour de l'agglomération parisienne	Essentiellement agricole, avec 3 injections urbaines équidistantes	Mosaïque entre territoires urbain et ruraux

La population du bassin de la Seine (densité de 202 hab/km²) est essentiellement concentrée dans la partie centrale du bassin (région parisienne) et la zone estuarienne. En périphérie s'organise des activités agricoles, comprenant de grandes cultures céréalières, tandis que l'élevage est concentré dans les bordures périphériques.

Le bassin de la Somme présente un réseau hydrographique nettement moins dense que les deux autres bassins, principalement situé en zone crayeuse. Les activités agricoles occupent la plus grande partie de la surface du bassin versant. La densité de population sur le bassin de la Somme est relativement faible (101 hab/km²), avec seulement trois grands centres urbains : Saint Quentin, Amiens et Abbeville, situés respectivement en amont, dans la partie centrale et à l'aval.

De taille intermédiaire, le bassin de l'Escaut possède une densité de population bien plus élevée (496 hab/km²), ainsi que des activités d'élevage intensif (70 UGB/km²). A l'inverse de la Seine, l'Escaut offre des centres urbains de taille inférieure, mais mieux répartis sur l'ensemble du bassin.

Dans ce contexte d'eutrophisation avérée, il est essentiel de comprendre le fonctionnement des trois hydrosystèmes contributifs, afin de pouvoir relier la distribution et l'intensité des activités humaines sur les bassins versants aux manifestions indésirables de l'eutrophisation à la zone côtière. L'utilisation d'un modèle écologique est alors requise pour intégrer à l'échelle d'un territoire plurirégional, les transferts d'azote, de phosphore et de silice, qui impliquent des processus à des résolutions spatiales beaucoup plus fines.

La démarche de modélisation Riverstrahler

Le modèle écologique Riverstrahler (Billen et al., 1994; Garnier et al., 1995), en accord avec le concept de continuum fluvial (Vannote et al., 1980) prend en compte, de manière détaillée, la physiologie des développements algaux et, simule les cycles biogéochimiques du carbone, de l'azote, du phosphore et de la silice dans l'ensemble d'un réseau hydrographique.
Cette approche est structurée autour d'une hypothèse forte, celle de l'unicité des processus à travers l'ensemble du continuum aquatique. Le module RIVE est utilisé pour représenter les interactions microscopiques dans la colonne d'eau et à l'interface avec le sédiment (Figure 0.3). Le modèle simule les concentrations en oxygène, en nutriments (ammonium : NH4 ; nitrate : NO3 ; phosphates : PO4 ; phosphore inorganique particulaire : PIP ; silice dissoute : SiO2 ; silice biogénique : BSi), en matières en suspension (MES) et en carbone organique dissous et particulaire (COD, COP, 3 classes de biodégradabilité). Les compartiments biologiques sont représentés par trois classes taxonomiques d'algues (diatomées, chlorophycées, cyanobactéries, Garnier et al., 1995), deux types d'organismes zooplanctoniques (rotifères à temps de génération court et microcrustacés à temps de génération lent, Garnier et al., 1999a), deux types de bactéries hétérotrophes (petites bactéries autochtones et grandes

bactéries allochtones, Garnier et al., 1991) ainsi que des bactéries nitrifiantes (Brion and Billen, 1998; Brion et al., 2000). Les bactéries fécales sont également prises en compte en temps que variables d'état. Le modèle considère également des processus de dégradation de la matière organique (MO) et variables benthiques associées (les formes de l'azote, phosphore et silice précédemment mentionnées, Thouvenot et al., 2007).

Les paramètres cinétiques caractérisant ces processus sont fixés *a priori*, à partir d'observations, de la littérature, d'expérimentations de terrain ou de laboratoire, et ne font donc l'objet d'aucune procédure de calage (cf. Annexe 1)

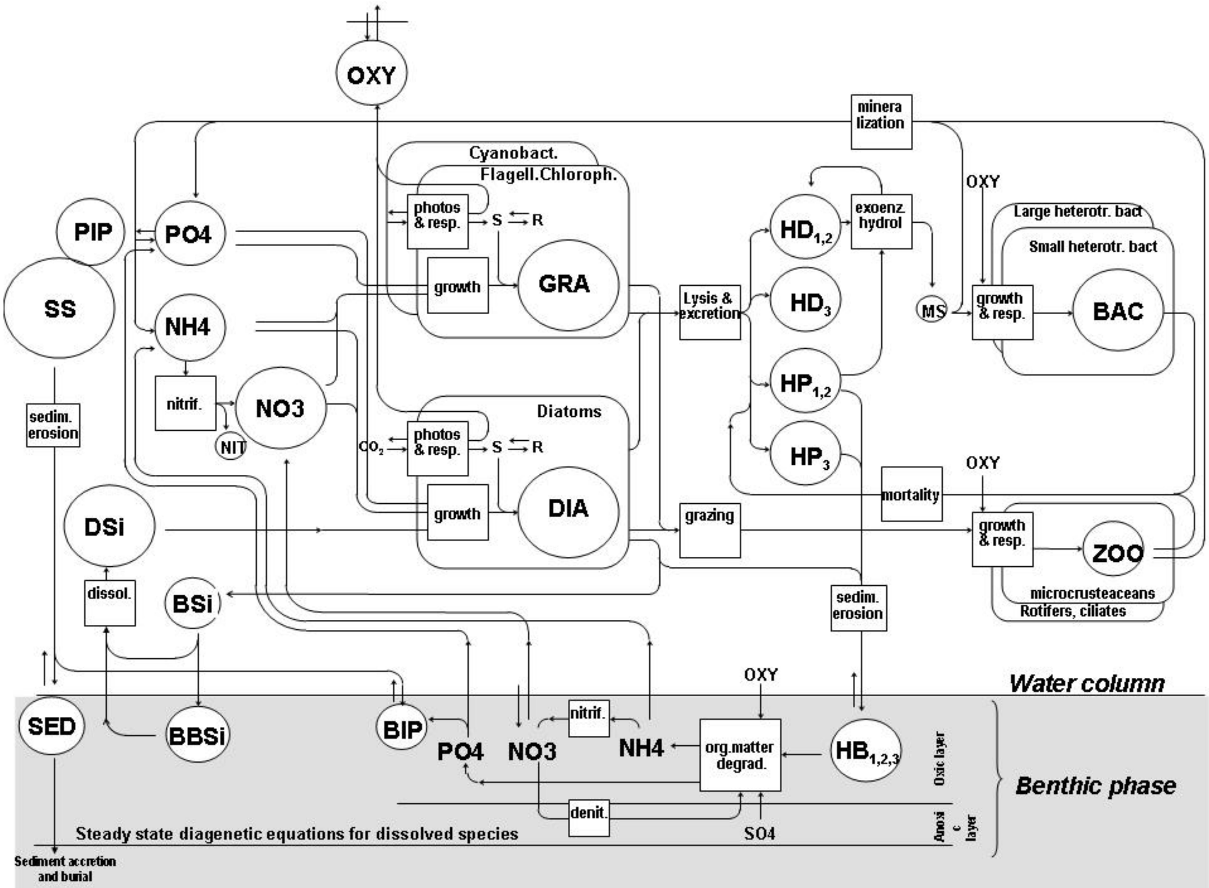

Figure 0.3 : Représentation schématique du modèle RIVE des processus biogéochimiques dans les réseaux hydrographiques (d'après Thouvenot et al., 2007).

En considérant que les même processus opèrent dans tous les écosystèmes aquatiques et obéissent aux mêmes cinétiques microscopiques, la démarche, proposée par le modèle Riverstrahler, fait également l'hypothèse que toute la variabilité du fonctionnement des écosystèmes s'explique par la diversité des contraintes s'appliquant au continuum aquatique. La description des ces contraintes est alors l'élément clé pour appréhender le fonctionnement d'un hydrosystème.

Conceptuellement, le modèle Riverstrahler permet de décrire le réseau hydrographique comme une combinaison de trois types d'objet : des sous-bassins amont structurés selon la classification de Strahler (1957), des tronçons de rivières avec une résolution kilométrique, et des milieux stagnants (étangs ou réservoirs) en connexion. Doté d'une interface SIG (Seneque : Ruelland et al., 2007), le modèle Riverstrahler peut ainsi être implémenté à des résolutions variables, depuis la représentation la plus simple où un fonctionnement moyen est simulé pour chaque niveau hiérarchique (ordre de Strahler), jusqu'à l'analyse kilométrique de tous les tributaires du réseau.

Toutefois, tendre vers une représentation détaillée du réseau hydrographique, n'a de sens que si les contraintes externes qui s'y appliquent sont également décrites avec une finesse suffisante, et ce, afin de pouvoir obtenir *in fine* une validation cohérente de notre connaissance du fonctionnement des hydrosystèmes à cette échelle.

Objectifs et organisation des travaux

L'ambition initiale de ce travail, est de relier successivement, l'intensité et la distribution des activités humaines, les quantités de nutriments qu'elles émettent ainsi que leurs transferts, et l'impact de ces apports sur les proliférations algales indésirables à la zone côtière. Les problèmes d'eutrophisation côtière sont généralement abordés à l'exutoire d'un seul système. L'originalité de cette étude est de proposer une vision cohérente qui intègre la contribution des trois bassins versants : la Seine, la Somme et l'Escaut, qui constituent les principaux apports terrigènes à la zone côtière franco-belge de la Manche-Orientale et la Mer-du-Nord.

Le programme PIREN a structuré une grande partie des travaux de recherche sur la Seine, notamment au travers de la conception du modèle Riverstrahler. La poursuite des développements en cours, sur le fonctionnement écologique de la Seine et la transposition de cette démarche de modélisation aux bassins versants de la Somme et de l'Escaut, constituent l'originalité des travaux présentés ici. Un premier objectif est de valider cette approche, pour la période récente, au travers une description homogène d'un territoire de plus de 100 000 km² afin d'apprécier le gradient anthropique qui s'y établit et qui contraint le fonctionnement des trois fleuves.
En caractérisant de façon précise, la nature, l'intensité et la spatialisation des activités humaines, il est alors possible de s'interroger sur la réponse des hydrosystèmes suite à une modification des usages sur les bassins versant. Cette démarche visant à caractériser les pressions anthropiques, constitue un enjeu de taille dans un contexte réglementaire, imposant aux gestionnaires de l'eau des obligations de résultats à courte échéance (Directive Cadres sur l'Eau). Elle mobilise depuis plusieurs années la communauté scientifique, à l'échelle locale, au travers de programmes de recherche se structurant autour de l'unité de gestion des bassins versants (cf. Le PIREN-Seine depuis 20 ans), ainsi qu'à

l'échelle globale. Dans ce dernier cas, on citera par exemple les travaux du SRES (Special Report on Emissions Scenarios) ou encore l'évaluation des changements écosystémiques pour le millénaire (Millenium Ecosystem Assessment).

L'intérêt d'une approche plurirégionale est d'offrir une espace de réflexion cohérent avec le fonctionnement écologique des systèmes étudiés, c'est-à-dire incluant à la fois les bassins, les réseaux hydrographiques et la zone côtière. La double ambition de cette approche intégrée est de pouvoir identifier les usages « terrestres » préjudiciables à l'équilibre du domaine côtier marin, et d'évaluer la possibilité de réguler ces usages pour prévenir aux phénomènes d'eutrophisation.

Pour répondre à ces ambitions, le présent mémoire est structuré en trois parties :

La première partie « **sources et transfert des éléments nutritifs (N, P, Si) dans les réseaux hydrographiques** » détaille la méthodologie mise en œuvre pour décrire de façon similaire les caractéristiques naturelles et anthropiques des trois bassins. Cet état des lieux, réalisé pour la période récente, forme alors une base solide pour analyser la variabilité du fonctionnement de chacun des hydrosystèmes. Différentes approches de modélisation sont présentées et l'intérêt d'une représentation fine des processus dans l'ensemble des réseaux hydrographique est souligné au travers de l'application du modèle Riverstrahler. Cette partie aboutit à une quantification détaillée de la rétention et du transfert des éléments N, P et Si, ainsi qu'à l'identification des sources responsables de ces niveaux d'émissions.

La seconde partie « **analyse prospective d'une modification des usages dans les bassins versants** » s'appuie sur le constat d'un dysfonctionnement global quantifié à l'exutoire des trois systèmes. Elle propose de s'interroger sur la possibilité de rétablir l'équilibre des flux terrigènes exportés à la mer. Le bilan des émissions, établi dans la première partie, sert alors de base pour la mise en œuvre de mesures sanitaires et agro-environnementales, permettant de réduire respectivement les apports ponctuels et diffus aux réseaux hydrographiques. Dans une prospective à plus long terme, l'apport des modèles globaux dans le cadre d'une démarche de modélisation à l'échelle plurirégionale sera également analysé.

La troisième partie « **approche intégrée du continuum rivière – zone côtière** » présente une modélisation complète du fonctionnement des systèmes aquatiques, couplant le modèle de réseau hydrographique Riverstrahler avec le modèle côtier MIRO. Le dysfonctionnement des écosystèmes côtiers est caractérisé pour la période récente et l'impact des mesures de gestions, présentées dans la seconde partie, est établi. Une évaluation économique de la réduction des émissions dans les bassins est réalisée, afin de rapprocher efficacité écologique et efficacité économique.

Le document se termine par une conclusion générale sur l'ensemble des travaux menés et des éléments de perspective.

Première partie

Sources et transfert des éléments nutritifs (N, P, Si) dans les réseaux hydrographiques

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L'*usage* d'un bassin versant se caractérise par la nature et l'intensité des activités humaines qu'il supporte. Au niveau biogéochimique, ces activités se traduisent par une circulation d'éléments biogènes (C, N, P, Si) qui altèrent les flux d'éléments nutritifs naturellement transférés du milieu continental à l'hydrosystème. Selon Meybeck et Vörösmarty (2005) l'*usage* fait par l'homme des bassins versant a contribué à l'apparition de nouvelles sources d'éléments biogènes, mais également à l'apparition de nouveaux filtres comme les réservoirs, dérivant une partie des écoulements, piégeant les formes particulaire, et soutenant une activité biogéochimique importante (rétention de silice, dénitrification, cf. Garnier et al., 2005a). Parallèlement, l'augmentation des pressions agricoles et l'urbanisation conduit à la réduction de certains filtres naturels, comme les zones humides, les plaines inondables et les estuaires.

A l'échelle du continuum « terre – océan », les réseaux hydrographiques sont à la fois des *vecteurs*, puisqu'ils acheminent par confluence les quantités de nutriments qui leur parviennent, et des *filtres*, en étant le siège de processus de transformation, d'immobilisation et d'élimination (Billen et al., 1991). Ces hydrosystèmes sont en relation permanente avec leur bassin et la modification de certains filtres naturels, riverains ou lentiques décrits par Meybeck et Vörösmarty (2005) affecte directement tant les apports parvenant aux cours d'eau que la dynamique des processus opérant dans les parties lotiques. Les écosystèmes aquatiques s'écartent donc un peu plus de leur fonctionnement naturel pré-anthropique, et les flux d'azote, de phosphore et de silice sont altérés à la fois dès leurs sources et tout au long des réseaux hydrographiques, avec une modification tant des flux exportés, que des capacités épuratrices et de rétention.

En corollaire de cette notion de continuum aquatique, toute tentative de remédiation aux problèmes d'eutrophisation côtière doit préalablement intégrer une analyse fine des secteurs amont. Cette étape est notamment indispensable pour établir un bilan qualitatif et quantitatif des émissions dans les bassins versants contributifs. Ensuite, la traduction de ces émissions en flux exportés vers la mer, nécessite l'usage de modèles écologiques plus ou moins complexes capables d'intégrer le transfert (rôle *vecteur*) et la rétention (rôle de *filtre*) dans les réseaux hydrographiques.

L'objectif de cette première partie est donc de proposer une description suffisante de l'espace couvert par les bassins Seine, Somme et Escaut pour permettre la modélisation des transferts fluviaux de N, P et Si, et l'évaluation des quantités entrantes, retenues et exportées vers la zone côtière franco-belge.

Dans un premier chapitre, la singularité de chacun des bassins est relevée au travers des propriétés physiques (morphologie, hydrologie) et de l'*usage* fait de ces territoires. L'inventaire de ces contraintes a constitué une condition nécessaire à l'implémentation du modèle Riverstrahler, dans sa version spatialisée, aux trois hydrosystèmes Seine, Somme et Escaut. La comparaison des résultats du modèle aux observations disponibles, des concentrations et des flux de nutriments à l'exutoire et en divers points des réseaux hydrographiques, a permis de valider, pour la période récente, notre représentation des processus qui gouvernent les transferts de l'azote, du phosphore et de la silice.

La mise en perspective de ces résultats avec ceux d'autres modèles décrivant le devenir de l'azote (uniquement) fait l'objet d'un second chapitre, qui souligne l'avantage et les limites de ces différentes approches.

L'intérêt de ce travail réside donc dans l'analyse du fonctionnement de trois hydrosystèmes très contrastés, sur la base d'une représentation commune des contraintes et des usages, et en s'appuyant sur un outil de modélisation du fonctionnement biogéochimique appliqué de façon uniforme.

Il est ainsi montré au travers de cette première partie, que la diversité des activités humaines sur les bassins (Fig. I.1) se caractérise par des différences marquées en termes d'apports concentrés par les réseaux de collecte (sources ponctuelles) et ceux transférés par le ruissellement des eaux de pluie (sources diffuses) jusqu'aux réseaux hydrographiques (Fig. I.2). La quantification de ces apports permet d'abord d'actualiser les travaux du programme « PIREN Seine » sur la Seine et ceux de Billen et al. (2005) sur l'Escaut, mais en utilisant cette fois ci, une représentation spatialisée des contraintes sur cet ensemble de bassins versants, qui conduit de fait à une analyse plus fidèle de la distribution de ces émissions. La généralisation de ce travail a également permis d'établir un premier bilan spatialisé des transferts de nutriments pour le bassin de la Somme.

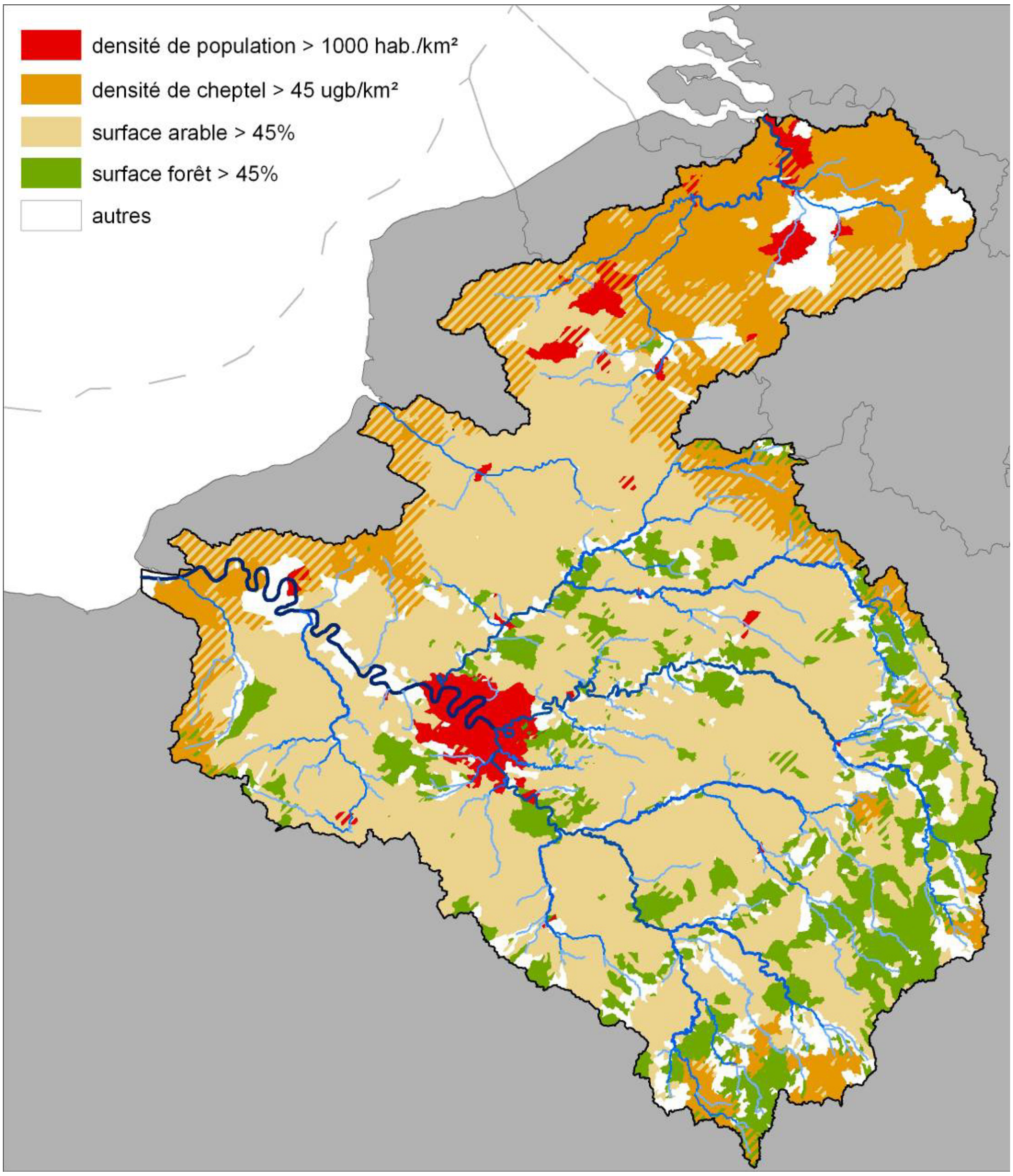

Figure I.1 : Carte synthétique des usages dominants par bassins versant élémentaires, étendue aux bassins de la Seine, de la Somme et de Escaut.

Les apports totaux d'azote et de silice apparaissent essentiellement diffus et sont largement dépendant des quantités d'eau transférées par les bassins, avec un facteur 10 entre les surfaces drainées par la Seine et la Somme. Les émissions de phosphore sont plus contrastées, avec des apports à dominance ponctuelle ou diffuse selon l'ordre. En effet, alors que la distribution des territoires agricoles décroît d'amont en aval (proportionnellement avec les surfaces drainées par ordre), les populations des trois bassins ont une organisation longitudinale très différentes, avec des agglomérations très concentrées (Seine) ou plus uniformément distribuées dans tout le bassin (Escaut).

A l'échelle d'une année, les quantités exportées par l'ensemble des trois bassins versant (Seine, Somme et Escaut) à la zone côtière franco-belge sont estimées entre 128 et 217 kT d'azote, 8 et 10 kT de phosphore et entre 54 et 116 kT de silice selon les conditions hydrologiques.

La contribution respective des trois bassins est largement conditionnée par leur taille. La normalisation de ces flux par les surfaces drainées (flux spécifiques) autorise la comparaison interbassins, et indique que la Somme, minoritaire par la taille, délivre proportionnellement plus d'azote, soit jusqu'à 5571 $kg.km^{-2}.an^{-1}$, contre 3961 $kg.km^{-2}.an^{-1}$ pour la Seine et 1568 $kg.km^{-2}.an^{-1}$ pour l'Escaut. A cela s'ajoute, l'analyse du rapport molaire de ces flux (N:P:Si) indiquant une carence importante de la silice par rapport à l'azote et au phosphore pour les trois bassins. Ces conditions apparaissent comme favorables au développement d'algues non siliceuses, caractérisant le plus souvent les phénomènes d'eutrophisation marine côtière (cf. parties II et III).

L'analyse détaillée des flux retenus dans les réseaux, souligne l'importance des processus benthiques pour le phosphore (jusqu'à 45% des apports à la rivière) et pour la silice (jusqu'à 35%). Pour l'azote, il est montré que les quantités retenues et/ou dénitrifiées sont plus importantes au niveau du sol et des parties riveraines que dans la colonne d'eau et les zones benthiques. L'analyse des transferts d'azote par d'autres modèles de bassins versants, présentés dans le chapitre 2, confirment cette tendance sans cependant intégrer l'effet des variations hydrologiques -qui apparaissent essentielles-, dans la variabilité des flux de exportés. Bien qu'étant moins informatif sur la nature des processus de rétention, ces modèles proposent néanmoins de s'interroger sur l'établissement des quantités nettes d'azote apportées aux bassins versants.

L'épandage de fertilisants inorganiques constitue la principale source d'azote dans les trois bassins étudiés qui affichent des niveaux d'application bien supérieurs à la moyenne des bassins européens. Dans l'évaluation des apports nets d'éléments biogènes au bassin versant, compte aussi la différence entre les quantités produites ou importées, et celles consommées localement ou exportées hors du bassin versant. Selon ce critère, l'Escaut apparaît comme un importateur net de produits agricoles, pour soutenir ses activités intensives d'élevage, alors que la Seine et la Somme exportent une grande partie de leur production agricole (Fig. I.3).

Une quantification des transferts *terrestres* complète ainsi le bilan *aquatique* de l'azote, mettant en évidence l'impact des différents *usages* en place sur les bassins, et notamment ceux qui seront à considérer dans le cadre d'une politique de maîtrise de la contamination des eaux de surface et des zones côtière associées (voir partie II).

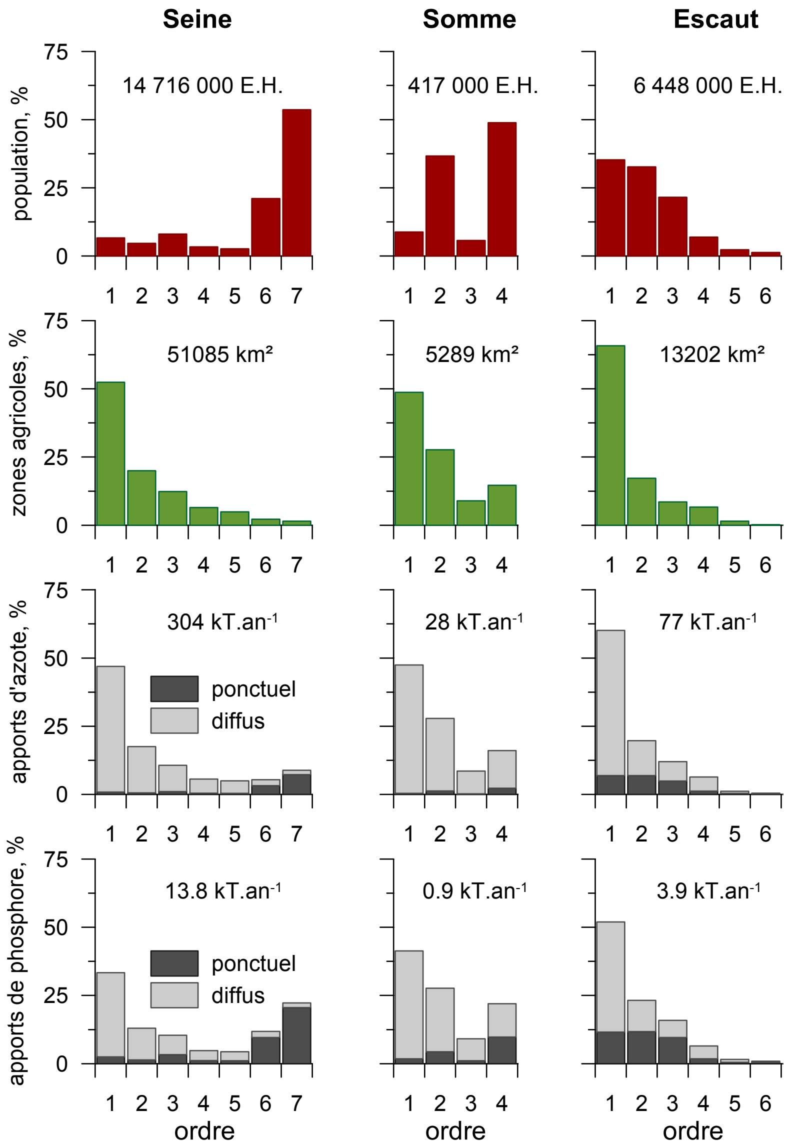

Figure I.2 : Bilan qualitatif et quantitatif des apports d'azote et de phosphore en lien avec la distribution des territoires agricoles et les densités de population

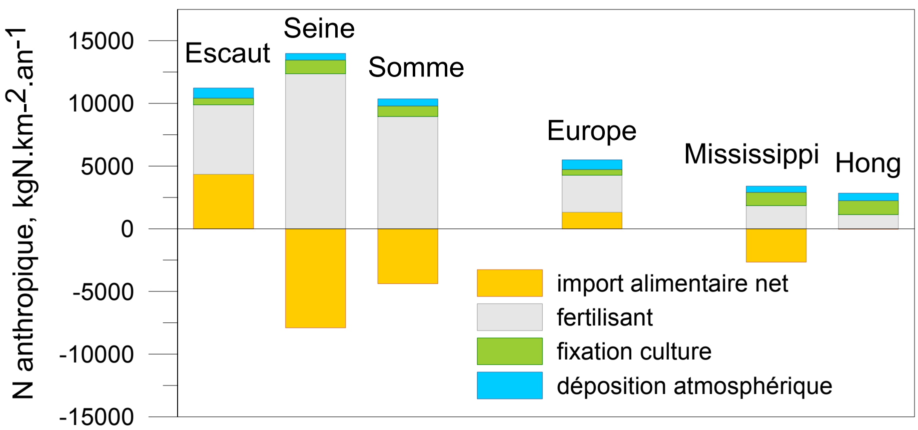

Figure. I.3 : Apports nets d'azote anthropique aux bassins versants pour l'année 2000

CHAPITRE

1 - Bilan des transferts à l'échelle des continuums fluviaux : Seine, Somme et Escaut

Nutrient transfer in three contrasting NW European watersheds: the Seine, Somme, and Scheldt Rivers. A comparative application of the Seneque/Riverstrahler model

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[1] UPMC Univ Paris 06, UMR 7619 Sisyphe, F-75005 Paris, France
[2] CNRS, UMR 7619 Sisyphe, F-75005 Paris, France

Abstract

An understanding of the ecological functioning of an aquatic continuum on a multi-regional scale relies on the ability to collect suitable descriptive information. Here, the deterministic Seneque/Riverstrahler model, linking biogeochemistry with the constraints set by geomorphology and anthropogenic activities, was fully implemented to study the Seine, Somme, and Scheldt Rivers. Reasonable agreement was found between calculated and observed nutrient fluxes for both seasonal and inter-annual variations along the networks. Nutrient budgets underline: i) a clear partition of diffuse and point sources with respect to the specific activities of the watersheds, ii) the importance of riparian retention, responsible for 25–50% of nitrogen retention, iii) the role played by benthic processes, resulting in the retention of up to 45% of the phosphorus and 35% of the silica entering the river systems. Nutrient ratios confirmed that fluxes to the Eastern Southern Bight of the North Sea are imbalanced, supporting the potential for undesirable algal blooms

1.1. Introduction

Several studies have emphasized the effect of nutrient delivery from river basins on eutrophication at the coastal zone (Billen and Garnier, 1997, 2007; Conley, 2000; Lancelot et al., 2005; Officer and Ryther, 1980; Turner et al., 1998a; and many others). In the last 50 years, the river loads of phosphorus and nitrogen have dramatically increased while the silica input has remained rather stable and even decreased with damming (Billen and Garnier, 1997; Garnier et al., 1999b; Humborg et al., 2000; 2006; Sferratore et al., 2008). Concomitant signs of coastal eutrophication in several places world-wide have been reported as in Northern Europe (Cugier et al., 2005; Lancelot et al., 2005), North America (Jaworski et al., 1992; Turner and Rabalais, 1994) or Adriatic (Justic et al., 1995; Marchetti et al., 1989).

The continental coastal zone of the Channel and Southern Bight of the North Sea offers the example of an area where ecological functioning has been strongly affected by coastal eutrophication during the last several decades, despite huge efforts in the last few years to improve the treatment of polluted effluents in wastewater treatment plants, as required by the European Water Framework Directive (EU-WFD, 2000). The French and Belgium costal zones are currently described as a "problem area" with regard to nutrient enrichment and eutrophication (OSPAR, 2005). Consequently, studies of the biogeochemical functioning of the major delivering river systems are needed to assess the transfer and retention of biogenic elements along the entire drainage networks to the sea. The Riverstrahler model (Billen et al., 1994) associated with a GIS-based approach (Seneque/Riverstrahler: Ruelland, 2004; Ruelland et al., 2007) is one of the few available tools for an in-depth analysis of any drainage network and was applied to evaluate three major rivers (Seine, Somme, and Scheldt, here called "3S") that discharge into the Eastern Channel and Southern Bight of the North Sea (Figure 1-1 a).

Within this multi-regional area and its trans-boundaries watersheds (France, Belgium, Netherlands), population density, land use, human activities and agricultural practices differ widely but their combined intensities have led to important enrichments of the surface water in nitrogen and phosphorus and in significant alterations of the ratios between those two nutrients and silica. Nutrients exported to the coastal zone are necessary for algal primary production, but an imbalance of these fluxes promotes the growth of undesirable or harmful algae, causing eutrophication.

We first briefly present the newly developed Seneque software, used in this study to implement spatially the Riverstrahler model in direct linkage with broad GIS-databases. Then we describe the contrasting landscapes of the 3S in terms of urbanization, agriculture, morphology of the drainage network, and impoundment of artificial reservoirs, each of which being factors that have direct consequences on nutrient emission, transport, and retention. Specific properties of the basins of the Seine, Somme, and Scheldt Rivers are considered in terms of hydrologic regimes, non-point nutrient sources, and urban discharge of wastewaters.

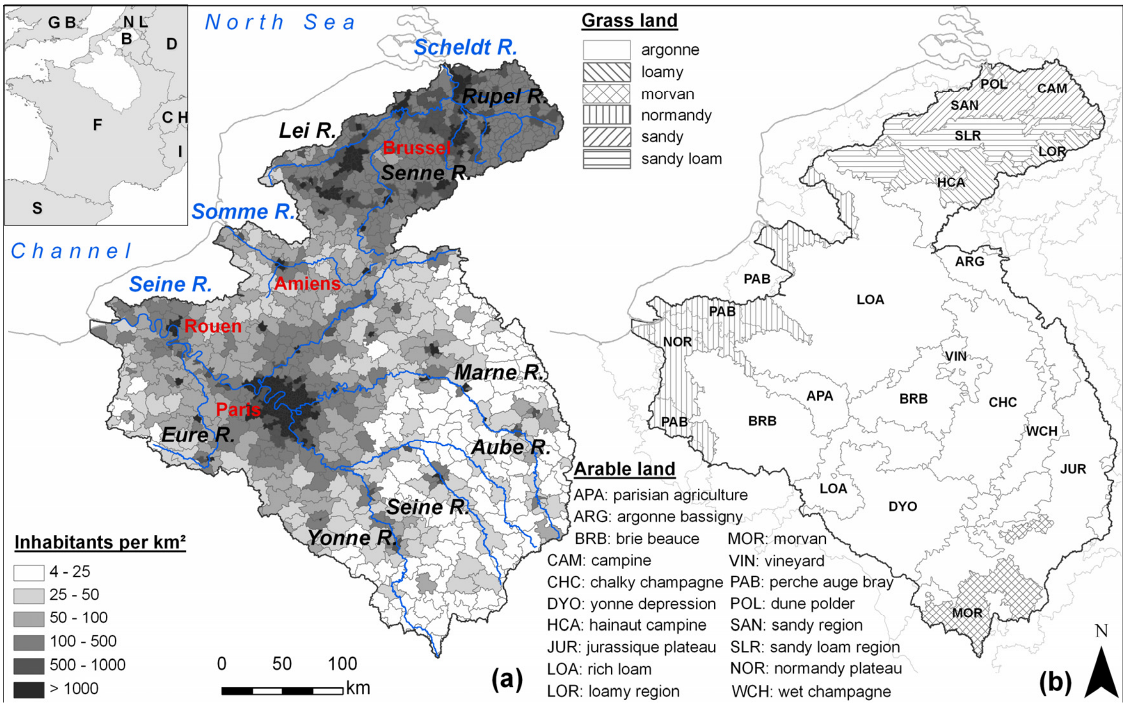

Figure 1-1: (a) General map of the Seine, Somme, and Scheldt Rivers, indicating the main tributaries (blue) and urban centers (red). Population density (inhabitants.km^{-2}) is symbolized in gray graduation levels. (b) Homogeneous agricultural zones used for a spatial segmentation of arable land and grassland.

Since our model is based on a deterministic approach, with all parameters set *a priori* without any calibration step, except for the hydrology, a major challenge was the comparison between the simulation results of biochemical water-quality variables and available observations. In being able to do so, we were able to validate our understanding of the basic processes taken into account herein and to test the coherence of the input data. Moreover, the results obtained for recent and contrasted hydrological years enabled us to calculate the budgets of nitrogen, phosphorus, and silica cycling. A major emphasis of the discussion is put on the N:P:Si ratios of the fluxes exported to the sea and their potential for supporting the growth of undesirable or harmful algae at the French and Belgium coastal zones.

1.2. Study site and modeling approach

1.2.1. The Riverstrahler approach

The aim of the deterministic Riverstrahler model is to simulate the impact of human activities on the quality of river systems by describing the kinetics of all microscopic processes occurring within drainage networks. Accordingly, the drainage network of any river system is considered as a combination of basins, idealized as a regular scheme of confluent tributaries of increasing stream order (Strahler, 1957), each characterized by mean morphologic properties and connected to branches represented with higher spatial resolution. The advantage of this representation of the drainage network is that the processes occurring in small first-order streams, headwater streams, and large tributaries can all be taken into account.

The model couples water flows routed through the defined structures of basins and branches with a description of the biological, microbiological, and physicochemical processes occurring within the water masses. The variables comprise nutrients, oxygen, suspended matter, dissolved and particulate non-living organic carbon, as well as algal, bacterial, and zooplankton biomasses. Most of the processes important in the transformation, elimination, and/or immobilization of nutrients during their transfer within the network of rivers and streams are explicitly calculated, including algal primary production, aerobic and anaerobic organic-matter degradation by planktonic and benthic bacteria, coupled with oxidant consumption and nutrient remineralization, nitrification and denitrification, phosphate reversible adsorption onto suspended matter and subsequent sedimentation, etc. A detailed description of the model and of the parameters used can be found in Garnier and Billen (2002) and, for benthic processes, in Thouvenot et al. (2007). The Riverstrahler model was first developed for the Seine River system in France (Billen and Garnier, 1999; Billen et al., 2001; 1994; Garnier et al., 1995; 2004; 2005a) , within the framework of the French interdisciplinary PIREN-Seine program. It has also been applied to the Mosel (France, Germany: Garnier et al., 1999a), Danube (Garnier et al., 2002a; Marchetti et al., 1989) and Scheldt (Belgium: Billen et al., 2005) Rivers.

For the 3S study, the implementation of the Riverstrahler, through the generic GIS Seneque interface, has involved the improvement of both databases description and software functionalities. Indeed, the Seneque application relies on a wide georeferenced database allowing the exploration of the ecological functioning of rivers with a more accurate spatial resolution, including a description of the drainage network morphology and the exact location of point sources as well as information about the land use, lithology, and fluvial corridors characteristic of each elementary watershed.

1.2.2. General description of the three basins

The adjacent drainage basins of the 3S rivers ranging from the north-western spread from north-western France to the Dutch border and include the main western part of Belgium, with more than 26,000 km of stream length and an overall 100,000 km^2 watershed area.

The Seine drainage consists of five main tributaries: i) The upper Seine (32,450 km^2) and ii) the Marne river (12,640 km^2) drain the south-eastern part of the basin and join immediately upstream from Paris, resulting in iii) the 7th-order River Seine, which downstream from Paris receives the effluents of the Paris conurbation and, on its right bank in iv) the 7th-order River Oise (17,330 km^2), which drains the northern part of the basin, while v) the River Eure (7,020 km^2) joins the Seine in its lower course at the entrance of its estuarine section. The Scheldt drainage network comprises three main sub-basins, with the upper Scheldt and the River Leie basins (respectively, 8,125 and 3,850 km^2)

originating from France and forming the eastern part of the watershed, whereas the Rupel (6,475 km^2) basin collects western tributaries. Surrounded by both the Seine and Scheldt basins, the Somme is the smallest river (6,800 km^2).

The distribution of the population and the organization of land use widely differ among the three basins. The Somme basin is dominated by intensive cereal agriculture, which forms the major part of the catchment area. The population density is relatively low (100 inhab/km^2), with only three large urban centers. These are settled in the upper part (Saint-Quentin 60,000 inhabitants), in the middle part (Amiens 160,000 inhabitants), and along the downstream canalized part of the drainage network (Abbeville 30,000 inhabitants). For the Seine basin, the population density is double (nearly 200 inhabitants/km^2) that of the Somme but mostly concentrated along the downstream part of the main Seine branch (from Paris to Le Havre). The middle part of the basin is one of the most intensive agricultural areas in the world, dedicated to the mass production of cereal and industrial crops. Animal farming is highly developed and mainly concentrated in the western and eastern peripheries of the basin. The Scheldt basin is densely populated, with more than 400 inhabitants/km^2. The Brussels conurbation, drained by the Senne, Dyle, and Rupel Rivers, represents a major attractive pole, with more than 2 million inhabitants, as is the case for the upstream southern part of the basin grouping the cities of Lille, Tourcoing, Roubaix (in France), and Mons (in Belgium). The area drained by both the Scheldt and Rupel Rivers, which join in Antwerp, is another main urban center and is dominated by industrial activities. Except in the southern part, the Scheldt landscape is dominated by urban areas; these prevent widespread and intensive agricultural activities and lead to a mosaic-type landscape, in which agricultural and cropland areas are mixed with surrounding cities. Agriculture is dominated by intensive cattle farming and, especially in the Flemish region, pig breeding.

1.3. Towards a restitution of the 3S heterogeneity

1.3.1. Morphology of the drainage network and hydraulic works

The descriptive database required by the Seneque model is organized around a digitalized representation of the hydrological network and its elementary watersheds (i.e., the direct basin of any first-order river or stretches of river between two confluences).

A reliable digital representation of the drainage network was obtained by classical GIS-based treatments (Tarboton et al., 1991) from the Digital Elevation Model (DEM that covers Europe with a resolution of 90-m grid cells (Shuttle Radar Topographic Mission). However, the downstream part of the Scheldt, with low elevation and a high drainage density, raises problems regarding DEM-derived output accuracy; thus, additional treatments based on drainage enforcement in DEM (Hutchinson, 1989; Saunders, 2000) were required. The slope and length of each stretch of river were calculated, and their mean width was evaluated by the following relationship (1-1):

W = 0.8 * (Awshd)1/2 (1-1)

where W is the mean width of a stretch of river (meters) and A_{wshd} is the area of its total upstream watershed (km²). This relationship has been successfully applied to the 3S river (n=97; r²=0.89), except in their downstream - and generally enlarged - parts, where navigation needs make this width information directly available.

The distribution of these basic morphological characteristics according to the Strahler's stream order (Table 1-1) highlights the differences between the three basins in terms of size and drainage density. The Scheldt River is characterized by wider streams but also by a lower slope, ranging from 0.002 to 0.001 m/m, respectively, for first and last stream orders.

Land use and population density in the elementary watersheds were calculated using, respectively, the National Statistics Census and the European Corine Land Cover 2000 database (CLC200: Bossard et al., 2000). Regarding the extent of arable land, the Somme basin is clearly dominated by agriculture, which account for up to 75% of total land use compared to 52% and 39%, respectively, for the Seine and Scheldt. The Seine sub-basins have the greatest percentage of forest land (25%), proportionally distributed within all its stream orders. The downstream position of the Paris area is marked at the 6th and 7th orders by a shift in population density, whereas urban areas along the Scheldt already occur at smaller orders.

Table 1-1: Comparison of total and mean (*) morphological, demographic, and land-cover characteristics of the Seine, Somme, and Scheldt drainage networks according to stream order.

	Number km	Length km	Width* m	Slope* m/m	Area km	Population density Inhab/km	Arable land %	Grassland %	Fores %t
Seine	***5163***	***22402***	***10.6***	***0.0098***	***76370***	***202***	***52.7***	***10.2***	***25.8***
Order 1	2594	11026	2.6	0.0151	39010	116	55.7	9.6	26.7
2	1263	4965	5.6	0.0062	14575	133	54.5	11.7	24.3
3	693	3078	12.2	0.0033	9554	170	48.2	13.2	26.4
4	290	1383	22.2	0.0023	4802	149	52.2	12.6	22.3
5	174	977	50.6	0.0014	3899	99	52.7	6.6	25.5
6	115	601	88.9	0.0013	2416	914	39.0	1.8	23.3
7	34	372	203.3	0.0011	2113	1909	21.8	8.0	27.0
Somme	***169***	***712***	***11.4***	***0.0074***	***6190***	***101***	***76.8***	***4.1***	***7.6***
Order 1	85	207	3.7	0.0130	2898	67	80.7	4.9	6.1
2	48	276	8.3	0.0023	1717	105	76.9	3.2	7.8
3	14	94	16.1	0.0013	611	120	71.4	1.1	13.3
4	22	135	44.8	0.0010	964	181	68.7	5.2	8.2
Scheldt	***446***	***3265***	***20.3***	***0.0020***	***19860***	***496***	***39.4***	***8.2***	***7.7***
Order 1	224	1637	5.8	0.0025	12556	395	43.1	8.3	8.2
2	106	811	12.2	0.0018	3495	567	34.7	9.5	6.2
3	54	370	21.7	0.0011	1873	890	38.3	6.7	9.3
4	45	322	46.2	0.0011	1414	521	31.5	7.2	5.4
5	10	84	77.9	0.0011	358	633	8.5	7.7	5.6
6	7	41	347.2	0.0010	165	1730	4.2	1.0	8.3

Specific features such as reservoirs connected to main river branches are also taken into account by the model. The Seine has three wide reservoirs; these are localized, respectively, in the upstream part of on the Aube (170×10^6 m^3), Marne (350×10^6 m^3), and Seine (205×10^6 m^3) Rivers. Daily water inflows and outflows provided by the Seine Basin Dam and Reservoir Management Institution (IIBRBS: Institution Inter-départementale des Barrages-Réservoirs de la Seine) were used to calculate the water balance of the reservoirs and the processes affecting the composition of the water released into the rivers. Along the Scheldt, hydraulic works dating back to the XIX century have deeply modified the natural river runoff, e.g. around the city of Gent, where most of the Leie discharge is diverted westwards through the Schipdonk Channel, which joins the North Sea at Heist.

1.3.2. Modeling the hydrology of the three basins

For Hydrostrahler, the hydrologic part of Riverstrahler, daily rainfall and evapotranspiration data are needed to calculate the seasonal variation of base flow and superficial runoff. This partition of runoff is necessary to integrate diffuse sources of nutrients coming from both the soil-root zone and the groundwater. Daily French meteorological data with an 8-km resolution are available from the SAFRAN grid analysis (MétéoFrance); Belgian data are published by the Belgian Royal Meteorological Institute (IRM) and are acquired at five stations (Uccle; Melle; Wasmuel; Koksijde; Chastre-Blaimont) homogeneously distributed and spatially spread over the Scheldt catchments by the Thiessen polygon method (Thiessen, 1911).

The core of Hydrostrahler is a simple rainfall-discharge model comprising two compartments (soil, aquifer) and four adjustable hydrological parameters (soil saturation, soil superficial runoff rate, infiltration rate from the soil to groundwater compartment, and groundwater runoff rate). Automated GIS procedures (see detailed description in Le et al., 2007) have permitted to optimize the adjustment of these parameters over several years of observed daily discharges. The goal of this calibration step over long time periods is to integrate the delayed response of groundwater, and *in fine* be able to compute the partition between groundwater and superficial runoff for a selection of the sub-basins within the 3S watersheds. Although this method is strongly influenced by the availability and the location of the observed discharge values, it has allowed us to improve Hydrostrahler simulations covering the three basins for a period of seven continuous years (Figure 1-2). The low seasonal variability of specific runoff of the Somme River, compared with the other two rivers, is directly related to the chalky nature of the watershed, with a high base flow index, especially during the low-water season. To assess the ecological functioning of the three drainage networks, contrasting hydrologic years have been selected: 1996 as a typical dry year, 2000 as an average year, and 2001 as a wet year. The mean specific runoff calculated for the years 1996 and 2001 showed higher values for the Scheldt (6.3 – 15.3 $L/km^2/s$) than for 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 former basin.

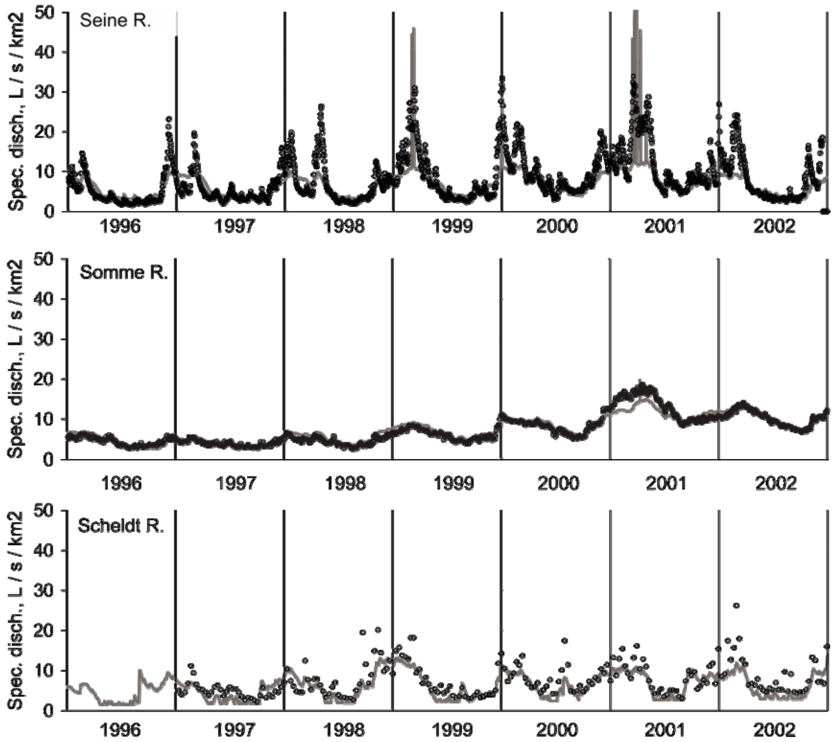

Figure 1-2: Specific runoff calculated (gray line) by the Hydrostrahler model and the observed value (dark circle). The data covered the period between 1996 and 2002 and was obtained at key stations of the Seine (Caudebec), Somme (Cambron) and Scheldt (downstream part of the Dendre affluent).

1.3.3. Land use and non-point sources of nutrients

Diffuse sources of nutrients entering the drainage network are taken into account by associating surface and sub-root water flow components, and by defining for each one, yearly mean concentration of nitrogen (nitrate and ammonium), total inorganic phosphorous (TIP), suspended matter (SM), and dissolved silica (DSi). These concentrations are related to different type of land use and the European CLC2000 database has provided an accurate description of the 3S land use area. For some generic land cover classes, such as urban or forest areas - among the highly detailed CLC nomenclature – nutrients diffuse sources are considered respectively homogenous whatever their location within the 3S territory. For these land-use classes, superficial and groundwater concentrations are based on observed values within the Seine basin, and are considered to be valid enough to be transposed to the Somme and the Scheldt. However, for arable land and grassland classes, a further distinction has been made to deal with the diversity of agricultural practices within and between the three basins. French “small agricultural regions” (SAR) and Belgian “agricultural regions” (BAR) have been considered as additional agricultural features as they provide well-documented zoning (Mignolet et al., 2007). Previous work performed on the Seine (Ledoux et al., 2007) basin was used to aggregate the 165 SAR, and the result was merged with that of the BAR to obtain a homogeneous segmentation of the entire territories, which were studied in the form of 18 agricultural areas (Figure 1-1 b). Within this seamless agricultural landscape, information regarding the spatial variability of diffuse sources was drawn from a compilation of the available literature (see below) and agricultural census (French: Agreste, 2000; Belgium: NIS, 2000) and then used to estimate mean values of nutrient concentrations for sub-root water and groundwater (Table 1-1).

Regarding suspended matter (SM), previous work on the Seine River (Garnier et al., 2005a) has proposed a relationship between SM in the river at high flow (> 20 L/km²/s, when it best represent the surface runoff composition) and the fraction of arable land in the catchments. This relationship has been retrieved by Billen et al. (2007b), who integrate the influence of buffer strip and riparian wetlands on the transfer of particulate matter, and proposed a mean concentration of suspended solids of 350 mg/l in all arable land before any retention. We assigned a mean concentration of 250 mg/l for most of arable lands of the 3S watersheds, and also integrated that the susceptibility of soils to be eroded and the types of agricultural practices in use, account for the variation in SM mean load. For this reason we made differences between arable lands, from no-cover soils (e.g. Vineyard: 300 mg/l) or sandy soils (e.g. Campine, Sandy Region: 150 mg/l).

Estimate of diffuse particulate inorganic phosphorous inputs is calculated from the SM inputs based on the exchangeable phosphorous content of soil (cPIP) which differs according to land use from 0.1 for forest to 1.7 gP/kg for the most intensive breeding area (Normandy grassland), intermediate values (0.7 - 1.4 gP/kg) being observed in the central area of the parisian basin (Némery et al., 2005). Associated to this, the dissolved inorganic phosphorus concentration (PO_4) is calculated using the adsorption-desorption equilibrium relationship proposed by Billen et al. (2007b):

$$cPIP = Pac * PO_4 / (PO_4 + KPads)$$
$$\text{or} \quad PO_4 = cPIP * Kpads / (Pac - cPIP) \qquad (1\text{-}2)$$

Where Pac = 0.0055 gP/kg (P saturation level) and KPads = 0.7 mgP/l (adsorption half saturation constant).

The ammonium concentration in soil leachate is relatively low (except in urban areas), with a mean of 0.06 mgN/L for arable land, where bacterial activity rapidly oxydises ammonium into nitrate and where adsorption prevents intense leaching. However, the exchange capacity of organic or sandy soil, as in the Belgian sandy agricultural region of Campine is low, yielding to a higher concentration of ammonium in leachate (0.14–0.42 mgN/L) as noted by De Becker (1986).

Table 1-2: Land-use types and corresponding nutrient concentrations in surface and base flows.

	Surface	**runoff**					**Base**	**flow**				
	NO_3	**NH_4**	**TIP**	**SM**	**DSi**	**BSi**	**NO_3**	**NH_4**	**TIP**	**SM**	**DSi**	**BSi**
	mgN/l	mgN/l	mgP/l	mg/l	mgSi/l	mgSi/l	mgN/l	mgN/l	mgP/l	mg/l	mgSi/l	mgSi/l
Urban area	0.7	0.7	0.16	70	3.64	2.52	0.4	0.35	0.13	20	3.64	0.10
Heterogeneous	11	0.05	0.25	200	3.64	1.01	3	0.02	0.12	20	3.64	0.10
Forest	0.4	0.02	0.03	100	3.64	1.01	0.3	0.01	0.02	20	3.64	0.10
Arable land												
APA	21	0.05	0.29	250	2.80	3.78	7	0.02	0.12	20	2.80	0.10
ARG	14	0.05	0.29	250	1.68	3.78	11	0.02	0.12	20	1.68	0.10
BRB	18	0.05	0.29	250	2.80	3.78	7	0.02	0.12	20	2.80	0.10
CHC	26	0.05	0.45	150	5.60	1.26	8	0.02	0.26	20	5.60	0.10
WCH	14	0.05	0.29	250	3.36	3.78	11	0.02	0.12	20	3.36	0.10
DYO	14	0.05	0.45	150	2.80	3.78	8	0.02	0.26	20	2.80	0.10
LOA	18	0.05	0.29	250	2.80	3.78	11	0.02	0.12	20	2.80	0.10
MOR	14	0.05	0.29	250	3.92	3.78	11	0.02	0.12	20	3.92	0.10
PAB	26	0.05	0.59	250	2.80	3.78	8	0.02	0.26	20	2.80	0.10
JUR	14	0.05	0.29	250	1.68	3.78	11	0.02	0.12	20	1.68	0.10
NOR	26	0.05	0.45	150	2.80	3.78	8	0.02	0.26	20	2.80	0.10
VIN	4	0.05	0.33	300	5.60	5.04	8	0.02	0.12	20	5.60	0.10
CAM	21	0.42	0.22	150	5.60	2.52	13	0.21	0.12	20	5.60	0.10
HCA	21	0.42	0.22	150	5.60	2.52	13	0.21	0.12	20	5.60	0.10
POL	18	0.21	0.65	250	5.60	3.78	13	0.11	0.29	20	5.60	0.10
LOR	18	0.05	0.29	250	2.80	3.78	11	0.02	0.12	20	2.80	0.10
SAN	21	0.14	0.49	150	5.60	2.52	11	0.07	0.29	20	5.60	0.10
SLR	28	0.06	0.25	200	4.20	3.53	11	0.03	0.12	20	4.20	0.10
Grassland												
Sandy loamy	4	0.06	0.14	70	4.20	0.35	3	0.03	0.11	20	4.20	0.10
loamy	3	0.05	0.14	70	2.80	0.35	2	0.02	0.11	20	2.80	0.10
sandy	5	0.14	0.37	70	5.60	0.35	4	0.07	0.29	20	5.60	0.10
argonne	3	0.04	0.14	70	1.68	0.35	2	0.02	0.11	20	1.68	0.10
mrovan	2	0.03	0.14	70	3.92	0.35	2	0.01	0.11	20	3.92	0.10
normandy	4	0.04	0.33	70	2.80	0.35	8	0.02	0.26	20	2.80	0.10

The greatest heterogeneity between agricultural practices and land use concerns nitrate contamination of superficial and base-flow runoff. For surface water, data from lysimetric (Ballif et al., 1996; Bonniface, 1996) or suction-cup experiments (Benoît et al., 1995) as well as results from nitrogen-transfer calculations (Lixim, Beaudoin et al., 2005) or agronomical models (STICS, Ducharne et al., 2007) have been used for the French part while in Belgium specific measurements in the Scheldt, as reported by Billen et al. (2005) and De Becker et al. (1984), were used to estimate the yearly mean value of nitrate in surface water. The latter yielded in-stream measurement values, with a mean 50% of riparian retention used to define the nitrate leaching value.

Nitrate concentrations in groundwater, resulting from infiltration and phreatic circulation together with long residence times, were determined from an inventory of aquifer compositions. Available observations (ADES, 2003) or model results (Viavattene, 2006) reported lower concentrations of nitrate in groundwater than in sub-root water since the two components are not yet in equilibrium following the recent intensification of agricultural practices, which have affected both surface and sub-root water. The Riverstrahler model also takes into account a calibrated term for riparian retention, since it influences the nutrient composition of sub-root water and groundwater flow from the watershed before the nutrients enter the surface water (Billen and Garnier, 1999; Sebilo et al., 2003).

Regarding their structural definition, the agronomical units (SAR aggregation and BAR) take into account lithological information and allowed us to associate a dominant parent material within each unit. According to previous works by Meybeck (1986) and Garnier et al. (2005b), dissolved silica concentrations, mostly resulting from rock weathering, have been defined for each type of soil/rock and transposed according to the scales of the different agronomical units. At this stage, the model does not take into account the potential influence of soil cover or other interactions with terrestrial vegetation, both of which might alter dissolved silica release. Biogenic forms of silica (BSi, from phytoliths erosion) were calculated on the basis of SM load, with a mean content of 4.9 mgSi/g, as observed in soils and winter suspended matter in the Seine catchments by Sferratore et al. (2006a).

1.3.4. Wastewater point sources

Point sources were taken into account as a mean daily amount of SM, organic matter, phosphorus, and the various forms of nitrogen discharged into surface water as urban wastewater. The present study focused on the recent period (1996–2002), with a total of 2216 domestic wastewater release points (for the year 2000). The data were gathered with the help of Artois-Picardie and the Seine Normandy Water Agencies for France and the upper Scheldt; for the Belgian Scheldt, data were provided by the SPGE (Société Publique de Gestion de l'Eau) and the VMM (Vlaamse Milieu Maatschapij), respectively, for the Walloon and Flemish regions. For each discharge point, the exact location was assigned and daily fluxes calculated based on per-capita emission, according to the nature of the performed treatment (including the absence of any treatment) and the effective capacity (in terms of inhabitant-equivalent) (Table 1-3). These discharged values were determined on a sample of water purification treatment observed on the Seine basin (Garnier et al., 2006; Servais et al., 1999) and assumed to be widely applicable to the 3S WWTPs. Dissolved and biogenic silica releases refer to the values quoted by Sferratore et al. (2006a) and remained unchanged irrespective of the treatment plant.

Organisation of population densities within the Seine basins leads to a highly contrasted distribution of point sources. First-order rivers, which drain more than 50% of the catchment areas, receive only 8.5% of the total wastewater inputs, while rivers of the highest orders (6 and 7) receive

40% of total wastewater inputs, although their direct watersheds represent less than 6% of the total area. For the Scheldt, population density remains high in the estuarine part and in the surroundings of Antwerp, but rivers with low stream orders (1–3), with important urban poles such as Lille (northern France) or Brussels, constitute the major part (more than 85%) of the total wastewater input. The situation of the Somme basin, where the mean population density is lower (100 inhab./km^2), is also very different; most of the total wastewater (70%) input is injected into the upstream, middle, and downstream parts of the main Somme branch from the only three large urban centers.

Table 1-3: Identification of the main water treatments for the three basins, effective treatment capacity as integrated by the Riverstrahler model, and corresponding per-capita values for the main components of the effluents (SM: suspended matter; OM: organic matter; NO_3: nitrate; NH_4: ammonium; PO_4: phosphate; DSi: dissolved silica; BSi: particulate biogenic silica).

	Effective treatment (Inhab. Equiv. x 1000)			Per capita emission						
				SM	OM	NO_3	NH_4	PO_4	DSi	BSi
	Seine	Somme	Scheldt	(g)	(gC)	(gN)	(gN)	(gP)	(gSi)	(gSi)
Lagooning	144	8	81	4	1.8	5	4	0.7	0.3	0.5
Biological treatment (Bt.)	8417	271	1393	10	4	0.1	8	1.6	0.3	0.5
Bt. + Dephosphatation	-	-	1376	8	3	0.1	8	0.2	0.3	0.5
Bt. + Nitrification (Nit.)	4539	-	57	8	2.4	8	2	1.7	0.3	0.5
Bt. + Nit. + Dpho.	32	-	251	8	2.4	8	2	0.2	0.3	0.5
Bt.+Nit.+Denitrification	1732	46	610	8	2	4	1	1.7	0.3	0.5
Bt. + Nit. + Dnit. + Dpho.	2495	93	1059	8	2	4	1	0.2	0.3	0.5
Other treatment	1038	-	42	-	-	-	-	-	-	-
No treatment	-	-	953	80	24	0	9	1.5	0	0.8

Regarding the level of the water treatment efficiency, the Scheldt is a step behind. In spite of an important sewage network, to which 90% of the population is connected, a large portion of urban emissions are released directly into the river, which, in 1995, had only a 35% connection to wastewater treatment plants (Billen et al., 2005). The situation has improved in the last decade and the connection to wastewater treatment plants in recent years is now more than 53% (Scaldit, 2004). The case of the Brussels conurbation is striking, as this large city did not have any wastewater treatment facilities before the Brussels South plant came on line in 2000 and the Brussels North plant in 2006, with respective capacities of 360,000 and 1,100,000 inhabitant-equivalents.

The Riverstrahler model integrates population constraints starting from the collection of household emissions into the sewage network until their release into the river with or without treatment. Difficulties for gathering reliable data on non-treated emissions led us to reconstruct the recent period by considering that, before the date of operation of a wastewater treatment plant, a discharge of non-treated water occurred at the same position, unless the new plant replaced an older one. This analysis was restricted to the most important point source, i.e., with capacities above 20,000 inhabitant-equivalents. It led to an overall estimate for the Scheldt population of a 49% connection to the wastewater treatment plant, in agreement with the official value previously quoted. By contrast, French Water Authorities Databases do not mention any non-treated effluent release for the Seine and the Somme basins.

The collection of information and the construction of the georeferenced database were essential to represent fully the urban discharges over the 3S basins and to ensure the spatial homogeneity of the model inputs in terms of accuracy. However, because of difficulties in acquiring reliable and homogeneous data on pollutant fluxes generated by French industrial activities, only industrial release connected to municipal sewage network were considered for the Somme and the Seine basin, i.e., excluding industries equipped with their own treatment facilities.

1.4. Simulated and observed surface water quality

For the three river systems, the Riverstrahler model was run for three different years: 1996 (dry), 2001(wet), and 2000 (hydrologically medium). The year 2000 was also defined as the reference for land-cover information, which was considered unchanged during the study period. Comparisons of simulations with observed values are presented for the last downstream stations to assess the contribution of the entire watershed to nutrient fluxes exported to the sea (Figure 1-3). Nitrate concentrations depend mainly on hydrological conditions. A good fit was obtained for the simulations involving the Seine River, while those for the Somme slightly overestimated the seasonal variations of nitrate. The simulations run for the Scheldt underestimated nitrate values systematically in summer, and in a large extend for whole dry year. This discrepancy suggests an underestimation groundwater contribution either in term of runoff or in term of nitrate concentration. It also pleads for a too low level of nitrification activities simulated in this part of the Scheldt, with respect the concomitant overestimation of ammonium and oxygen concentrations.

Regarding silica and phytoplankton biomass, in both the Seine and the Scheldt Rivers, the calculated seasonal variations of dissolved silica were in general agreement with available data, showing depletion during periods of intense phytoplankton growth. However, during dry years some of phytoplankton levels were underestimated in the Seine River, but this potential underestimation of nutrient uptake will not affect significantly nutrient concentrations (taking into account the ratios defined by Redfield et al., 1963 for the phytoplankton). No such variations were simulated for the Somme River, and, despite a lack of observations for silica and chlorophyll concentrations at this site, the findings were consistent with the view that intense algal development only occurs in large rivers when the residence time of the water is higher than the dilution factor (Garnier et al., 1995), which does not seem to be the case for the Somme drainage system.

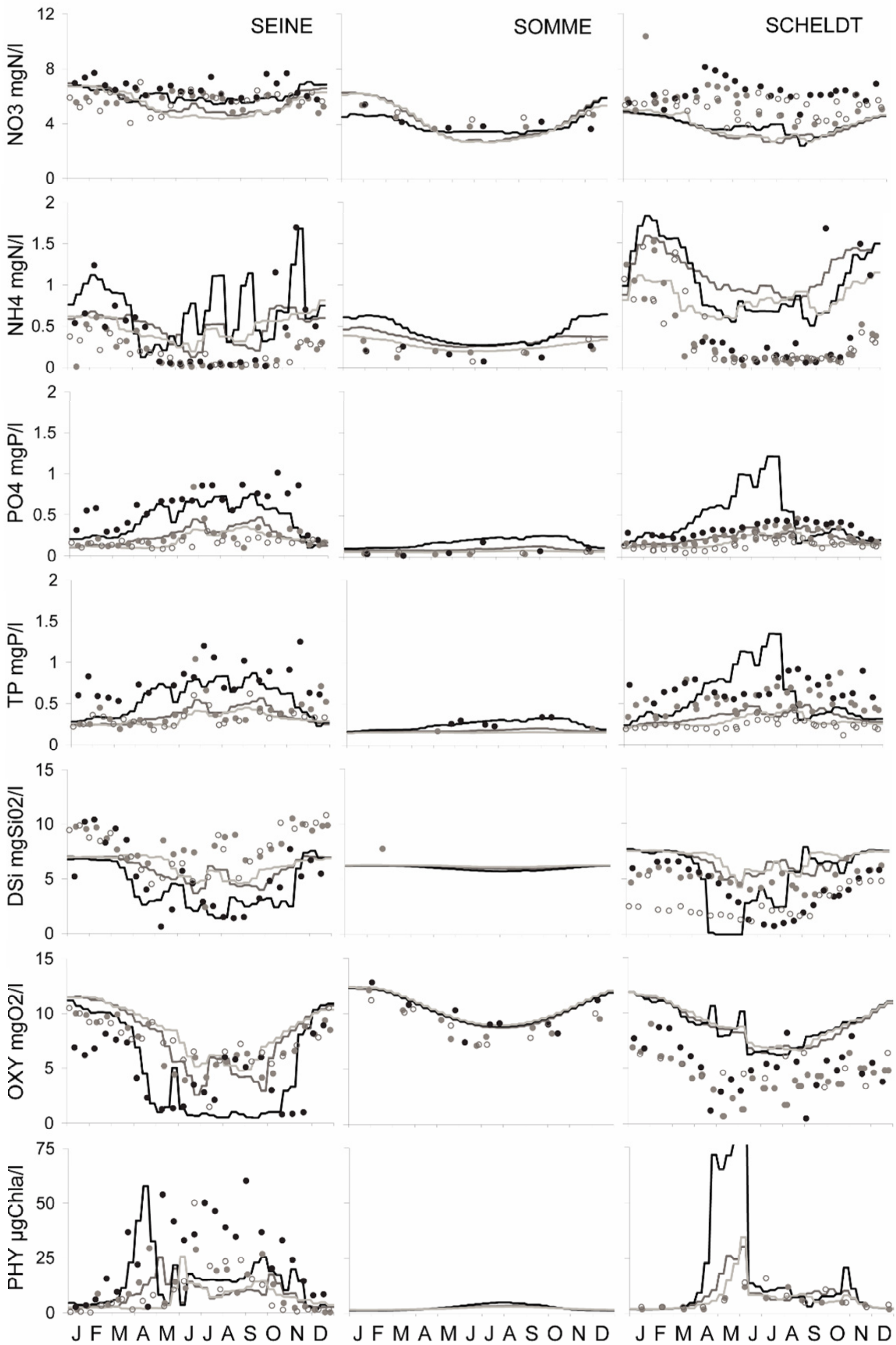

Figure 1-3: Results of simulations (line) and observations (circle) at the downstream station at Caudebec (Seine), Cambron (Somme), and Doel (Scheldt) in the dry year of 1996 (black), the hydrologically medium year 2000 (dark gray) and the wet year 2001 (light gray).

For ammonium and phosphate, seasonal variations are linked with the number of point sources, and concentrations are higher with lower discharge. These trends were correctly simulated by the model, except for the Scheldt in which phosphate levels during low water for the dry year (1996) were overestimated. This discrepancy is partially explained by the major changes that have occurred since 1996 with respect to the treatment of household effluents and to the method implemented to reconstruct the corresponding total untreated load. However, the model reproduced correctly the phosphorus profile of 2000 along the Senne affluent of the Scheldt, which receives two large urban discharges (Figure 1-4), and thus justifies use of the method for more-recent periods. Local analyses of point-source impacts on ammonium and phosphate concentrations were also carried out at the scale of the main branch of the Seine and the Somme Rivers, where the highest population density is found (Figure 1-4). This attests to the ability of the Seneque/Riverstrahler model to simulate correctly the impact of urban emissions on river-water quality once a high-resolution dataset with exact positioning of point sources is available.

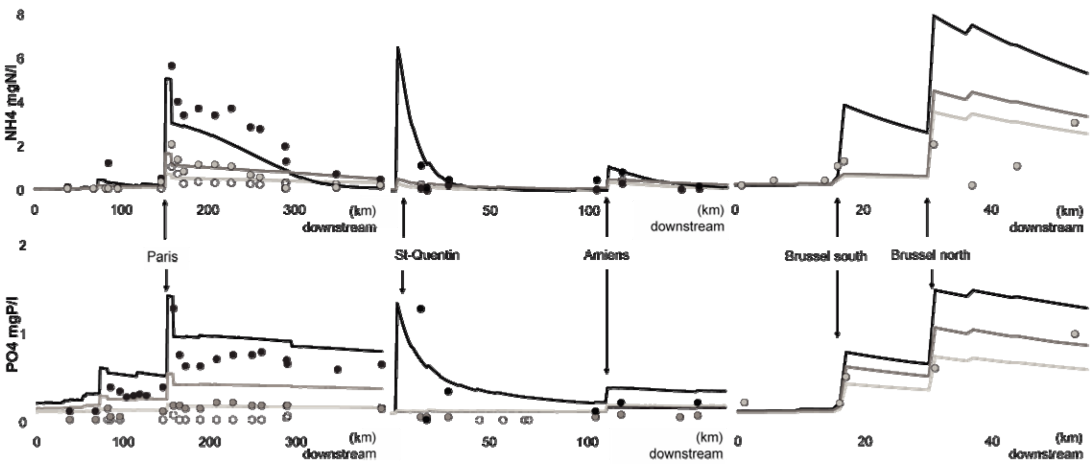

Figure 1-4: Comparison of the simulations (line) made by the Seneque/Riverstrahler model and the observations (circle) along longitudinal profiles in 1996 (black), 2000 (dark gray), and 2001 (light gray). Downstream part of the Seine (left), main axis from the upstream part to the outlet of the Somme (middle), and the Senne affluent of the Scheldt river (right) are shown.

With the aim of establishing nutrient budgets throughout the entire drainage network, the performance of the model had to be evaluated on the basis of nutrient flux predictions, especially at the outlet of the three basins. "Observed" nutrient fluxes were calculated as the product of discharge-weighted mean measured concentrations and mean discharges, according to the formula (Eq. 1-3) recommended by Moatar and Meybeck (2007), at the most downstream stations of the main tributaries of each river system:

$$F = \frac{\sum_{i=1}^{Ns} C_i Q_i}{\sum_{i=1}^{Ns} Q_i} \overline{Q}_r \qquad (1\text{-}3)$$

where C_i and Q_i are, respectively, the instantaneous values of river concentration and discharge at the day of sampling, N_s is the number of samples, and Q_r is the mean annual discharge as evaluated from daily discharge data.

For the sake of comparison, the fluxes were expressed per km^2 of watershed and compared with the values calculated by the model (Figure 1-5). Dissolved inorganic nitrogen (DIN) deliveries were correctly predicted, with a clear gradient related to discharge. The difficulties of the Riverstrahler model to simulate correctly the variability of particulate SM (brought in during wet conditions) points out a shortcoming of the model for total phosphorus. Conclusions regarding silica fluxes resulting from rock weathering were acceptable for the three contrasted hydrological years, with a slight underestimate for the wet year.

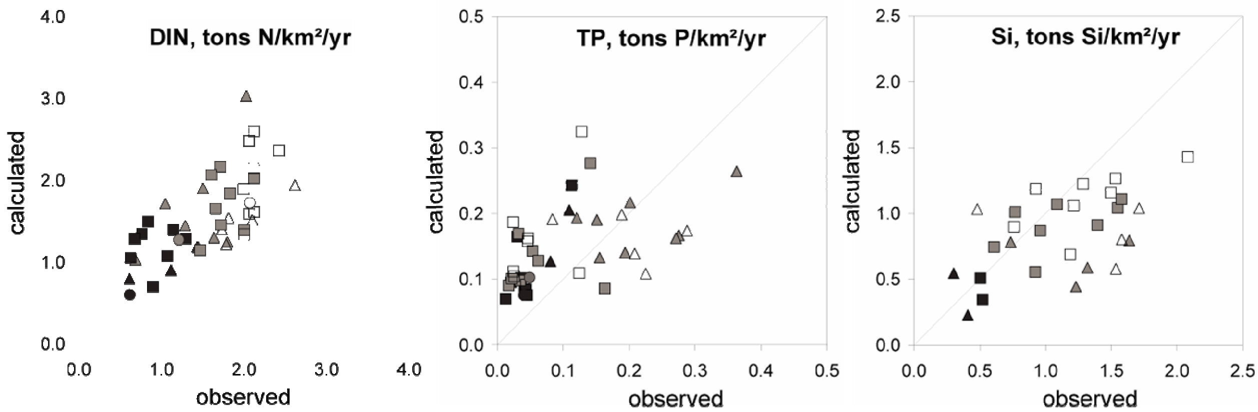

Figure 1-5: Comparison between model results (calculated) and data-based estimated fluxes of nutrients (observed) at several downstream stations of the Seine (square), Somme (circle), and Scheldt (triangle) drainage networks, in the years 1996 (black), 2000 (dark gray), and 2001 (light gray).

Heterogeneities in point sources and land-use distributions have led to different types of contamination of the three studied rivers. The Seneque interface of the Riverstrahler model is an optimized tool for assessing the spatial distribution of overall water quality.

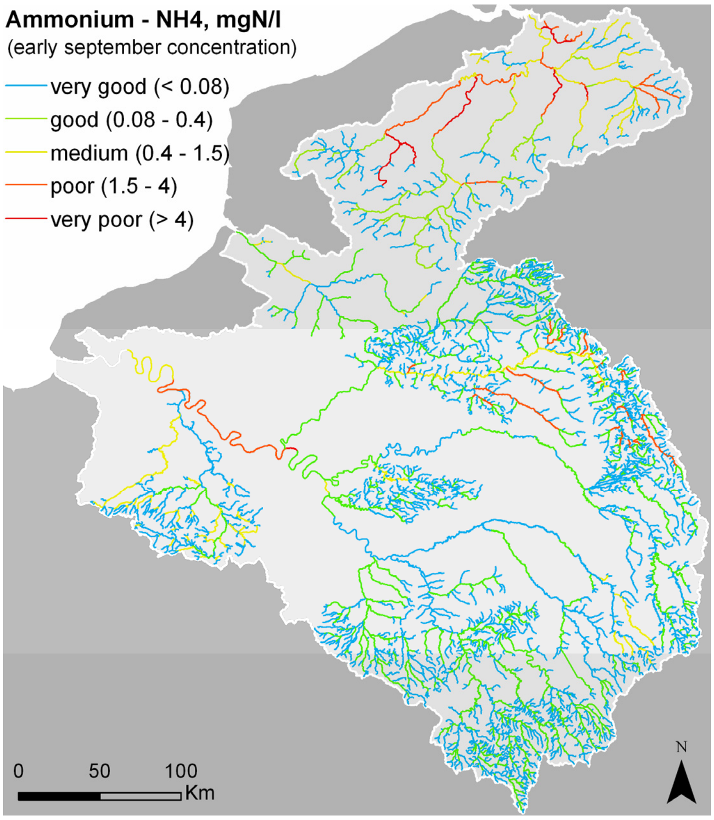

Figure 1-6: Distribution of ammonium concentrations over the three drainage networks beginning in September 2000, as calculated by the Seneque/Riverstrahler model.

In the case of ammonium contamination (Figure 1-6), the central position of the Paris conurbation in the Seine resulted in a shift of the water-quality status from medium (0.4–1.5mgN-NH_4/L) to very poor (>4mgN-NH_4/L). In the Scheldt basin, important urban emissions upstream have led to poor water quality (1.5–4 mgN-NH_4/L) starting from small-order streams and sustained in the downstream part by the overall high population density. In the Somme, upstream ammonium released by the St. Quentin wastewater treatment plant is rapidly transformed into nitrate, leading to very good water quality (below 0.08 mgN-NH_4/L) until the conurbation of Amiens, where the quality reverts to medium.

1.5. Computation of N, P, and Si budgets

The deterministic Seneque/Riverstrahler model was used to establish a detailed budget of nutrient transfer and retention on the scale of the entire drainage network. This budget (Table 1-4) quantified the respective contributions of reservoir, riparian or in-stream benthic retention processes, and the temporary storage of nitrate in aquifers. It also considered the year 2000 for point and non-point sources, as well as the hydrological conditions of the two contrasting years 1996 (dry) and 2001 (wet).

Table 1-4: N, P, Si transfer budgets calculated by the Riverstrahler model and ratios of the nutrients delivered yearly at the outlets of the Seine, Somme, and Scheldt Rivers, for wet and dry conditions. Redfield ratios are in parentheses.

	Seine				**Somme**				**Scheldt**			
Yearly hydrological conditions	dry		wet		dry		wet		dry		wet	
Nitrogen, kgN/km²/yr		*%*		*%*		*%*		*%*		*%*		*%*
Inputs												
Diffuse sources from watershed	1970	*78.1*	3961	*87.7*	1838	*90.8*	5571	*96.8*	1583	*61.0*	3087	*75.3*
Urban point sources	553	*21.9*	553	*12.3*	186	*9.2*	186	*3.2*	1013	*39.0*	1013	*24.7*
Outputs												
Delivery to coastal zone	1378	*53.7*	2311	*51.8*	639	*32.2*	1829	*31.1*	985	*44.0*	1568	*40.7*
Diversion & withdrawals	110	*4.3*	107	*2.4*	0	*0.0*	0	*0.0*	228	*10.2*	516	*13.4*
Storage in vadose zone and aquifers	356	*13.9*	707	*15.9*	427	*21.5*	877	*14.9*	193	*8.6*	392	*10.2*
Riparian retention	524	*20.4*	1161	*26.0*	885	*44.6*	3072	*52.2*	607	*27.1*	1167	*30.3*
Retention in reservoirs	13	*0.5*	40	*0.9*	-	-	-	-	-	-	-	-
In-stream benthic denitrification	119	*4.6*	65	*1.5*	24	*1.2*	27	*0.5*	174	*7.8*	126	*3.3*
In-stream retention in sediments	66	*2.6*	67	*1.5*	11	*0.6*	81	*1.4*	51	*2.3*	81	*2.1*
Phosphorus, kgP/km²/yr		*%*		*%*		*%*		*%*		*%*		*%*
Inputs												
Diffuse sources from watershed	46	*39.2*	92	*56.2*	20	*45.7*	96	*80.1*	64	*39.0*	117	*53.8*
Urban point sources	72	*60.8*	72	*43.8*	24	*54.3*	24	*19.9*	101	*61.0*	101	*46.2*
Outputs												
Delivery to coastal zone	87	*68.7*	100	*61.4*	31	*77.9*	61	*54.1*	74	*49.6*	96	*47.8*
Diversion & withdrawals	5.6	*4.4*	4.1	*2.5*	-	-	-	-	14.8	*9.9*	23.4	*11.6*
Retention in reservoirs	0.1	*0.1*	0.8	*0.5*	-	-	-	-	-	-	-	-
In-stream retention in sediments	34	*26.8*	58	*35.6*	9	*22.1*	52	*45.9*	61	*40.5*	82	*40.6*
Silica, kgSi/km²/yr		*%*		*%*		*%*		*%*		*%*		*%*
Inputs												
Diffuse sources from watershed	861	*94.3*	1688	*97.0*	445	*95.8*	1645	*98.8*	1027	*91.8*	1891	*95.4*
Urban point sources	52	*5.7*	52	*3.0*	20	*4.2*	20	*1.2*	91	*8.2*	91	*4.6*
Outputs												
Delivery to coastal zone	525	*60.5*	1143	*62.1*	363	*84.0*	1140	*66.0*	636	*59.5*	1157	*56.9*
Diversion & withdrawals	50	*5.7*	58	*3.2*	-	-	-	-	113	*10.6*	315	*15.5*
Retention in reservoirs	6	*0.7*	23	*1.3*	-	-	-	-	-	-	-	-
In-stream retention in sediments	287	*33.1*	615	*33.4*	69	*16.0*	588	*34.0*	319	*29.9*	561	*27.6*
Nutrients ratios												
N/P (Redfield ratio = 16)	35.0		51.0		45.4		66.5		30.1		38.7	
Si/N (Redfield ratio = 1)	0.2		0.2		0.3		0.3		0.3		0.4	
Si/P (Redfield ratio = 20)	6.7		12.6		12.9		20.7		9.3		13.7	
Limiting element (P or N)	P		P		P		P		P		P	
ICEP indicator	6.5		4.2		1.2		-0.2		5.4		4.3	

Within the three drainage networks, the Scheldt -with the highest population density and the lowest wastewater treatment efficiency- has proportionally the most important point-source inputs of nitrogen and phosphorus. Within the Somme basin, the dominance of agricultural lands make urban emissions of nitrogen insignificant (<10%). Even if their respective distributions widely vary, agricultural diffuse sources of nitrogen in the Scheldt are equivalent to those calculated for the Seine, ranging from 2,000 to 4,000 kgN/km^2/yr depending on the hydrological conditions. Silica production as a by-product of human activities was very low, such that inputs were dominated by diffuse contributions and controlled by hydrological conditions. Phosphorus point and non-point sources were well-balanced for the 3S, non point sources prevailing during dry years and the opposite during wet years.

Outputs from the Scheldt drainage network included those of the Leie tributary diversion, representing 10–15% of N, P, and Si outputs. It was assumed that the corresponding nutrient fluxes are discarded from the Scheldt system at its point of derivation in Gent, and that there is no contribution from any process occurring within the Schipdock channel.

The processes involved in phosphorus retention are the uptake by planktonic algae followed by immobilization within the sediment as well as the deposition of particulate phosphorus associated to SM having adsorbed dissolved phosphorus released by point sources. The incidence of the latter process increases by high discharge, such that benthic retention during wet years increased for the Seine (34–58 $kgP/km^2/yr$) and the Somme (9-52 $kgP/km^2/yr$). Specific benthic retention of phosphorous (61–82 $kgP/km^2/yr$) and nitrogen (51–81 $kgN/km^2/yr$) within the Scheldt drainage network was higher than in the two other basins.

This contradicts the view that higher retention is generally related to lower specific runoff, as suggested by the comparative study of Behrendt and Opitz (2000), because in this study specific runoff was higher in the Scheldt than in the other basins. However, this may have been related to the morphology of the Scheldt River, whose course is very flat, leading to lower flow velocities and increased sedimentation.

Regarding nitrogen, riparian processes are the most efficient retention mechanism, eliminating 25–50% of total nitrogen inputs, while in-stream processes such as benthic storage in sediments or denitrification contribute 2–10%. Denitrification in the Seine reservoirs is also quite important, with up to 50% removal of incoming nitrogen, i.e., close to the percentage retention based on experimental values (Garnier et al., 1999b). However, this denitrification only affects a small fraction of the river discharge. Temporary storage in aquifers represents a significant part of the apparent retention of nitrogen, with a mean 15% retention of the total nitrogen inputs from the watershed. Among the various processes accounting for nitrogen retention, riparian areas efficiencies integrate some of the uncertainty about the diffuse sources of nitrate.

The major uptake of dissolved silica is due to diatom blooms during springtime, such that the deposition of biogenic silica in the sediment accounts for the significant 15–30% retention of the total inputs.

Nutrient fluxes exported to the sea integrate the impact of hydrologic conditions, the respective importance of point and non-point sources, and the retention processes occurring within the drainage network, all of which, in turn, affect the ratio between nitrogen, phosphorus, and silica. Analyses of mean annual fluxes delivered to the coastal zone showed that Si/N and Si/P ratios are systematically below the requirements of diatoms, based on the Redfield ratios, except for the Somme during wet conditions (Table 1-4). This means that nitrogen and phosphorus are delivered in excess over silica. Intensification of agricultural practices and the recent prohibition of polyphosphates in washing powders have contributed to increase greatly the N/P ratio. Consequently, coastal marine phytoplankton production is now mainly controlled by phosphorus, which is the limiting nutrient at the outlet of the three basins. This nutrient imbalance at the coastal zone could promote the development of non-siliceous algae, with subsequent blooms having the negative impact of eutrophication (Billen and Garnier, 1997; Cugier et al., 2005; Lancelot et al., 2007). Based on nutrient fluxes and Redfield ratios, Billen and Garnier (2007) introduced the ICEP (indicator of coastal eutrophication potential), which, as shown in the following equation (Eq. 3), expresses the potential for new production of non-siliceous algae sustained by riverine deliveries:

$$\begin{aligned} ICEP &= [NFlx/(14*16) - SiFlx/(28*20)]*106*12 \qquad \text{if } N/P<16 \text{(N limiting)} \\ ICEP &= [PFlx/31 - SiFlx/(28*20)]*106*12 \qquad \text{if } N/P>16 \text{(P limiting)} \end{aligned} \tag{1-4}$$

where ICEP is expressed as carbon per day (kgC/km²/day), PFlx, NFlx, and SiFlx are, respectively, the mean specific fluxes of total nitrogen, total phosphorous, and dissolved silica (expressed as kgP/km²/day, kgN/km²/day, and kgSi/km²/day) delivered at the outlet of the river basin. A negative ICEP value, as for the Somme (-0.2 kgC/km²/day) during wet conditions, indicated that silica was present in excess over the limiting element (P); i.e., there was no "potential" eutrophication problem. On the contrary, positive values indicated that the amount of phosphorus (limiting nutrient) was in excess of the requirements for diatom growth, possibly leading to the development of non-siliceous algae, which are often responsible for harmful blooms. Wet conditions provide a trusty assessment of ICEP values that have been calculated from both model observations ($ICEP_{Seine}$ = 4.2; $ICEP_{Scheldt}$ = 5.4 kgC/km²/day) and observed nutrient fluxes ($ICEP_{Seine}$ = 4.0; $ICEP_{Scheldt}$ = 3.6 kgC/km²/day). During dry years ICEP values (observation: $ICEP_{Seine}$ = 6.5; $ICEP_{Scheldt}$ = 4.4 kgC/km²/day) tend to increase as nitrogen fluxes are reduced. This trend is well reproduced but affected by the underestimated TP simulations that lead to high ICEP value (simulation: $ICEP_{Seine}$ = 11; $ICEP_{Scheldt}$ = 13 kgC/km²/day) especially on the Scheldt.

Billen and Garnier, (2007) have proposed to calculate the ICEP for a number of rivers from different climatic regions of the world and for different time periods. Among these rivers, most of the temperate European and North American rivers have a positive value of ICEP and are know to suffer from eutrophication problem in their coastal areas. As for example the Gulf of Mexico under the influence of the Mississippi River or the North Adriatic Sea dominated by the inflow of the Po River with respectively ICEP values of 2 and 3.

1.6. Conclusions

Implementation of the Riverstrahler model has led to a comprehensive description of the ecological functioning of the Seine, Somme, and Scheldt Rivers systems.

The assumption of process unity throughout the three drainage networks and the complete control of nutrient fluxes by morphological or anthropogenic constraints, provides an elegant and reliable deterministic approach.

For the first time, the ability of the model to calculate a consistent and conservative budget of nutrient transfer and retention based on detailed calculations of all in-stream processes, has been demonstrated (with a mean of less than 5% discrepancy between total input and total calculated output).

The calculated imbalance of N,P and Si exports from the three watersheds have highlighted their potential to sustain the development of harmful algae in receiving coastal and marine areas.

As presently validated, the model - in its spatially implemented version (Seneque) - offers an optimized tool to explicitly assess the impact of any change in human activity and to investigate management scenarios for recovering well-balanced nutrient deliveries along the French and Belgian coastal zones.

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CHAPITRE

2 - Evaluation comparative du devenir de l'azote dans les basins versants

Modeling the N cascade in regional watersheds: the case study of the Seine, Somme and Scheldt rivers

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[1] UPMC Univ Paris 06, UMR 7619 Sisyphe, F-75005 Paris, France
[2] CNRS, UMR 7619 Sisyphe, F-75005 Paris, France

Abstract

The watersheds of the Seine, Somme and Scheldt rivers (France, Belgium, the Netherlands), flowing into the continental coastal zone of the English Channel and Southern North Sea, are among the regions of the world with the highest anthropogenic inputs of reactive nitrogen through fertilizer use, legume fixation and deposition of atmospheric nitrogen. They also represent examples of widely open systems, either exporting a large fraction of their N inputs under the form of agricultural products (case of the Seine basin) or importing high amounts of nitrogen as feed for livestock nutrition (case of the Scheldt basin), and delivering up to 2000 $kgN.km^{-2}.yr^{-1}$ at river outlet into the sea. Taking these three watersheds as a case study, we review the different approaches developed so far for describing and predicting the fate of reactive nitrogen inputs to regional systems and its cascade from soils to sea. These approaches range from simple lumped input-output budget, to detailed process-based, spatially distributed models of nutrient transfers. The merits and the limits of these approaches are discussed. Their combination allows to establish a reasonably consistent budget for the three basins, emphasizing the various 'retention' terms linked to both landscape and in-stream processes, including storage in long residence time compartments (soil organic matter, vadose zone, aquifers,…), denitrification (in soil, riparian zones or river benthos) or sediment burial. Root-zone and riparian denitrification processes appear as major terms of landscape retention in all three investigated watersheds. Retention of nitrogen associated with collection and treatment of urban wastewater is also a major term in the two most populated watersheds

2.1. Introduction

Environmental problems caused by excessive introduction of reactive nitrogen into the biosphere are a growing matter of concern at both the global and regional scales (Galloway and Cowling, 2002; Galloway et al., 2004). Nitrate contamination of drinking water resources, aquatic ecosystem eutrophication, emission of atmospheric pollutants (including ammonia and nitrogen oxides), reduction of biodiversity in natural and semi-natural areas receiving atmospheric nitrogen deposition, etc, are among the most severe interrelated problems caused by human induced perturbation of the nitrogen cycle.

The watersheds of the Seine, Somme and Scheldt rivers (France, Belgium, the Netherlands) together represent a contiguous area where a number of these environmental problems are particularly exacerbated in Europe and in the world, together with the Mississippi, the Pô and several others (Artioli et al., 2005; Justic et al., 2005). The Seine and the Somme basins are among the most productive agricultural areas in Europe in terms of yield, largely oriented toward cereal, rapeseed and sugar beet production, supported by intense application of synthetic nitrogen fertilizers (Mignolet et al., 2007). A large part of the Scheldt basin is devoted to intensive animal farming including dairy cattle, pig and poultry farming (Billen et al., 2005). These basins represent examples of widely open systems, either exporting a large fraction their N inputs under the form of crop production (case of the Seine and Somme basin) or importing high amounts of nitrogen as feed for livestock nutrition (case of the Scheldt basin). Nitrogen contamination of surface and groundwater is severe in all three basins(Billen et al., 2007b; 2005). The three rivers flow into the continental coastal zone of the English Channel and the Southern North Sea and contribute to the occurrence of harmful algal blooms, causing severe ecological damage (Cugier et al., 2005; Lancelot et al., 2007).

Several approaches have been developed for describing and predicting the fate of reactive nitrogen once introduced in the watersheds by synthetic fertilizer application, atmospheric deposition or legume crop nitrogen fixation, and to represent its cascade from soils to sea. The common purpose of these approaches is to relate the nitrogen inputs into the system to the outputs, resulting either from gaseous emissions or from river export to coastal areas, by taking into account the possible retention processes in several environmental compartments. In this paper, after a presentation of the major contrasting characteristics of Seine, Somme and Scheldt basins and their nitrogen budget, we evaluate the advantages and weaknesses of several of these approaches applied to the special case of these three watersheds.

2.2. Description of the three basins

The Seine, Somme and Scheldt basins drain the northern regions of France and Belgium, including the cities of Paris, Rouen, Amiens, Lille, Brussels and Antwerp (Figure 2-1a). Their watershed area, upstream from the last gauging station before their estuarine zone, are respectively 71 700 km² (downstream the Eure confluent at Poses), 5 570 km² (at Abbeville) and 19 860 km² (at Doel). Three large reservoirs, with a total volume of 750 10^6 m^3, have been established in the upstream part of the three main tributaries of the Seine River for the purpose of discharge regulation (Garnier et al., 1999b). No such infrastructure exists in the Somme and Scheldt basin. However, a significant part of the discharge of the Leie, a major tributary of the Scheldt basin, is diverted upstream from Ghent directly to the North Sea, through the Schipdonck channel (Figure 2-1a), (Billen et al., 2005).

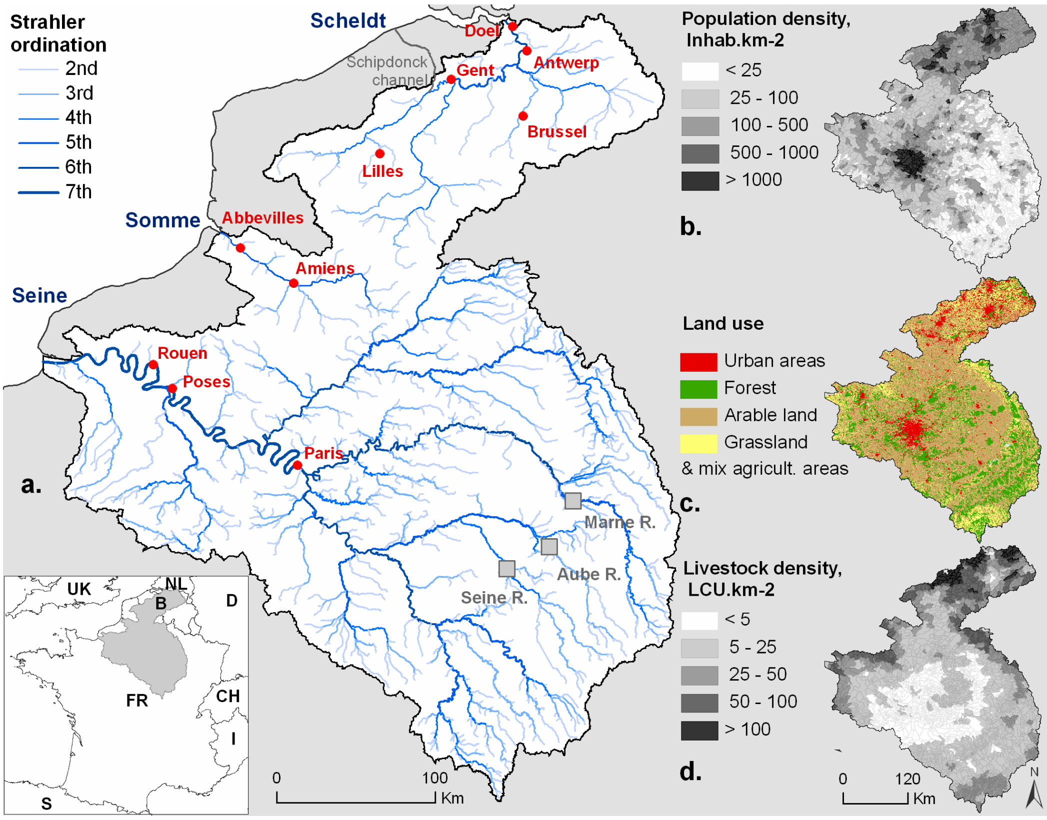


Figure 2-1: (a.) General map of the Seine, Somme en Scheldt basins and drainage network.(b.) Population density (data by cantons (France) and communes (Belgium) from the national statistics census).(c.) Land use (Corine Land Cover 2000, available on http://dataservice.eea.europa.eu/dataservice/). (d.) Livestock density (data by cantons from Agreste 2000 for France and by communes from INS for Belgium)

The population density of the Seine basin (mean 213 inhab.km^{-2}), with the huge Paris agglomeration (10 millions inhabitants) is twice that of the Somme basin (mean 99 inhab.km^{-2}). In the latter however, the population density is more uniformly distributed, with medium cities located upstream in the basin, while the population density of the upper Seine basin is very low (Figure 2-1b). The Scheldt basin (mean 497 inhab.km^{-2}) has the highest population density, twice higher than the Seine, with a rather uniform distribution (Figure 2-1b).

Land use (Figure 2-1c) is dominated by arable land in most of the Seine and Somme basins, excepted the eastern and western fringes of the former, while urban areas, as well as grassland and mixed agricultural areas dominates in the Scheldt basins; these differences are in accordance with the orientation of agriculture toward cereals and industrial crop production in the Seine and Somme basins, and toward livestock farming (including intensive pig and poultry production) in the northern part of the Scheldt basin. The distribution of livestock density, expressed in LCU (Large Cattle Unit = 1 milk cow, or 10 fattening pigs, or 143 laying hens, 333 battery chickens,…) displays the same features (Figure 2-1d, with a mean of 15, 23,70 LCU.km^{-2} for the Seine, Somme and Scheldt basins respectively.

A detailed description of urban and agricultural activities in the Seine and Scheldt basins, including their historical trends, can be found in Billen et al. (2001; 2007b for the Seine; 2005 for the Scheldt).

2.3. The input-output budget of the watersheds

Figure 2-2 offers a conceptual representation of the nitrogen flows through a regional watershed, which will be used as a support throughout this paper. Four major sub-systems within the watershed are distinguished (semi-naturals or forested areas, agricultural land, the urban system) and the hydrosystem.

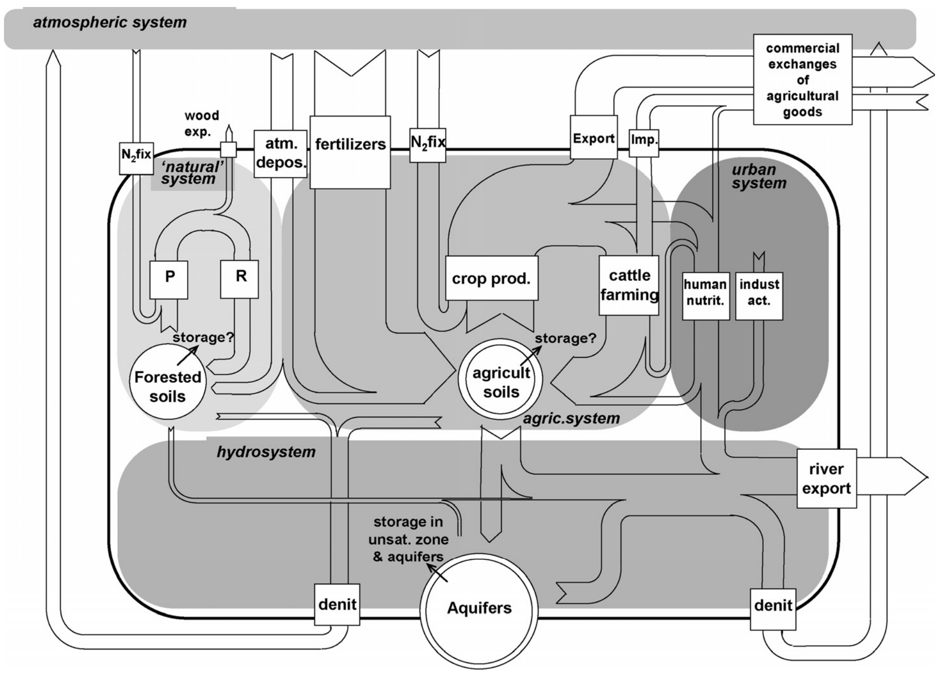


Figure 2-2: Conceptual representation of nitrogen fluxes through a watershed.

The major sources of reactive nitrogen to the watershed are atmospheric deposition, synthetic fertilizer application, biological N_2 fixation and commercial inputs of food and feed from other areas. Three major nitrogen outputs are considered: to the atmosphere in gaseous form, to other territories as food and feed, to the coastal seas as river loading. Soil organic matter and nitrate in deep aquifers are the major stocks where nitrogen can be either stored or remobilized.

Data on atmospheric deposition of nitrogen as nitrogen oxides and as ammonium are available from the calculation of the EMEP (2000) project. Owing to the much shorter residence time of NH_3 than NO_x in the atmosphere, we consider that a large part of deposited reduced nitrogen represents short distance re-deposition of emitted ammonia, so that we here only took into account the figures for oxidized nitrogen deposition (Figure 2-3), and calculated the corresponding annual deposition rate for the year 2000 by intersecting the EMEP grid with the 3 basins contours (Table 2-1).

Synthetic fertilizer application in the Seine and Somme catchment (as well as for the French part of the Scheldt basin) were deduced from the data base on agricultural practices obtained from direct enquiries by Mignolet et al. (2007) at the level of the Agricultural Districts (Table 2-1). For the Belgian part of the Scheldt basin, we made use of the figures of the 2000 national fertilizer application by culture communicated by EFMA (European Fertilizers Manufacturers Association).

Table 2-1: Input/output Nitrogen budget of the Seine, Somme and Scheldt watersheds for the medium hydrological year 2000, compared to that of Europe as a whole (excluding former Soviet Union, van Egmond et al, 2002; unpublished data for riverine delivery), of the Missisipi (USA, Boyer et al., 2006) and the Red River (China and Vietnam, Le et al., 2005).

	Seine	**Somme**	**Scheldt**	**Europe**	**Mississippi**	**Hong**
($kgN.km^{-2}.yr^{-1}$)						
oxN atmo.deposition(1)	550	530	800	770	495	610
synthetic fertilizer applic.	8950	12345	5540	2940	1840	1120
crop N_2 fixation	845	1095	535	460	1060	1105
Autotrophy	*6855*	*10500*	*5355*	*5235*	*3100*	*2090*
Heterotrophy	*2475*	*2590*	*9695*	*6560*	*435*	*2050*
net commercial inputs	-4380	-7910	4340	1325	-2665	-40
net total **N inputs**	5965	6060	11215	5500	730	2795
riverine total N delivery	1950	1430	2310	840	570	855
retention/elimination (diff)(2)	4015	4630	8905	4655	160	1940

(1) oxN : oxidized nitrogen forms

(2) diff : difference between net total inputs and riverine delivery

Crop N_2 fixation (Table 2- 1) was calculated from agricultural census of crop areas (Agreste, 2000 for the France; Statistiques Agricoles Belges, 2005 for Belgium) and considering a fixation rate of 150 $kgN.ha^{-1}.yr^{-1}$ for peas and beans, 250 $kgN.ha^{-1}.yr^{-1}$ for alfalfa, 150 $kgN.ha^{-1}.yr^{-1}$ for artificial meadows, 15 $kgN.ha^{-1}.yr^{-1}$ for permanent grassland and 5 $kgN.ha^{-1}.yr^{-1}$ for fallow. These values are those recommended by French agricultural authorities for calculating rational N fertilization rates and are within the range cited by Smil (1999).

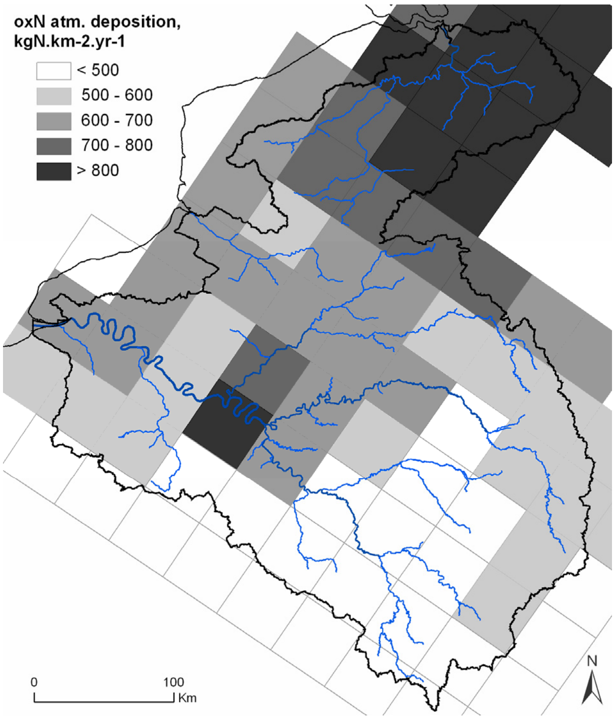


Figure 2-3: Distribution of total yearly atmospheric deposition of oxidized nitrogen (kgN/km²/yr) in the Seine, Somme and Scheldt watershed, according to EMEP (2000).

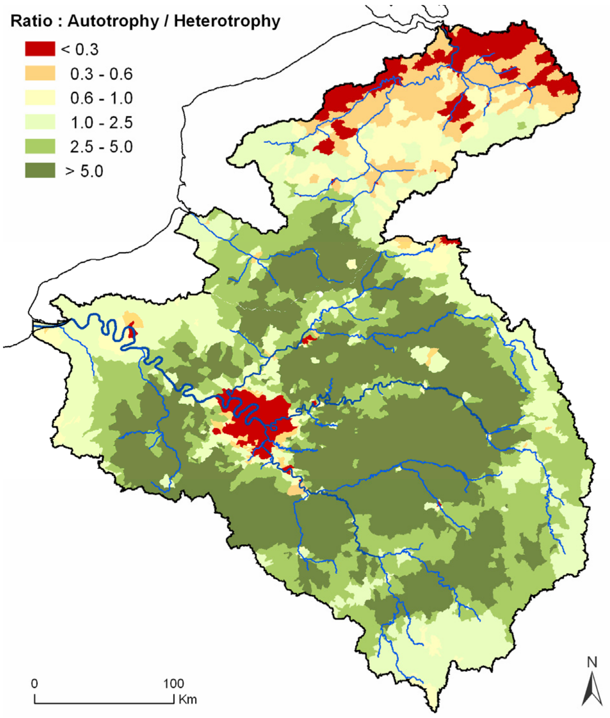


Figure 2-4: Distribution of the autotrophy: heterotrophy ratio (both calculated in $kgN.km^{-2}.yr^{-1}$ at the scale of the French cantons and Belgian communes) in the Seine, Somme and Scheldt watersheds. Autotrophic areas (local N crop production > organic N consumption) appears green, while heterotrophic areas are red or yellow.

Net commercial input/output of nitrogen in agricultural goods can be deduced from a budget of food and feed production by agriculture *versus* consumption by human and domestic animals. Such a budget defines whether a given territory is autotrophic or heterotrophic, in the sense that the yield of agriculture (autotrophy) exceeds or not consumption by man and livestock (heterotrophy) (see Billen et al., 2007a; Billen et al., 2005; Le et al., 2005). Cities (where food is consumed but not produced) are obviously heterotrophic. Most rural regions of the Somme and Seine basins are strongly autotrophic, while the eastern and western fringes of the Seine basin, as well as most of the Scheldt watershed, with intensive animal farming appear as heterotrophic, implying that they have to import feedstuff from outside (Figure 2-4). As a whole the Scheldt basin is strongly heterotrophic, while the Seine and Somme are autotrophic in average (Table 2-1). The fluxes of nitrogen associated with commercial import or export of food and feed in these basins are the second larger fluxes after synthetic fertilizers application. Only a limited part of the agricultural production is consumed locally in the Seine (36%) and Somme (25%); in the Scheldt basin, 45% of the food and feed consumed in the Scheldt basin is imported. This shows the extreme openness of the present agricultural market in Western Europe. Figure 2-5, based on agricultural data from the last half of the XXth century, shows that this phenomenon is rather recent and evidences the gradual opposite specialization of the agricultural system of the Seine and Scheldt basins since the last 50 years.

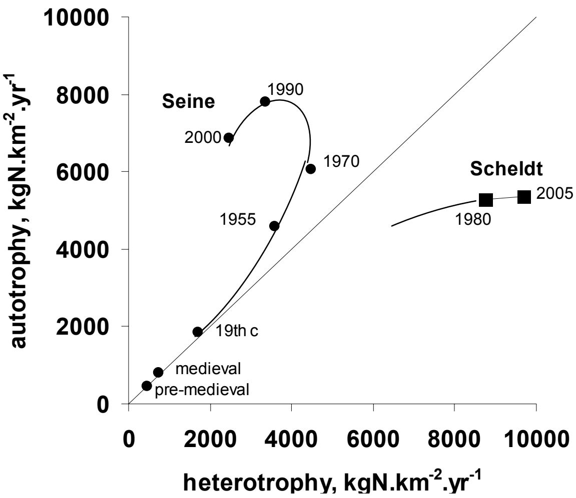


Figure 2-5: Past trends of the autotrophic/heterotrophic character of the Seine, Somme and Scheldt watershed, showing the opposite historical evolution of the 3 basins since the XXth century (historical data from Billen et al., 2008 for the Seine; from Billen et al, 1985 and De Becker, 1986 for the Scheldt).

Finally, water quality data available at the mouth of the Seine, Somme and Scheldt rivers allowed to calculate the riverine output of nitrogen for a medium hydrological conditions (2000), and dry (1996) and wet (2001) hydrology (Thieu et al., 2009). The mean of observed flux values of total dissolved inorganic nitrogen (DIN) for these three conditions is used here (Table 2-1). Based on a few available measurements of dissolved organic nitrogen, the fluxes of total nitrogen were estimated by increasing the DIN fluxes by 10%. The input-output budget of the three basins (Table 2-1) raises some remarks. First, the budget is not balanced, as no direct estimate of the output to the atmospheric N_2 pool or of the storage in soil and aquifers can be directly obtained. As a consequence, this 'retention/elimination' term of the budget, which obviously represents a major component in the N budget for the all 3 basins, has been estimated by difference in Table 2-1.

Further, although the three basins show a wide contrast in terms of the dominant N fluxes, they represent extreme situations when compared to the corresponding estimates for the mean European situation (total Europe excluding the former Soviet Union, based on data from van Egmond et al., 2002), for a North American watershed (Mississippi Boyer et al., 2006) or for a South Eastern Asian basin (The Red River Le et al., 2005) (Table 2-1). In particular, the specific fluxes of nitrogen exported by the three rivers to the sea (from 1430 to 2310 $kgN.km^{-2}.yr^{-1}$) are among the highest in the world.

2.4. The SCOPE/NANI approach

Howarth et al. (1996; 2006), Boyer et al. (2002) and Alexander et al. (2002) showed that the net sum of anthropogenic nitrogen inputs (NANI, $kgN.km^{-2}.yr^{-1}$) to a watershed (i.e. atmospheric deposition, synthetic fertilizer application, crop N_2 fixation, net import of food and feed) is an excellent predictor of the flux of riverine nitrogen exported at the outlet (Nflx, $kgN.km^{-2}.yr^{-1}$). Howarth et al. (2006) proposed the following general relationships:

$$\text{Nflx} = 0.26 * \text{NANI} + 107 \qquad (2\text{-}1)$$

The independent term (107 $kgN.km^{-2}.yr^{-1}$) would represent the 'background' or pristine nitrogen riverine export, while the factor 0.26 is the fraction of NANI that is transferred by the river at the outlet of the basin, implying that, as a mean, 74 % of the net sum of anthropogenically introduced nitrogen would be retained or eliminated in the watershed. Looking further to the variability of this 'retention' factor, Howarth et al (2006) found that it could be correlated with a simple climate variable such as precipitation or discharge (Q, $mm.yr^{-1}$). One of the empirical relationships they propose can be written:

$$\text{Retention} = 1 - \text{Nflx}/\text{NANI} = 1 - [\,0.00087 * Q - 0.096\,] \qquad (2\text{-}2)$$

This implies that 'retention' would decrease from 96 to 40 % when the specific discharge increases from 150 to 800 $mm.yr^{-1}$. The effect of specific discharge on N retention was already underlined by Behrendt and Opitz (2000). Schaefer and Alber (2007), by comparing the data from the Northeastern US coast of Boyer et al. (Boyer et al., 2002) with their own data from the Southeastern coast, have also suggested a positive control of temperature on watershed 'retention'.

The data available for the Seine, Somme and Scheldt basins and several of their sub-basins (Table 2-2) were used to test relationships (2-1) and (2-2) (Figure 2-6a, b).

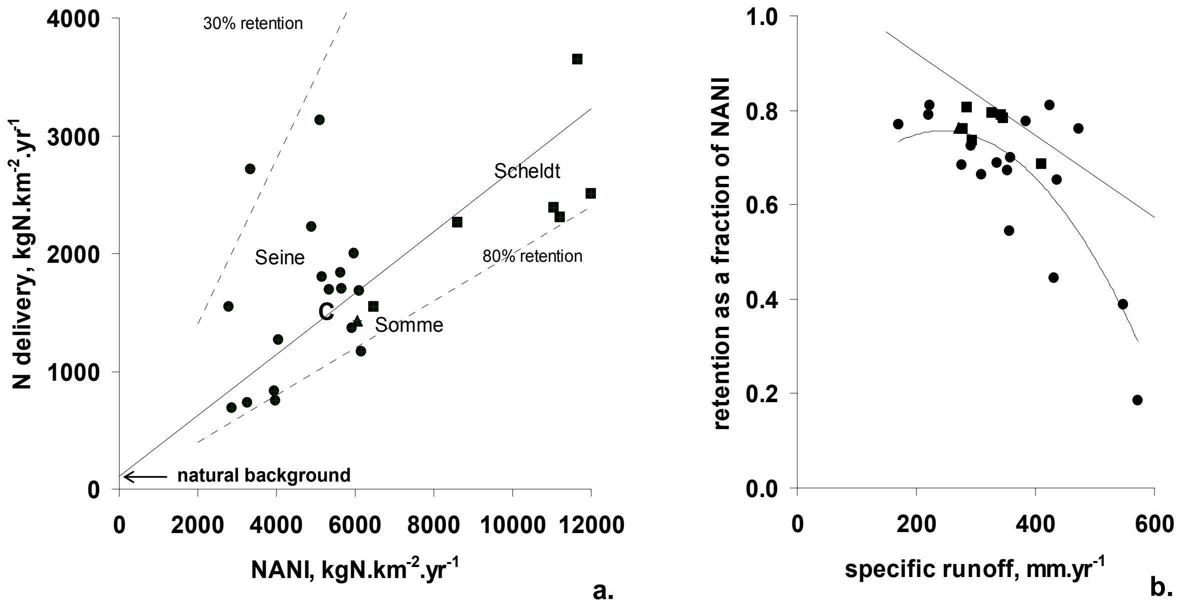


Figure 2-6: (a) Specific N flux delivered at the outlet of sub-basins of the Seine, Somme and Scheldt drainage network as a function of the net anthropogenic nitrogen inputs (NANI, $kgN.km^{-2}.yr^{-1}$) in their watershed. The solid line represents the relationship proposed by Howarth et al. (2006). The dotted lines respectively represent N retention of 30 and 80% of NANI. The best fit linear regression line gives the relationship: delivery = 834 $kgN.km^{-2}.yr^{-1}$ + 0.16 NANI (r^2=0.38)(circle: Seine; triangle: Somme, square: Scheldt).(b) Apparent retention of N in the watersheds, expressed as a fraction of NANI, in function of specific runoff ($mm.yr^{-1}$). The upper straight line is the linear relationship proposed by Boyer et al. (2006), while the lower curve is a best fit quadratic function through all the data (retention = 0.51 + 0.002. runoff - 0.000004. runoff ^{2} ; r^2=0.6)

The export fluxes were calculated from daily discharge measurements and monthly river water quality analysis over a 5 year period (1999-2003) for the Seine and Somme sub-basins, and over the years 1996, 2000 and 2001 for the Scheldt sub-basins. For the Scheldt the results were corrected for the diverted part of the Leie discharge, i.e. the diverted discharge and the corresponding N flux were added to the fluxes observed at the outlet of the sub-basins when these include the Leie basin. The results fit reasonably with relation (2-1), although a large dispersion is observed, with an apparent retention ranging from 30 to 80% of NANI. The regression line would provide the following relationship (2-3) although with a rather poor regression coefficient:

$$Nflx = 0.16 * NANI + 834 \qquad (r^2=0.38) \qquad (2\text{-}3)$$

Part of the variability can be explained by differences in specific runoff between the different sub-basins, as shown in Figure 2-6b: apparent retention decreases more rapidly with increasing runoff than predicted by relation (2-2).

$$\text{Retention} = 0.51 + 0.002\ Q - 0.000004\ Q^2 \quad (r^2=0.6) \qquad (2\text{-}4)$$

From this we conclude that the NANI approach is quite applicable to the Seine, Somme and Scheldt basins and sub-basins, provided the pronounced effect of specific runoff on nitrogen retention is taken into account.

2.5. The partitioned retention approach

While the SCOPE/NANI approach is based on a pure black-box input-output budget of the watershed, the SPARROW model (Spatially Referenced Regression on Watershed Attributes, Alexander et al., 2001; Alexander et al., 2000; Smith et al., 1997), as well as a number of other similar models like MONERIS (Behrendt et al., 2002a), GREEN (Geospatial Regression Equation for European Nutrient losses, Grizzetti and Bouraoui, 2006; Grizzetti et al., 2005) and NEWS-DIN (Dumont et al., 2005), are based on a distinction between point and non point sources of nitrogen from the watershed to the hydrosystem, which allows to differentiate between ‘in-stream’ and ‘landscape’ (or ‘terrestrial’) retention processes (the latter including retention in riparian wetlands). Basically, these models express the annual riverine nitrogen export at the outlet of a sub-catchment (Nflx, $kgN.km^{-2}.yr^{-1}$) as a function of (i) point sources (pointS), affected by river retention only, and (ii) diffuse sources (diffS, affected by both landscape and in-stream retention:

$$\text{Nflx} = (1 - \text{in-stream ret}) \cdot [(1 - \text{landscape ret}) \cdot \text{diffS} + \text{pointS}] \qquad (2\text{-}5)$$

Diffuse sources are defined as the excess of inputs to the terrestrial watershed as atmospheric deposition, synthetic fertilizer, manure and crop N_2 fixation over agricultural production.

In-stream and landscape retentions are arbitrarily parameterized as functions of some morphological or hydrological properties of the drainage network and of the watershed respectively. Thus in-stream retention is exponentially related to residence time in the drainage network and landscape retention is inversely related to runoff. The parameters of these relationships are calibrated by suitable statistical procedures, which finally allow to assess both landscape and in-stream retentions.

The data available on the Seine, Somme and Scheldt sub-basins (Table 2-2) offer a good opportunity to test this approach, as they include much contrasted situations regarding to the respective contribution of point and non-point sources of nitrogen. We used the following parameterization of i) in-stream and ii) landscape retention (similar to the approach of Alexander et al., 2002; Grizzetti and Bouraoui, 2006):

i) $\text{In-stream ret} = 1 - \exp[-\alpha * (drdens * A) / Q]$ (2-6)

where *drdens* is the drainage density of the basin ($m.km^{-2}$), A is the weighted average wetted section (m^2) calculated for the whole river network and Q is the mean annual specific discharge at the outlet of the basin ($m^3.km^{-2}.yr^{-1}$), so that the expression (*drdens* * A) / Q represents the mean annual residence time of water in the drainage network.

ii) Landscape ret = β / Q (2-7)

where Q is the mean annual specific discharge ($m^3.km^{-2}.yr^{-1}$). Adjustment of the two parameters α et β to 50 yr^{-1} and 280 $m^3.km^{-2}.yr^{-1}$ respectively over the entire set of data, provides a reasonable fit of observed and calculated N fluxes at the outlet of the basins, with a coefficient of determination (r^2) of 0.52 and a Nash-Sutcliff efficiency of 0.64. This allows the separate estimation of overall in-stream *and* landscape retention for the Seine, Somme and Scheldt sub-basins (Table 2-3). According to this analysis, the overall landscape retention varies between 56 and 64% for the Seine and Scheldt basin respectively, while the in-stream retention varies from less than 10 up to 60% of total (point and non-point) inputs to the drainage network as a function of the size of the watershed (Figure 2-7).

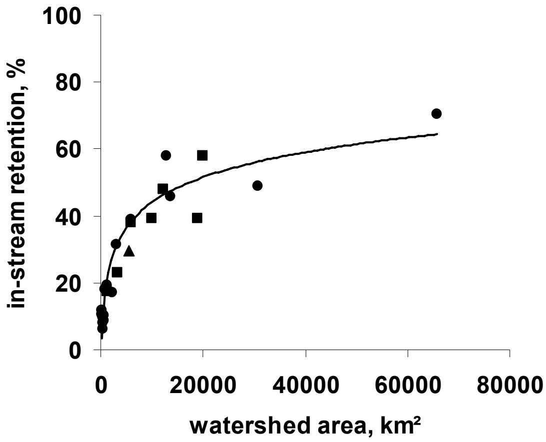


Figure 2-7: In-stream retention calculated according to the Sparrow-like approach in a number of sub-basins of the Seine, Somme and Scheldt watersheds, as a function of the basin area. (circle: Seine; triangle: Somme, square: Scheldt)

2.6. Process-based models

Although the above discussed regression approaches can be used at finer geographical scale, in fully distributed versions of the models (see e.g. Darracq and Destouni, 2005; de Wit, 2001), they leave several questions unanswered, about the exact nature of 'retention' processes and their precise location in the watershed. As they are based on time integrated data over several years of observations, they cannot reveal the effect of seasonal and year to year variability of hydrological conditions. Only mechanistic models, at finer temporal resolution, can address these issues. A number of detailed process-based models of soil-vegetation-water interactions have been developed addressing the issue of nitrogen transfers at the seasonal scale. Most of them however deal either with processes at the plot scale (e.g. STICS, Brisson et al., 2003; DNDC, Li, 2000; Li, 2001), or at the scale of small (< 100 km²) watersheds (e.g. SWAT, Neitsch et al., 2005; INCA, Wade et al., 2002; Whitehead et al., 1998).

The Seneque/Riverstrahler model (Billen and Garnier, 1999; Ruelland et al., 2007), is one of the few process-based, fully distributed models, with seasonal resolution, adapted to the scale of large regional watersheds. Chained with other models it can provide a comprehensive account of the nitrogen cascade from agricultural soil to the sea throughout the drainage network (Ducharne et al., 2007; Even et al., 2007).

The application of the Seneque/Riverstrahler approach involves the following steps:

1. The whole territory is divided into 'elementary watersheds' (EW), i.e. the catchment of a river stretched between two confluences. These constitute the spatial 'grid cell' of the model, and set its maximum spatial resolution.

2. Based on distributed meteorological data (daily rainfall and potential evapotranspiration), a hydrological model is adjusted on the basis of discharge data at gauged stations to provide the seasonal variations of surface runoff, infiltration and base flow, for each EW.

3. The distribution of land use classes (with corresponding agricultural practices) is defined for each EW and the corresponding mean sub-root water composition is determined either through empirical relationships or using a plot scale soil-plant-water model like STICS (Brisson et al., 2003). The mean sub-root water composition of the grid cell, taking into account its land use distribution, is assigned to the surface flow into the drainage network, and to the infiltration flow into the aquifer.

The base flow has the composition of deep groundwater (aquifer), which might be different from the infiltration flow when the aquifers have not yet reached their equilibrium with land use. Surveys of groundwater quality or model calculation (Ledoux et al., 2007) are used to assign a groundwater composition to each grid cell.

5. The point sources of urban wastewater are considered at their exact input location in the drainage network, based on the census provided by Water Authorities

6. The model then explicitly calculates the various biogeochemical processes affecting nutrients both in the water column and the benthic phase during the downstream transfer of water masses throughout the whole drainage network. As far as nitrogen is concerned, it namely involves algal uptake, planktonic and benthic ammonification, nitrification and denitrification, and permanent burial in sediments.

The application of this procedure to the Seine, Somme and Scheldt watersheds is described in full details by Thieu et al. (2009). Their calculated budget of nitrogen transfer for representative recent wet and dry hydrological conditions are shown in Figure 2-8.

Table 2-2: Morphological characteristics of a number of sub-basins of various sizes selected within the Seine, Somme and Scheldt watersheds and associated N cycle.

bassins	area km²	pop.dens. inhab/km²	urban %	spec N Flx kgN.km²/yr	spec runoff mm/yr	drain dens m/km²	wet sect* m²
Seine at Poses	65690	213	88	2004	310	305	25.1
Seine at Alfortville	30712	101	78	1692	277	298	12.8
Oise at Creil	13563	71	63	1840	353	348	12.6
Marne at Noisiel	12832	163	85	1702	358	333	19.0
Eure at Lery	6002	100	63	1363	171	239	7.2
Armançon at outlet	2983	27	36	2225	356	385	7.1
Saulx at outlet	2140	34	51	1799	436	353	4.7
Aube at Bar-sur-Aube	1291	14	20	1549	432	265	7.2
Therain at outlet	1215	130	62	1686	293	253	5.0
Yonne at Dornecy	757	16	8	751	425	566	3.1
Rognon at Donjeux	629	17	6	2719	572	259	4.9
Cure at Saint Père	563	12	0	689	473	555	1.6
Blaise at Wassy	389	12	4	3128	547	198	3.6
Marne at Langres	368	49	61	1265	336	475	1.5
Cousin at Avallon	350	36	58	729	384	513	1.3
Serre at Chaourse	251	25	25	1170	222	404	1.4
Armancon at Brianny	230	16	0	828	221	327	1.5
Somme at Abbeville	5566	99	62	1431	271	115	16.9
Scheldt at Doel	19860	497	100	2311	327	164	35.1
Scheldt at Temse	12306	441	100	2508	343	159	28.9
Scheldt at Schelle	18990	476	100	2390	346	165	21.3
Scheldt at Melle	10015	444	100	2338	285	160	18.1
Scheldt at Asper	5959	348	100	2266	294	158	18.2
Dijle at Haacht	3292	352	100	1546	278	184	8.1
Zenne at Eppegem	1137	1286	100	3649	410	160	10.1

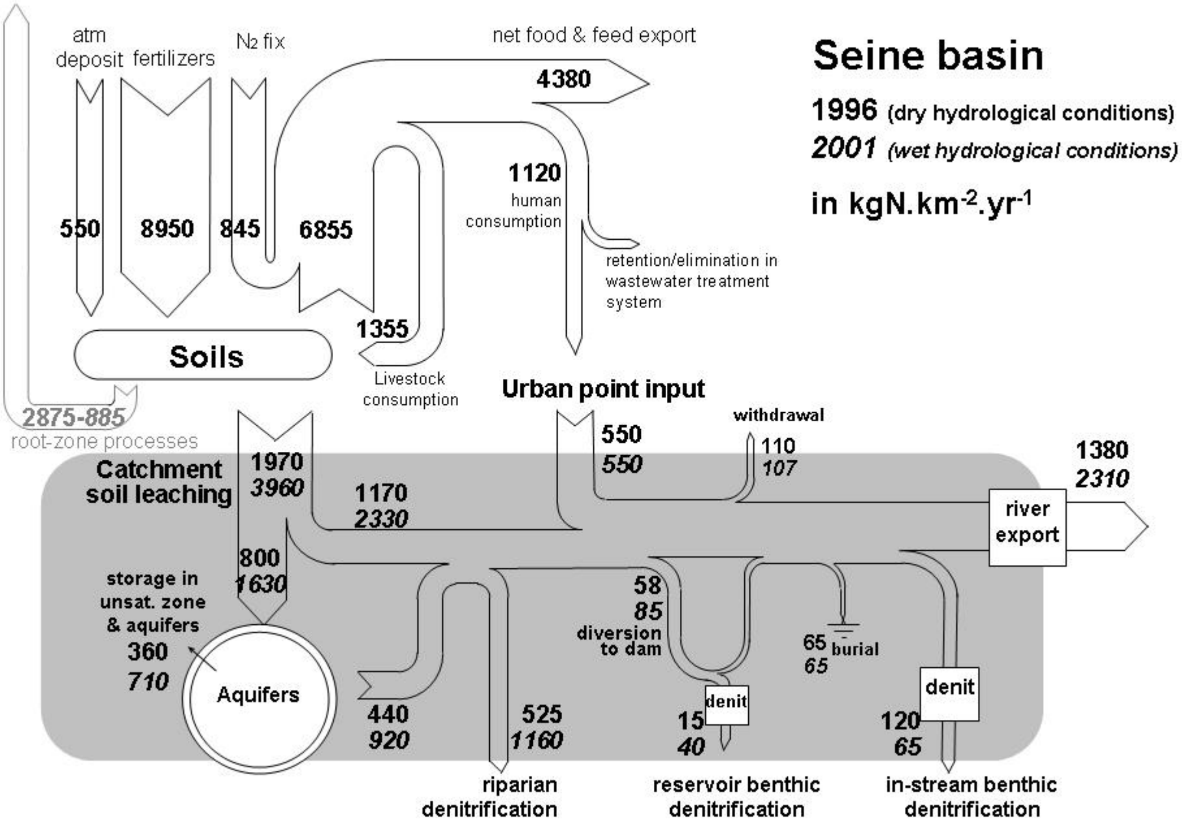
Strictly speaking the Seneque/Riverstrahler model is a drainage network model, representing the kinetics of in-stream (planktonic and benthic) processes, but not the processes occurring in watershed soils. However, the procedure used to define diffuse inputs, which distinguishes among (sub)-surface runoff and base (aquifer) flow, allows the estimation of the retention of nitrogen in groundwater, based on the difference in nitrate concentrations in infiltrating sub-root water and in base flow groundwater. This long term storage of nitrogen in aquifers represents, according to the hydrological conditions, 360 – 710 $kgN.km^{-2}.yr^{-1}$, 425 – 875 $kgN.km^{-2}.yr^{-1}$, 195 - 390 $kgN.km^{-2}.yr^{-1}$ in the Seine, Somme and Scheldt respectively, corresponding to the well documented increase of nitrate contamination of groundwater in the aquifers of these basins (Billen and Garnier, 1999; Billen et al., 2005; Ledoux et al., 2007). The model also incorporates an adjusted term of 'riparian retention' (Billen and Garnier, 1999; Sebilo et al., 2003) which accounts for the fact that dissolved nitrate in water at the base of the root zone may undergo significant denitrification processes, namely in riparian wetlands or in waterlogged down-slope areas of the watershed, before reaching surface waters.

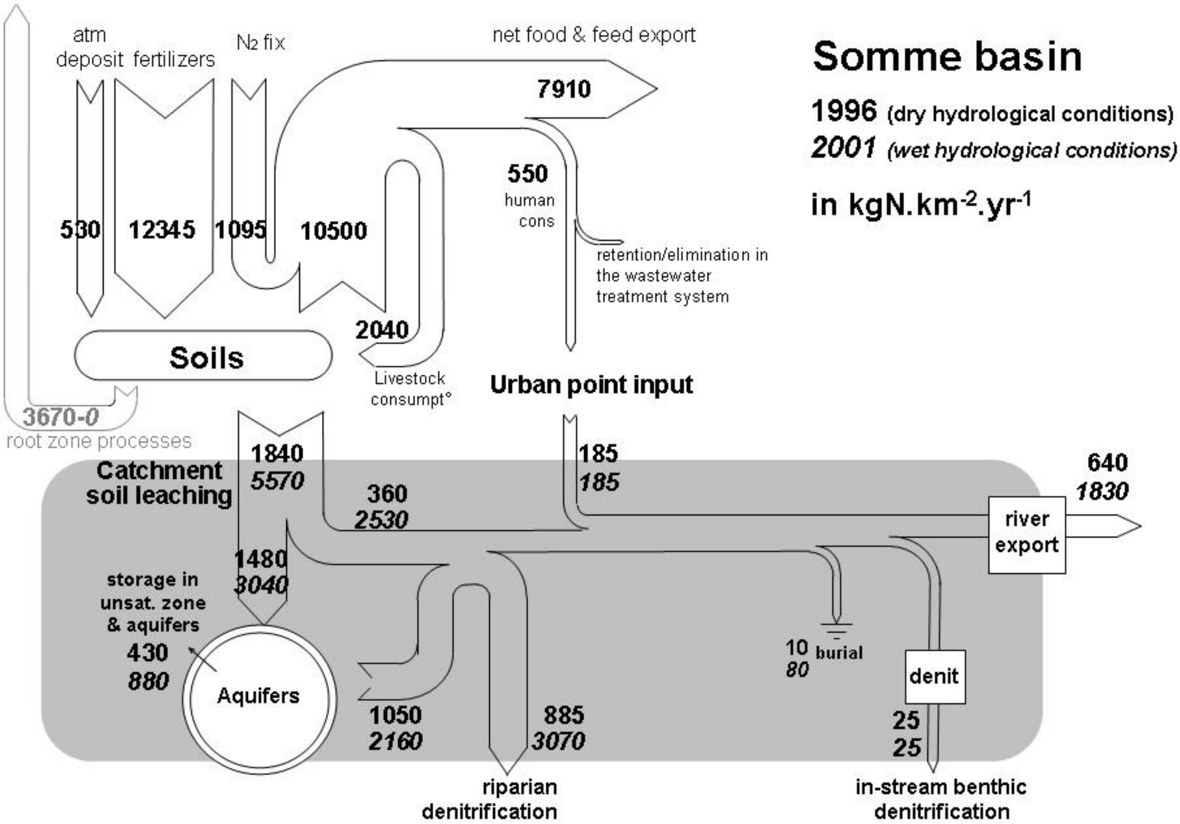
Table 2-2 (continue)

crop prod kgN/km²/yr	atm N2 fix kgN/km²/yr	inorg fertil kgN/km²/yr	atm depos kgN/km²/yr	human cons kgN/km²/yr	livestock cons kgN/km²/yr	food/feed imp kgN/km²/yr	total NANI kgN/km²/yr	retention wrt NANI %
6736	841	7297	550	1172	1389	-4175	5972	66
6288	781	7291	450	557	1099	-4632	5349	68
7920	1016	7922	600	389	2035	-5496	5627	67
6793	877	7421	550	897	1231	-4665	5667	70
8136	897	9227	550	549	991	-6596	5923	77
5514	662	6031	450	146	1911	-3457	4892	55
5697	460	6542	550	188	1809	-3699	5161	65
4321	435	5000	450	77	1154	-3091	2794	45
8302	916	7610	650	709	3012	-4581	6118	72
4050	1402	2121	450	90	3547	-413	3984	81
5030	464	4883	550	94	2379	-2557	3340	19
2684	1105	1238	450	64	2457	-163	2878	76
5670	387	6671	550	63	1782	-3824	5118	39
5902	732	4267	550	271	3288	-2343	4059	69
3396	872	1760	450	200	3015	-181	3253	78
9676	1202	7742	450	137	4754	-4784	6158	81
5290	1028	3467	450	92	3516	-1682	3955	79
10501	1093	10288	530	547	2044	-7911	6058	76
5356	535	4618	800	2735	6962	4341	11217	79
6341	589	5977	700	2427	7448	3534	11995	79
5481	539	4748	800	2617	6886	4023	11058	78
6633	594	6598	650	2444	7077	2888	12049	81
6770	572	6722	650	1913	4172	-684	8605	74
4344	278	2536	850	1936	4720	2312	6483	76
4349	287	2557	900	7076	4679	7406	11662	69

In the three basins, this appears as a very significant process, representing 525 – 1160 $kgN.km^{-2}.yr^{-1}$, 885 – 3070 $kgN.km^{-2}.yr^{-1}$, 610 – 1170 $kgN.km^{-2}.yr^{-1}$ in the Seine, Somme and Scheldt respectively. On the other hand, the model is particularly well suited to assess the effect of in-stream processes such as benthic denitrification or sediment storage in rivers as well as in reservoirs. Benthic denitrification is the dominant process, accounting for 135 – 105 $kgN.km^{-2}.yr^{-1}$, 25 $kgN.km^{-2}.yr^{-1}$, 175 – 125 $kgN.km^{-2}.yr^{-1}$ in the Seine, Somme and Scheldt respectively, while burial of particulate nitrogen adds 10 – 80 $kgN.km^{-2}.yr^{-1}$ to the in-stream retention terms. Thus, in contrast to the conclusion of the above partitioned retention regression approach, in-stream retention as calculated by the Seneque/Riverstrahler approach is one order of magnitude lower than the sum of aquifer and vadose zone storage and riparian retention, which should be considered as components of landscape retention.

Another important conclusion from the process-based modelling approach is the importance of hydrological conditions in controlling retention processes, the yearly average of which can vary by a factor of two between wet and dry situations.


Seine basin
1996 (dry hydrological conditions)
2001 (wet hydrological conditions)
in kgN.km-2.yr-1
atm deposit
fertilizers
N2 fix
net food & feed export
4380
550
8950
845
6855
1120
human consumption
retention/elimination in wastewater treatment system
1355
Livestock consumption
Soils
2875-885
root-zone processes
Urban point input
withdrawal
Catchment soil leaching
1970
3960
1170
2330
550
550
110
107
1380
2310
river export
800
1630
storage in unsat. zone & aquifers
360
710
Aquifers
58
85
diversion to dam
65
65
burial
denit
denit
440
920
525
1160
15
40
120
65
riparian denitrification
reservoir benthic denitrification
in-stream benthic denitrification


Somme basin
1996 (dry hydrological conditions)
2001 (wet hydrological conditions)
in kgN.km-2.yr-1
atm deposit
fertilizers
N2 fix
net food & feed export
7910
530
12345
1095
10500
550
human cons
retention/elimination in the wastewater treatment system
2040
Livestock consumpt°
Soils
3670-0
root zone processes
Urban point input
Catchment soil leaching
1840
5570
360
2530
185
185
640
1830
river export
1480
3040
storage in unsat. zone & aquifers
430
880
Aquifers
10
80
burial
denit
1050
2160
885
3070
25
25
riparian denitrification
in-stream benthic denitrification

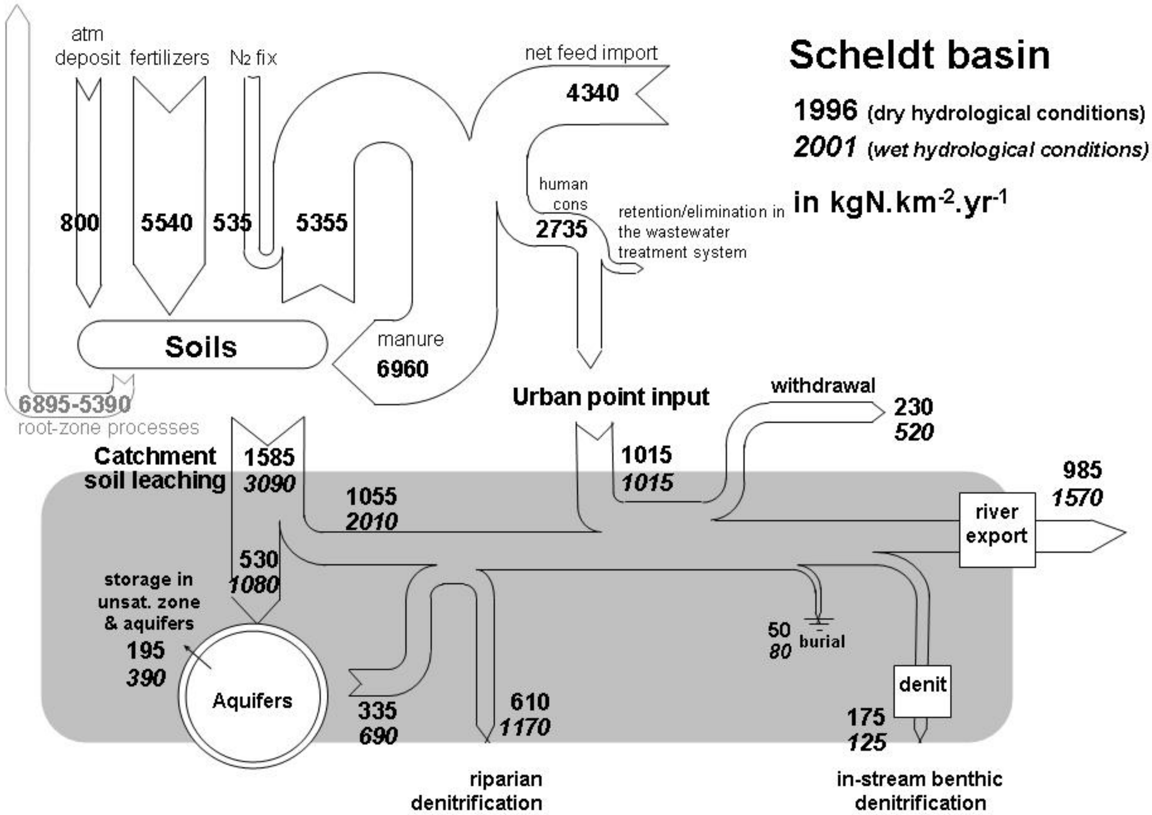


Figure 2-8: Budget of nitrogen transfers and transformations in the Seine, Somme and Scheldt drainage networks as calculated by the Seneque/Riverstrahler model for recent typical wet (2001) and dry (1996) hydrological conditions (results from Thieu et al., 2009).The main nitrogen fluxes to the terrestrial watershed (see Table 2-1) are also shown for comparison.

2.7. Discussion

The nitrogen cascade (Galloway et al., 2003) is defined as the suite of transfer, transformation, immobilization and elimination processes undergone by reactive nitrogen, starting from its introduction by human action into watersheds, until it reaches the outlet of the basin. Although it might be misleading, the term 'retention' is widely used to designate the processes leading to either immobilize some reactive nitrogen by storage into a long residence time compartment, or to eliminate it by conversion into the non-reactive atmospheric form. Van Breemen et al. (2002) already addressed the question of the fate of the large fraction of nitrogen anthropogenically introduced into US East coast watersheds which is not delivered at the outlet of the basins. In this paper, we explored three types of approaches for describing and quantifying the nitrogen cascade and the associated retention processes at the scale of regional watersheds. The first one (the NANI approach) involves establishing an overall nitrogen budget based on available detailed statistical data over a few year period and trying to relate riverine delivery to the net input term. This allows an overall retention term to be defined for the whole basin, which appears to be related to the mean inter-annual runoff.

The partitioned retention approach has in common with the NANI one to be based on the analysis of the inter-annual average budget, but differs in that i) it distinguishes between diffuse inputs of nitrogen spread over the terrestrial basin and point inputs of nitrogen directly released into the drainage network, and ii) it separately assesses the retention linked to the landscape processes and those linked to processes occurring within the drainage network owing to a calibration procedure. Finally, the Seneque/Riverstrahler process-based modelling approach allows a detailed account of the processes occurring within the drainage network under actual, seasonally variable hydrological conditions, and can be used to estimate the range of annual nitrogen retention resulting from the processes of in-stream benthic denitrification, burial in sediments, riparian denitrification and storage in groundwater.

The three contiguous basins of the Seine, the Somme and the Scheldt rivers are among the areas of the world where the nitrogen cycle has been the most perturbed by human activity, with net anthropogenic nitrogen inputs over 6000 $kgN.km^{-2}.yr^{-1}$ and riverine delivery close to 2000 $kgN.km^{-2}.yr^{-1}$. The major differences among the three basin lies on the autotrophy of the Seine and Somme, exporting a large part of their agricultural production while the Scheldt is heterotrophic, importing large amounts of feed for livestock breeding. The inter-annual nitrogen budgets for individual sub-basins indicate an overall retention varying from 20 to 80% in relation to runoff (Figure 2-6). With the Seneque/Riverstrahler model, the total amount of nitrogen retained in the Seine, Somme and Scheldt basins is respectively estimated to 4015, 4630 and 8905 $kgN.km^{-2}.yr^{-1}$, i.e. 67 – 79 % of net anthropogenic nitrogen inputs (Table 2-1). Application of the partitioned retention regression approach to the different sub-basins suggest that landscape processes would retain 40 – 80 % of net inputs of nitrogen to the catchments' soils, while in-stream processes would represent 10 - 70 % of total (point and non-point) inputs to the drainage network (Table 2-2, Figure2-7).

For the whole Seine system this approach leads to estimate landscape retention to 2760 $kgN.km^{-2}.yr^{-1}$, and in-stream retention to 2510 $kgN.km^{-2}.yr^{-1}$. The values for the Somme would be 3110 and 1020 $kgN.km^{-2}.yr^{-1}$, and for the Scheldt river 3930 and 3850 $kgN.km^{-2}.yr^{-1}$, for landscape and in-stream retention respectively. Note that the total retention figures are not entirely consistent with those of the NANI budgets due to the different definitions of diffuse inputs for the two approaches. Moreover, the partitioned retention regression approach rests on a rather arbitrary parameterization of the retention of both landscape and in-stream retention terms. The Seneque/Riverstrahler approach also differs in the input terms, as it considers the drainage network only and it requires an independent estimate of nitrogen leaching from soils, distinguishing surface runoff and base flow. As mentioned above, this can be obtained from an associated soil model or from empirical data; it also uses a direct estimate of actual wastewater discharge into surface water rather than total excretion by urban population. The intensity of processes affecting nitrogen (and the other nutrients) is calculated under seasonally variable hydrological and meteorological conditions. Combined with the budget data, it leads to separately estimate three components of the landscape retention (i) soil root-zone processes, (ii) vadose and groundwater storage and (iii) riparian denitrification), (Figure 2-8), as well as two components of in-stream processes, (i) benthic denitrification and (ii) burial of particulate nitrogen in alluvial sediments.

Soil root-zone processes are estimated as the gap in the soil budget. They might represent soil denitrification or storage in soil organic matter reservoir (although the latter is unlikely in the Seine and Somme watersheds because soil organic matter content is generally considered as decreasing due to continuous cropping with few inputs of organic manure, Arrouays and Pelissier, 1994). These root-zone processes constitute the largest term of the landscape retention, followed by riparian retention and storage in groundwater. Total landscape retention is estimated by our model approach to 3760 - 2755 $kgN.km^{-2}.yr^{-1}$ in the Seine basin, according to wet or dry hydrological conditions, to 4980 – 3945 $kgN.km^{-2}.yr^{-1}$ in the Somme basin and to 7700 – 6950 $kgN.km^{-2}.yr^{-1}$ in the Scheldt basin.

Regarding in-stream retention, the values issued from the Seneque/Riverstrahler approach (i.e. 200 – 170 $kgN.km^{-2}.yr^{-1}$ for the Seine, 35 – 105 $kgN.km^{-2}.yr^{-1}$ for the Somme, and 225 – 205 $kgN.km^{-2}.yr^{-1}$ for the Scheldt) are low compared to those issued from the partitioned retention regression approach. This discrepancy can be partly explained by a different way of estimating point inputs. Indeed, the input values used by the Seneque/Riverstrahler model refer to actual direct discharge of nitrogen from wastewater, after treatment in sewage treatment facilities, as recorded by Water Authorities, the partitioned retention approach, as well as the NANI approach used an overall figure based on urban population. The difference between the two estimates corresponds to the amount of nitrogen removed or lost during the stages of wastewater collection and treatment (Figure 2-8), and, in a way, is included in the overall in-stream retention term issued from the partitioned retention analysis. This term is particularly important in the two most populated Seine and Scheldt watersheds (Table 2-2).

2.8. Conclusions

As a whole, the combination of the three approaches described in this paper provides a reasonably coherent account of the nitrogen cascade from watershed soils to the outlet of the basin (Table 2-2). It stresses the importance of root zone and riparian denitrification processes in the overall nitrogen retention. Some discrepancy remains however; in particular the partitioned retention approach yields higher in-stream retention than estimated by the Seneque/Riverstrahler model. The accuracy of these estimates at regional scale is obviously poor, not only because they rest on input data subject to large uncertainties, but also because they refer to processes which are intrinsically highly variable from year-to-year. Hydrological variations are the most important cause of variability. The Seneque/Riverstrahler model, the only of the three tested approaches able to account for them, shows that retention terms considerably vary between wet and dry years. Among the processes affecting the cascade of reactive nitrogen introduced in the biosphere, those related to landscape and in-stream retention clearly require further experimental, field and modelling research.

Table 2-3: Independent estimates of the intensity of various retention processes in the Seine, Somme and Scheldt watersheds (in $kgN.km^{-2}.yr^{-1}$) according to the NANI, the partitioned retention and the Seneque/Riverstrahler approaches. (Figures in brackets refers to estimations obtained by difference)

a. Seine basin		NANI	partitioned retention	Seneque/Riverstrahler
Landscape retention	Root-zone processes	4015	2760	(2875 -885)
	Groundwater storage			360 - 710
	Riparian processes			525 - 1160
In-stream retention	Sewage treatment		2510	(570)
	Benthic denitrification			135 - 105
	Sediment storage			65
b. Somme basin		NANI	partitioned retention	Seneque/Riverstrahler
Landscape retention	Root-zone processes	4630	3110	(3670 – 0)
	Groundwater storage			430 – 880
	Riparian processes			885 – 3070
In-stream retention	Sewage treatment		1021	(365)
	Benthic denitrification			25
	Sediment storage			10 – 80
c. Scheldt basin		NANI	partitioned retention	Seneque/Riverstrahler
Landscape retention	Root-zone processes	8905	3930	(6895 – 5390)
	Groundwater storage			195 – 390
	Riparian processes			610 – 1170
In-stream retention	Sewage treatment		3850	(1720)
	Benthic denitrification			175 – 125
	Sediment storage			50 - 80

Acknowledgements

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Deuxième partie

Analyse prospective d'une modification des usages dans les bassins versants

Maîtriser la contamination des eaux de surface afin de limiter l'enrichissement des zones côtières associées n'est pas une préoccupation nouvelle. Dès 1992, la convention de Paris (PARCOM 92/7), repris par la convention OSPAR, a émis des recommandations sur la limitation des émissions terrestres en provenance des bassins versants, afin de protéger l'Atlantique du Nord-Est et ses ressources. Cependant, il faudra attendre la Directive Cadre sur l'Eau (DCE, 2000/60EC) pour qu'une unité de gestion cohérente soit établie, celle des bassins versants. Bien qu'elle n'inclue pas l'espace marin dans son ensemble, la DCE intègre une vision moins sectorisée des problèmes, étendue aux « rivières, lacs, eaux côtières et souterraines », et est donc plus compatible avec l'idée d'un *continuum aquatique.*

Dans la partie précédente, nous avons pu mettre en évidence l'intérêt d'une approche intégrée des transferts de nutriments dans les bassins versants, au travers de l'identification des usages en place et des quantités de nutriments qu'ils émettent, mais également grâce à l'utilisation d'outils de modélisation permettant de simuler leur transport jusqu'à la côte. L'utilisation du modèle Riverstrahler, en s'appuyant sur une quantité importante d'informations spatialisées à l'échelle des trois bassins (cf. Chapitre 1), offre un cadre privilégié pour évaluer la réponse des systèmes aquatiques à une modification des *contraintes* environnementales.

Cette deuxième partie s'inscrit donc dans une démarche prospective qui vise à analyser la possibilité de rééquilibrer les quantités de nutriments exportées vers la zone côtière, grâce à une modification des usages dans les bassins de la Seine, de la Somme et l'Escaut. La réflexion porte également sur la définition des niveaux qu'il serait souhaitable d'atteindre à l'exutoire des trois bassins, au vu de la persistance de la contamination nitrique. Cette partie est composée de trois chapitres.

Le premier (chapitre 3), présente l'évaluation à l'aide du modèle Riverstrahler de l'impact de mesures *réalistes*, dérivées de la représentation détaillée des contraintes (cf. chapitre 1 et 2). Ces mesures concernent l'amélioration du traitement des rejets ponctuels et l'application de diverses mesures agro-environnementales dont la mise en œuvre est actuellement considérée par les autorités en charge de la gestion des eaux.

Le recours à des mesures plus radicales, dites *alternatives* est évoqué dans un second chapitre (chapitre 4), avec l'exemple de l'agriculture biologique, dont les potentialités sont analysées à l'échelle de notre cas d'étude.

Enfin, dans le cadre d'une prospective à plus long terme des *futurs changements écosystémiques* (Millenium Ecosystem Assessment), le dernier chapitre de cette partie (chapitre 5) analyse la possibilité d'intégrer localement des sorties de modèles globaux, seuls capables de prendre en compte des schémas de développement dont l'emprise tend à se mondialiser.

Il est ainsi établi que l'effort actuellement consenti par les gestionnaires pour améliorer le traitement du phosphore en station est nécessaire (pour la Baie de Seine, cf. Cugier et al., 2005), et qu'une généralisation de cette mesure peut permettre de réduire efficacement les niveaux de contamination estivale. Plus précisément, une application uniforme de cette mesure à l'ensemble des stations traitant plus de 20,000 équivalents habitants permettrait de ne pas dépasser 0.5-1 mg/l de phosphate en aval des grandes agglomérations, quelques soit les conditions hydrologiques. L'analyse des simulations faites à l'échelle globale par les modèles du groupe Global*NEWS*, soutient que la réduction des émissions de phosphore, observée sur les dernières décades, va se prolonger en 2030-2050. Cette réduction est simulée indépendamment des futurs schémas de développement économique et démographique qui se mettraient en place à l'échelle des bassins de la Seine, de la Somme et de l'Escaut, en suivant les quatre scénarios du Millenium Ecosystem Assessment (Global Orchestration, Order from Strengh, Techno Garden et Adapting Mosaic).

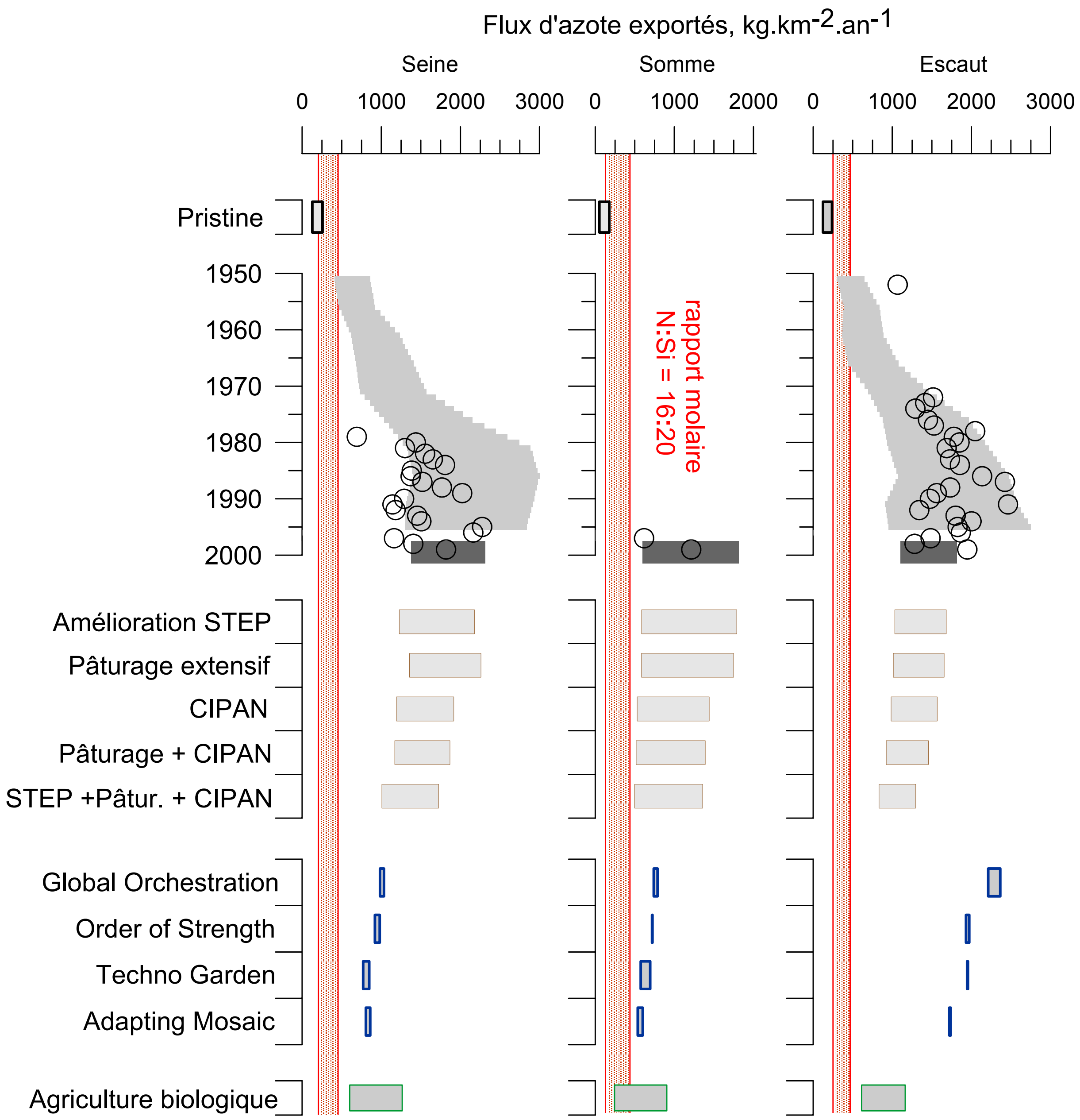

Figure II-1 : Impacts des scénarios analysés dans les chapitres 3 à 5 sur les quantités d'azote exportées à la zone côtière. Situation de référence, voir chapitre 1. econstitution historique pour la Seine et l'Escaut, voir Billen et al., (Billen et al., 2007b; 2005).

L'exploration des scénarios visant à réduire les quantités d'azote apportées à la mer (Figure II-1), indique les limites d'une gestion ciblant de façon préférentielle les apports ponctuels, par implémentation de traitements secondaire et tertiaire. En effet, une telle gestion ne peut être suffisante si l'on considère l'importance des apports diffus d'azote en lien avec les activités agricoles présentes sur les trois bassins (Partie I). En tenant compte de la diversité de ces usages, de leur intensité, ainsi que de leur répartition sur les bassins, une diminution de 13 à 27 % des quantités d'azote peut réalistement être atteinte à l'exutoire des trois bassins, en combinant l'introduction de cultures intermédiaires, la réduction de la fertilisation azotée et la substitution des cultures fourragères au profit de surfaces pâturées. Egalement, on estime que la conversion du système de production agricole actuelle, vers une agriculture biologique pourrait sur le long terme permettre une réduction de 25 à 67% des flux d'azote aux exutoires des trois bassins. L'évaluation des scénarios du Millenium Ecosystem Assessment prévoit une réduction possible des flux d'azote exportés de 29, 37 et 16% respectivement pour la Seine, la Somme et l'Escaut d'ici 2050 grâce des stratégies de gestion proactives des écosystèmes, et combinée à une vision du monde régionalisée, économiquement et politiquement (scénario Adapting Mosaic).

Les chapitres de cette seconde partie s'articulent avec des degrés de réalisme différents, mais tous s'accordent sur la nécessité des outils de modélisation pour évaluer l'impact d'une modification des usages sur les bassins. L'utilisation d'un modèle de réseau hydrographique, Riverstrahler, nous a amené à mobiliser des données expérimentales, des sorties de modèles agronomiques à l'échelle régionale et globale, afin d'intégrer le transfert des nutriments dans les parties terrestres des bassins. Cette méthodologie est tout à fait perfectible et pourra être actualisée si des informations plus précises (voir plus localisées) devenaient disponibles. Néanmoins, les résultats établis pour les trois bassins, soulignent la persistance de la contamination nitrique en lien avec des changements d'usage qui apparaissent difficilement réversibles.

A l'échelle de la zone côtière franco-belge, la capacité de ces quantités d'azote à soutenir des développements algaux nuisibles a été estimée par rapport aux flux de silice. En effet, l'épuisement de la silice, privilégie la croissance d'espèces non siliceuses, potentiellement nuisibles, qui se développent sur les quantités excédentaires d'azote essentiellement, et de phosphore dans une moindre mesure désormais. Les quantités d'azote requises pour soutenir une production diatomique (siliceuse) sont ainsi estimées entre 150-450 $kgN.km^{-2}.an^{-1}$, alors que la contribution propre des trois bassins est par ailleurs estimée entre 650-2300 $kgN.km^{-2}.an^{-1}$ pour la période récente. Aucune des mesures testées ne permet de diminuer les flux d'azote à de tels niveaux.

Cette quantification de l'excédent d'azote, par rapport au flux de silice - essentiellement terrestre et peu altéré - est un indicateur pertinent du potentiel d'eutrophisation des zones côtières réceptrices (Billen and Garnier, 2007). Certaines limites sont cependant à noter. Tout d'abord l'existence de développements algaux siliceux nuisibles, comme le genre Pseudo-Nitzschia dont certaines espèces toxiques sont observées depuis quelques années.

D'autre part, les flux de nutriments fluviaux sont calculés en amont de la zone de turbidité maximale et ne prennent pas en compte le rôle *filtre* des estuaires aval. Malgré des aménagements important qui ont contribué diminuer leur rôle de filtre (notamment l'estuaire Seine), la rétention au niveau de ces estuaires macrotidaux peut être importante durant l'étiage. On s'interroge alors sur la période à considérer pour caractériser l'impact des apports fluviaux sur les développements algaux côtiers.

Pour la Seine, les travaux de Cugier et al. (2005) mettent en évidence des blooms de Dinophysis (algues non siliceuse toxiques) principalement en fin d'été. Ces efflorescences sont par ailleurs restreintes au panache immédiat en Baie de Seine, où les temps de séjours ne dépassent pas quelques jours. Dans cette configuration, les flux annuels de nutriments ne sont pas suffisamment indicatifs et il sera nécessaire de raisonner sur des apports terrigènes saisonniers, qui, en étiage, sont susceptibles d'être réduits lors de leur passage dans le secteur estuarien avec l'allongement du temps de séjour et la remontée du bouchon vaseux dans le chenal de l'estuaire interne.

Le secteur de la Baie Sud de la Mer du Nord intègre la contribution de l'Escaut, mais également la remontée des apports fluviaux de la Seine et de la Somme, qui a lieu tout au long de l'année. Au niveau de cette zone côtière, le temps de séjour des masses d'eau douce peut être de plusieurs mois (Gypens et al., 2007; Lancelot et al., 2005). Les nutriments apportés durant l'hiver (période de débit important) forment l'essentiel du stock initial printanier et contribuent alors directement aux processus d'eutrophisation, caractérisés par des blooms de Phaeocystis aux mois d'avril-mai. Cette dynamique justifie l'utilisation des flux annuels de nutriments pour caractériser l'équilibre des apports à la zone côtière. On peut alors considérer qu'à l'échelle d'une année, la rétention estuarienne est négligeable.

Cependant, à l'issue de cette partie il n'est pas encore possible de conclure sur la nature et l'intensité des niveaux de développement algaux qui seront potentiellement atteint à la zone côtière. Seul un couplage du modèle des 3 réseaux hydrographiques avec un modèle intégrant les dynamiques du domaine côtier-marin, peut permettre d'apprécier l'efficacité des mesures implémentées à l'échelle des bassins versants, sur les proliférations algales indésirables à la zone côtière. Ce couplage fait l'objet de la troisième partie.

CHAPITRE

3 - Impact d'une réduction des apports fluviaux, sur les flux exportés à la zone côtière

Assessing the effect of nutrient mitigation measures in the watersheds of the Southern Bight of the North Sea

Thieu, V.[1], Garnier, J. [1,2] and Billen, G.[1,2] (2009)
Science of The Total Environment, in press, doi:10.1016/j.scitotenv.2009.12.031

[1] UPMC Univ Paris 06, UMR 7619 Sisyphe, F-75005 Paris, France
[2] CNRS, UMR 7619 Sisyphe, F-75005 Paris, France

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.

3.1. 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., 2007b). 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., 1998b).

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, 2000) 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., 2009b; Thieu et al., 2009). Based on the RIVERSTRAHLER model, these works provided accurate descriptions of the past and 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.

3.2. Methodology

3.2.1. 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 (Figure 3-1). 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 3-1) 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., 2007b) 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 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 km²) or the central part of the Seine are mostly dedicated to cereal and industrial crops, while the Scheldt (496 inhabitants per km²) and the western fringes of the Seine support intensive animal farming activity.

Table 3-1: 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, km²	76370	6190	19860
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

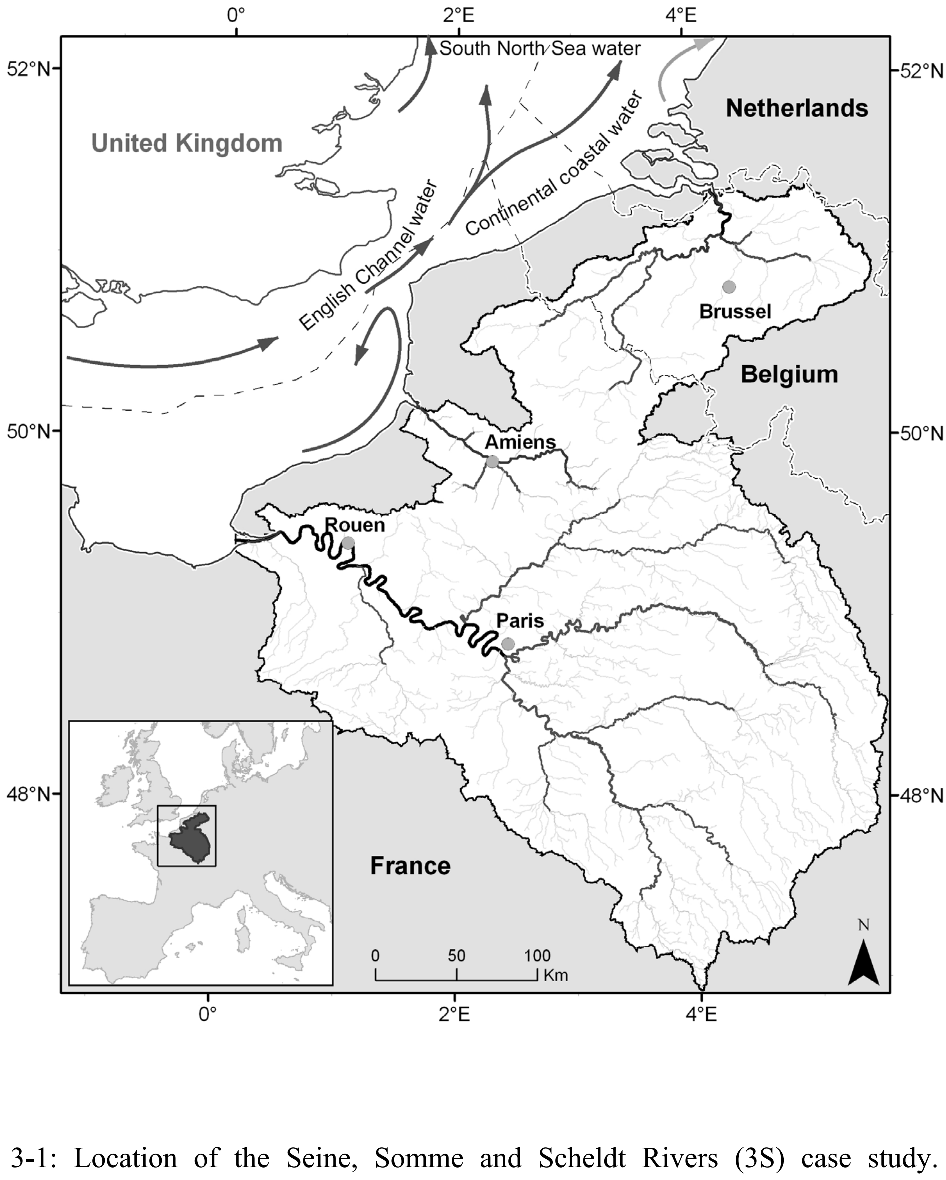

Figure 3-1: Location of the Seine, Somme and Scheldt Rivers (3S) case study. Boundaries of adjacent coastal waters shown for each country are those of the EEZ (Exclusive Economic Zone) limit areas. The schematic circulation of water masses in the southern North Sea is derived from Turrell et al. (1992).

3.2.2. 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 (MONERIS: Behrendt et al., 2002b; Howarth et al., 1996) to fully distributed approaches (GREEN: Grizzetti et al., 2005; SPAROW: 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. SWAT: Neitsch et al., 2005; INCA-model: Wade et al., 2005), our approach includes a multi-basins system exceeding over 100.000 km², in which we consider a combination of 46 “sub-basins” (1592 km² 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.

3.2.3. Definition of reference conditions

3.2.3.1. 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 (Figure 3-2). 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.

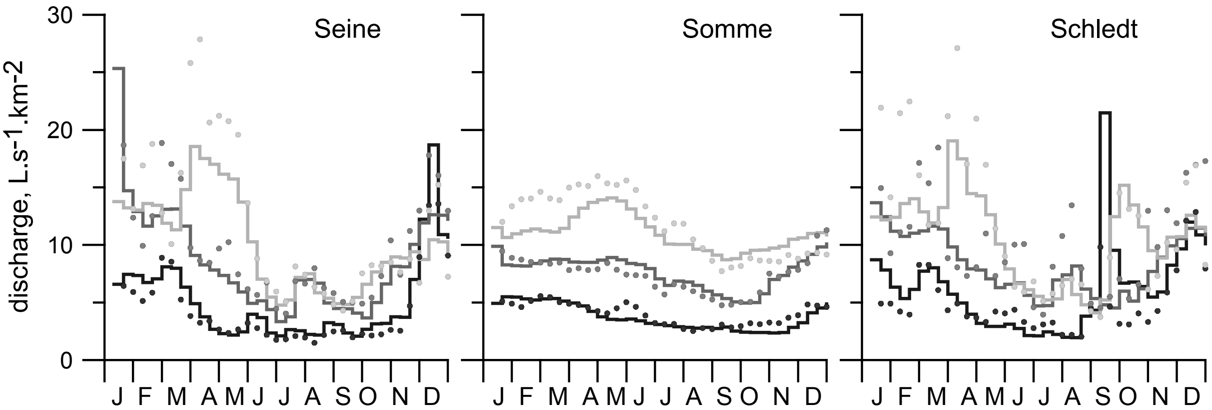

Figure 3-2: Specific runoff calculated at downstream stations of the Seine (Caudebec), Somme (Cambron), and Scheldt (Doel) Rivers for the years 1996 (dark grey lines), 2000 (grey line), and 2001 (light grey line). Observations are given for comparison.

3.2.3.2. 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} yr^{-1} (e.g. from 25 to 400 g C m-2 yr^{-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.

3.2.3.3. 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 (CLC: Bossard et al., 2000) and agricultural statistics (Agreste, 2000; 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 (Figure 3-3). 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).

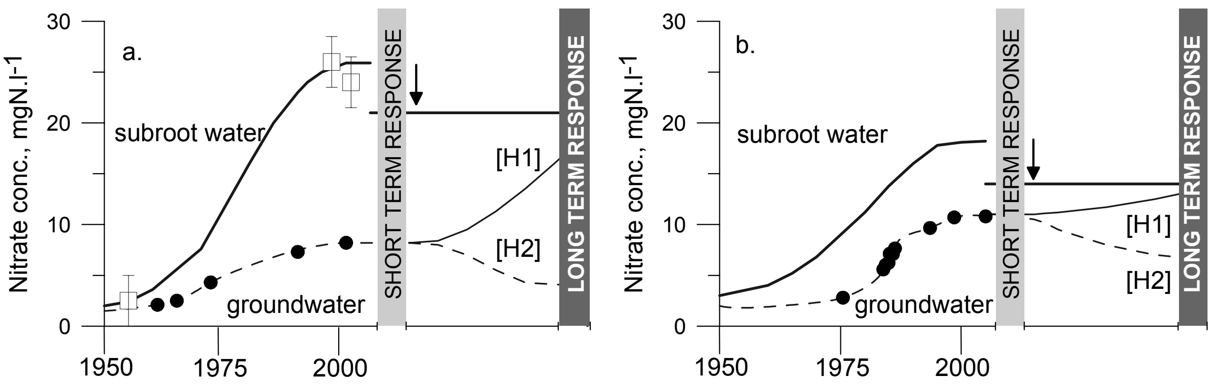

Figure 3-3: Evolution of sub-root (modeled) and groundwater (literature data) nitrate concentrations from 1950 to the present, and hypothetical responses to a simulated change in short-term and long-term land use as assumed in the modeling approach. A) Case of the chalky Champagne district, sub-root 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.

3.2.4. 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 (Directive 91/271/EC). 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.

3.2.4.1. 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.

3.2.4.2. 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 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 3-2. 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 (SS) 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.

Table 3-2: 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, mgN/l (sub-root)	NO3, mgN/l (sub-root)	Fodder, % (arable converted)	NO3, mgN/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, mgN/l (sub-root)	NO3, mgN/l (sub-root)	Fodder, % (arable converted)	NO3, mgN/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	-	-

3.2.4.3. 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 3-2.

3.2.4.4. 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 Figure 3-3. 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 (Figure 3-3).

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 different measures. The results in terms of nutrient inputs to the river system and export to the coastal zone are summarized in Figure 3-4.

3.3.1. Nutrient budgets in the present state

3.3.1.1. 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} yr^{-1} *vs.* only 553 kg N km^{-2} yr^{-1} for the Seine River). Within the Somme basin, urban emissions of nitrogen are comparatively low (186 kg N km^{-2} yr^{-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} yr^{-1} depending on wet or dry hydrological conditions, although the spatial distribution of these sources within the basins widely differs too.

3.3.1.2. Phosphorus sources

Point sources follow the population-density gradient, with mean emission values of 24, 72, and 101 kg P km^{-2} yr^{-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).

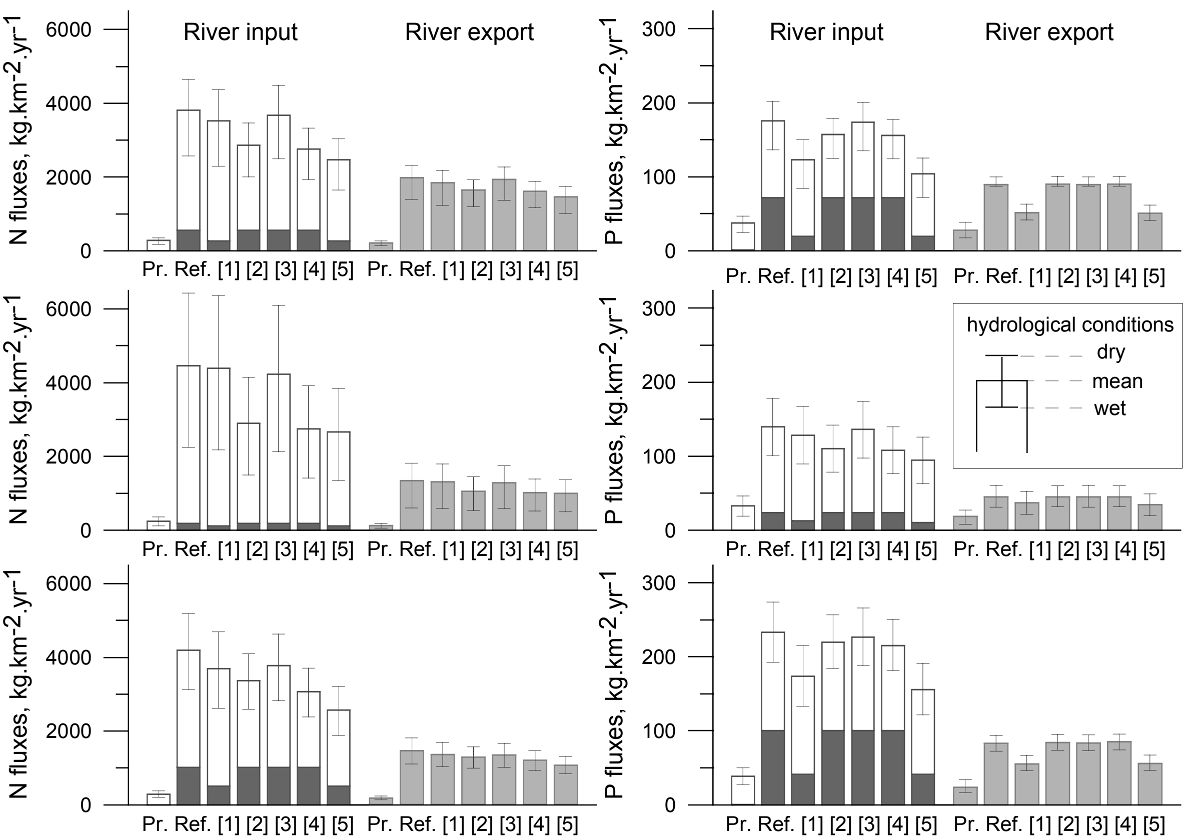

Figure 3-4: Modification of nitrogen and phosphorus fluxes in response to several mitigation measures, as simulated by the Seneque-Riverstrahler model for the Seine, Somme, and Scheldt Rivers. River input fluxes are characterized by the contributions of point sources (dark grey) and diffuse sources (white), 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); [4]: [2]+ [3] and [5]: [1] + [2]+ [3].

3.3.1.3. 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.

3.3.1.4. 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} yr^{-1} for the Seine, 639–1829 kg N km^{-2} yr^{-1} for the Somme, and 985–1587 kg N km^{-2} yr^{-1} for the Scheldt. Phosphorus fluxes were in the range of 87–100 kg P km^{-2} yr^{-1} for the Seine, 31–61 kg P km^{-2} yr^{-1} for the Somme, and 74–96 kg P km^{-2} yr^{-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.

3.3.2. Reduction of point sources by improved sanitation

3.3.2.1. 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 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 (Figure 3-5). 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.

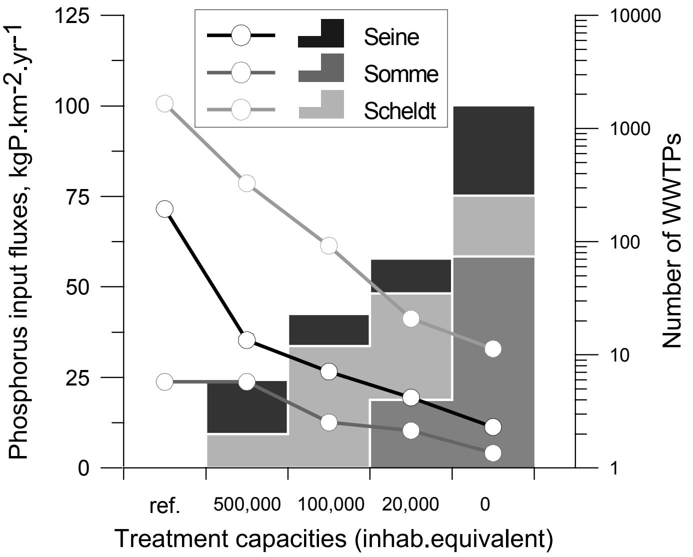

Figure 3-5: Decrease in phosphorus emission levels within the three basins in response to progressive improvements in P treatments by wastewater treatment plants (WWTPs) with capacities of 500,000; 100,000; 20,000 inhabitant-equivalents and less. The numbers of WWTPs are indicated on the second Y axis (log scale). Ref: present situation.

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. Figure 3-6 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} yr^{-} 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).

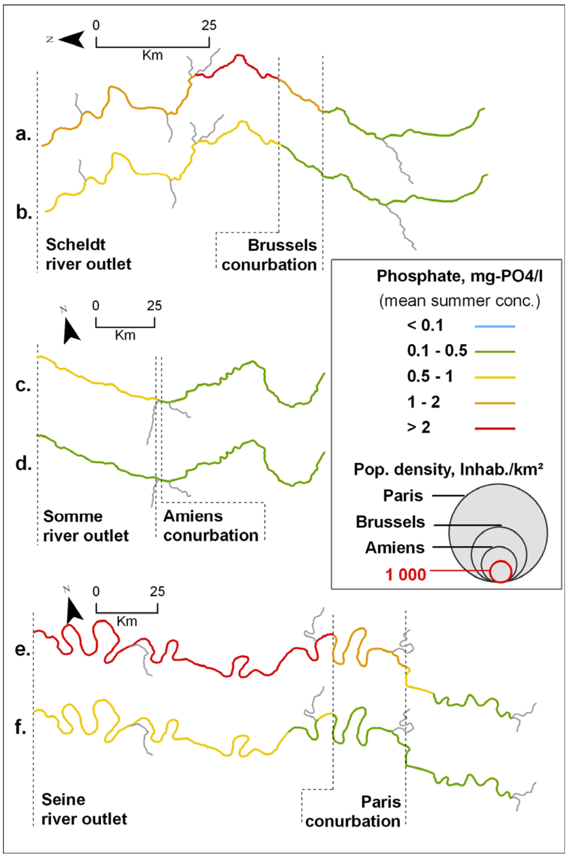

Figure 3-6: Distribution of mean summer concentrations of phosphate as calculated by the Riverstrahler model for sanitation state and hydrological conditions defined for the year 2000 for the downstream portions of the Seine (e), Somme (c) and Scheldt (a) rivers. Improved sanitation situations refer 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. Rivers sections directly impacted by urban releases are delineated with dotted lines. Acceptable quality range from 0.5 and 1 mg P04.l^{-1} according tp the French Water Agency classification, whereas 0.1 to 0.5 mg P04.l^{-1} is required for good quality status by both French and European regulation (circular DCE 2005/12).

3.3.2.2. 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 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.

3.3.3. Limiting N leaching through agro-environmental measures

3.3.3.1. Converting fodder corn crop area

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.

3.3.3.2. 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.

3.3.3.3. 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.

3.3.3.4. 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 (Figure 3-7). 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 (Figure 3-3a) whereas within the Scheldt basin surface water and groundwater N concentrations are more similar (Figure 3-3b), 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 (Figure 3-7).

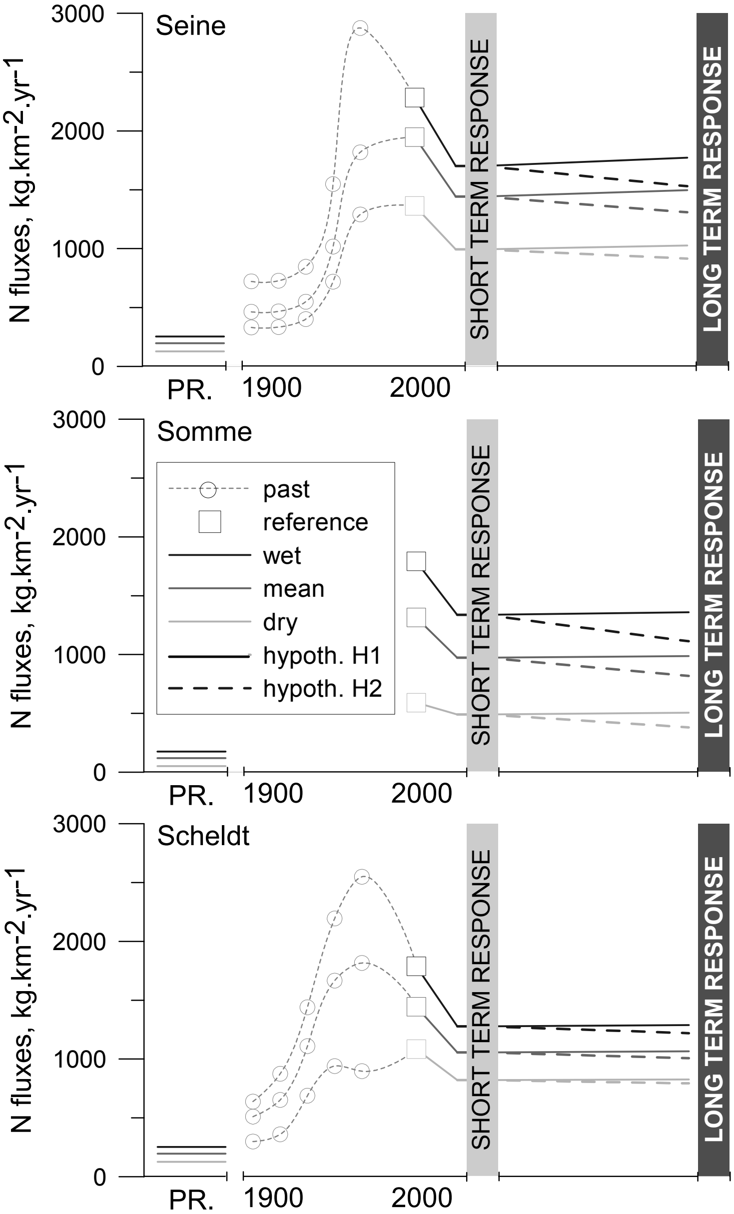

Figure 3-7: Impact of combined agricultural mitigation measures on nitrogen export fluxes for dry, mean, and wet years (light grey to dark grey). Historical (1950–1990) data were extract from Billen et al. (2007) for the Seine, and Billen et al. (2005) for the Scheldt. Theoretical long-term responses 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.

3.3.4. Imbalance of riverine fluxes

3.3.4.1. 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, 2006), account for only 1–8% of total silica input (Thieu et al., 2009). On the other hand, artificial regulation and damming have contributed to increased silica trapping. The modelled 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 3-3), 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 (Cugier et al., 2005; Skogen et al., 2004).

3.3.4.2. 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 3-3 for both the short and the long term (H1 and H2).

Table 3-3: 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}.yr^{-1})	127	253	52	176	123	235
P fluxes (kg.km^{-2}.yr^{-1})	17	38	8	27	16	33
Si fluxes (kg.km^{-2}.yr^{-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}.yr^{-1})	-112	-257	-88	-285	-177	-335
P surplus over Si (kg.km^{-2}.yr^{-1})	-16	-32	-11	-37	-25	-46
Present condition						
N fluxes (kg.km^{-2}.yr^{-1})	1378	2311	639	1829	985	1568
P fluxes (kg.km^{-2}.yr^{-1})	87	100	31	61	74	96
Si fluxes (kg.km^{-2}.yr^{-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}.yr^{-1})	1156	1833	448	1342	833	1331
P surplus over Si (kg.km^{-2}.yr^{-1})	57	36	11	-2	37	29
Short-term coupled scenario						
N fluxes (kg.km^{-2}.yr^{-1})	994	1702	492	1338	822	1278
P fluxes (kg.km^{-2}.yr^{-1})	41	61	19	49	46	66
Si fluxes (kg.km^{-2}.yr^{-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}.yr^{-1})	785	1253	350	893	565	815
P surplus over Si (kg.km^{-2}.yr^{-1})	12	-1	-1	-13	10	2
Long-term coupled scenario [H1]	1041	1798	514	1379	839	1306
N fluxes (kg.km^{-2}.yr^{-1})	41	62	19	49	46	67
P fluxes (kg.km^{-2}.yr^{-1})	529	1139	360	1127	649	1174
Si fluxes (kg.km^{-2}.yr^{-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}.yr^{-1})	830	1342	370	928	579	837
P surplus over Si (kg.km^{-2}.yr^{-1})	12	-1	-1	-13	11	2
Long-term coupled scenario [H2]						
N fluxes (kg.km^{-2}.yr^{-1})	928	1552	387	1128	804	1237
P fluxes (kg.km^{-2}.yr^{-1})	41	63	19	51	46	67
Si fluxes (kg.km^{-2}.yr^{-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}.yr^{-1})	717	1095	243	675	545	767
P surplus over Si (kg.km^{-2}.yr^{-1})	12	-1	-1	-13	11	2

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 (Figure 3-6). 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.

With silica fluxes as a reference, the required amount of nitrogen delivered should not exceed 200–450 kg N km^{-2} yr^{-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} yr^{-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.

3.4. 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.

Acknowledgements

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CHAPITRE

4 - Evaluation théorique d'une généralisation de l'agriculture biologique

Nitrogen cycling in a hypothetical scenario of generalised organic agriculture in the Seine, Somme and Scheldt watersheds

Thieu, V.[1], Billen, G.[1,2], Garnier, J.[1,2] and Benoit, M.[3]
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1 UPMC Univ Paris 06, UMR 7619 Sisyphe, F-75005 Paris, France
2 CNRS, UMR 7619 Sisyphe, F-75005 Paris, France
3 UR 055 INRA, 662 av L. Buffet, 88500 Mirecourt, France

Abstract

Nitrogen contamination of ground- and surface water in the Seine, Somme and Scheldt watersheds, as well as in the receiving coastal marine zones, results in severe ecological problems. Previous modelling results showed that the implementation of classical management measures involving improvement of wastewater purification and current good agricultural practices are not sufficient to obviate these problems. A more radical scenario was therefore established, consisting of a generalised shift to organic agriculture of all agricultural areas in the three basins. This scenario involves a significant reduction of agricultural production in the Seine and Somme watersheds along with an increased livestock density, while livestock density would be decreased in the Scheldt basin, bringing the three basins closer to autotrophy/heterotrophy equilibrium, while substantially decreasing their dependence on importations of animal products. Nitrate concentrations in most of the drainage network would drop below the threshold of 2.25 mgN/l in the most optimistic hypothesis. The excess of nitrogen over silica (with respect to the requirements of marine diatoms) delivered into the coastal zones would be decreased by a factor from 2 to 5, thus strongly reducing, but not entirely eliminating the potential for marine eutrophication.

4.1. Introduction

For the 50 last years, the rapid development of modern agriculture in industrialised countries has considerably affected the quality of water resources, to the point of jeopardising the capacity of rural territories to produce drinking water. With respect to the early traditional agricultural systems of north-western Europe, modern agro-systems are characterised by strong decoupling of crop and animal farming. While the essence of traditional agriculture was the synergy between animal husbandry and cereal cultivation (the former providing manure to the latter) without chemical inputs, the trend is nowadays, since the last three decades, to spatially separate these two activities, some regions restricting themselves to crop production, entirely sustained by synthetic fertilisation, while other regions specialise in animal husbandry, largely based on the importation of fodder from distant areas. In the former regions, leaching of mineral fertilisers and herbicides has induced severe alteration of ground- and surface water, while in the latter, manure produced in excess over local grass- and cropland plant growth requirements has led to considerable nitrogen and phosphorus soil surpluses and water contamination. Once transported to marine areas, nutrients in excess sustain the development of algal blooms, causing anoxia, fish kill or other harmful manifestations (Camargo and Alonso, 2006; Diaz and Rosenberg, 2008).

The three adjacent watersheds of the Seine, Somme and Scheldt rivers offer textbook examples of the situations described above. The Somme watershed, with a low population density (101 inhab/km²), is an intensive agricultural basin mostly dedicated to cereal and industrial crops. The Seine (202 hab/km²) is in the same situation as far the central Parisian basin regions are concerned, characterised by an increase of arable land at the expense of grassland (Mignolet et al., 2007), while mixed farming areas still exist at the eastern and western fringes of the watershed. The Scheldt basin, with a much higher population density (497 hab/km²), is characterised by very intensive animal farming activities. Changes in nutrient export in response to human activities within these watersheds over the last 50 years are characterised by opposite trajectories for nitrogen and phosphorous. While phosphorous has recently begun a rapid decline (Billen et al., 2001; 2007b; 2005), the advent of industrialised agriculture has increased the amount of nitrogen transported to coastal areas (Billen et al. 2001) or has stabilised at a higher level (Billen et al., 2005), threatening the supply of good-quality drinking water (Ledoux et al., 2007). Moreover, both the total amount and the proportions of the nutrients (N, P, Si) discharged at the outlet of the three basins into the coastal zones of the English Channel and the southern North Sea are the cause of severe eutrophication of these areas, which results in massive blooms of undesirable algal species (Lancelot et al., 2007; 2009; 2005).

Recent modelling on these three watersheds have shown that phosphate contamination could from now be easily restricted by the on-going improvement of waste water P-treatment (Thieu et al., 2010). Also, the establishment of the N budget within the three basins (Billen et al., 2009b; Thieu et al., 2009) emphasised the minor contribution of N point source emission, making diffuse agricultural sources the main target to mitigate nitrogen emission. However, implementation of current good agricultural practices included in the Common Agricultural Policy, such as introduction of catch crops (like perennial ryegrass), reduction of fertilisation, extensification of cattle farming, etc., would not be sufficient to meet the requirements of European regulations for both surface and groundwater contamination (Directive 2000/60/EC, 2000; Thieu et al., 2010) and the balance of winter N:P ratios (Lancelot et al., subm.; OSPAR, 2005). The critical delay of groundwater response to changes in agricultural practices complicates the assessment of short-term mitigation measures. The low sensitivity of nitrate contamination in the Seine aquifers to agro-environmental measures was mentioned by Ledoux et al. (2007), who showed that stopping new N inputs could significantly reduce nitrate concentrations, while implementing good agricultural practices only stabilised them on the horizon of 2050–2100.

This paper explores a more radical scenario consisting in a complete conversion of the agriculture of the three basins to organic farming (OA), defined, according to Stanhill (1990), as agricultural production practices "which seek to minimize the flow of inputs and outputs which sequester nonrenewable resources across the boundaries of the production area", thus excluding the use of synthetically compounded fertilisers and the importation of livestock.

We are aware that such a scenario has a low level of short-term socio-economic realism. Although there is an increasing demand for organic food products by the Western European population (Bonny, 2006), massive conversion to organic farming does not seem not to be presently on the French or Belgian political agenda. Our scenario is therefore conceived more as a hypothetical and idealised one than as the assessment of a realistic management program or as an evaluation of policy changes. Our goal here is simply to evaluate the potential of a drastic change in agriculture, starting from its ability to sustain food production for local populations, but also to meet with regulation expectancies in terms of freshwater quality and coastal ecosystem health.

4.2. Characterising the current N-status of the three watersheds

The present biogeochemical functioning of the Seine, Somme and Scheldt is characterised by a highly perturbed nitrogen cycle. Three aspects of these perturbations are examined hereafter, in terms of the balance between production and consumption, the deterioration of freshwater quality within the drainage network, and the impact on the coastal marine eutrophication.

4.2.1. Auto-heterotrophy status of the watershed

Billen et al. (2007b; 2009b) proposed to define the "autotrophy" of a given territory as its production of food and feed (harvested and grazed products), expressed as N content, and "heterotrophy" as the N content of food and feed required to sustain the local population and livestock. Urban areas, where food is consumed but not produced, are obviously purely heterotrophic. Rural regions specialised in crop production are autotrophic and export nitrogen as food and/or feed, while those characterised by intensive animal farming sustained by imported feed are heterotrophic. The net commercial export – or import – of agricultural products (i.e. the surplus – or shortage – of agricultural production over the requirements of the local population and livestock) can be calculated as the balance between autotrophy and heterotrophy. The Seine and Somme basins are typical examples of autotrophic territories (Billen et al., 2009b) with a potential export of nitrogen of, respectively, 4294 and 7909 $kgN.km^{-2}.yr^{-1}$, while the Scheldt basin presents a high degree of heterotrophy (−4342 $kgN.km^{-2}.yr^{-1}$). The negative value for the Scheldt indicates that most food and feed consumed is actually imported and is thus directly related to the Scheldt basin's population density and its intensive livestock farming activity (Table 4-1, Figure 4-6).

4.2.2. Deterioration of freshwater quality

Concomitant with the introduction of nitrogen by intensive cropping and farming activities, nitrate leaching from agricultural areas considerably affects the quality of water resources. We used the Seneque/Riverstrahler model (Ruelland et al., 2007) to calculate nutrient transfers and transformations in the three drainage networks, using meteorological constraints as well as point and non-point sources of nutrient as inputs. The reference simulation is derived from Thieu et al. (2009) and corresponds to the state of human activity in the watershed set at the year 2000. Running the Seneque-Riverstrahler model with the above-mentioned inputs allows one to calculate the resulting nitrogen concentration within the drainage network of the three basins. Figure 4-4 shows that the status of contamination is medium quality (i.e. 2.25–5.65 mgN/l) in most areas, with more intensive agricultural zones presenting poor quality (5.65–11.3 mgN/l). For example, the Brie-Beauce agricultural district in the central Seine basin supports intensive cereal production that clearly damages water quality from the first streamorders, as it is also the case in the French part of the Scheldt basin.

4.2.3. Balance of riverine fluxes: Redfield ratios and N-ICEP indicator

To assess the potential of riverine nitrogen delivery to sustain coastal eutrophication, a relevant indicator is the excess of nitrogen over silica with respect to the Redfield (1963) molar ratios N:Si = 16:20. Indeed, excess nitrogen delivered may lead to new production of nonsiliceous, often harmful algal blooms. The indicator of coastal eutrophication potential (ICEP: Billen and Garnier, 2007) defines the production of nonsiliceous algae expressed in carbon biomass units ($kgC/km^2/day$) sustained by nutrient delivered in excess over silica at the coastal zone. Calculation of the ICEP indicator (on the basis of N only) for the different scenarios is presented in Table 4-3. This indicator appears to be sensitive to hydrological conditions, showing higher potential for coastal eutrophication during wet years; however, both wet and dry simulation appears critical. Moreover, the results of Thieu et al. (2010) assessing a scenario of good agricultural practices still show a nitrogen surplus over silica of 785 and 1253 $kgN.km^{-2}.yr^{-1}$ for the Seine, 350 and 893 $kgN.km^{-2}.yr^{-1}$ for the Somme and 565 and 815 $kgN.km^{-2}.yr^{-1}$ for the Scheldt for dry and wet conditions, respectively (Table 4-3). Even a long-term assessment (including the response of groundwater) of such moderate management measures may not be sufficient to restore the balance of N:P and N:Si ratios (Figure 4-5), strengthening the idea that additional drastic measures to control nitrogen sources from agriculture are required to control eutrophication at the coastal zone.

4.3. Construction of an idealised OA scenario

4.3.1. Agronomical constraints

The detailed implementation of organic farming techniques can vary greatly from one region to another, as the essence of this alternative agriculture is to adapt itself to local conditions by choosing the most suitable plant varieties, crop rotations and livestock species. Our idealised scenario considers that all areas of the three basins presently devoted to agriculture (both as cropland and grassland) are converted to organic farming practices without changing the total agricultural area. The scenario implies that the only sources of fertilisation for croplands are (i) soil N_2 fixation and green-manure provided by legume crops inserted within rotations with cereals crops and (ii) animal manure produced by livestock fed with locally produced grass or fodder crops (see Figure 4-1). It also imposes the constraint that cattle feeding is ensured by local production because the import of fodder is out of our hypothesis. Therefore, given a certain cattle density (in livestock units, LU), the proportion of agricultural area devoted to fodder production is calculated assuming that cattle consumption equals the sum of cattle excretion (82 kgN/LU/yr) and meat/milk production (17 kgN/LU/yr) and using a mean yield of 110 kgN/ha/yr for N-fixing fodder crops. These figures correspond to the mean values currently observed in the three basins, based on official agricultural statistics (Agreste, 2000; NIS, 2000). The area devoted to cereal cultivation is then calculated by assuming that a certain proportion of the agricultural area not devoted to fodder crops is used for N-fixing green manure production. Theoretically, the percentage of green manure with respect to total non-fodder area may vary from 0 to 50%, the latter value corresponding to a bi-annual rotation. The total N-fertilisation of cereal cropland is then calculated using the calculated amount of livestock excreta plus green manure. The resulting production of cereal is then estimated using the relationship between yield and total N-fertilisation, observed in the three basins in the present and past situations (Figure 4-2).

This procedure implicitly assumes that organic farming techniques produce yields similar to conventional agriculture, when similar levels of fertilisation are applied. A number of studies have compared the yield of organic and conventional agriculture for different crops within the same pedo-climatic context in industrial countries (Badgley et al., 2007; Stanhill, 1990): they concluded that the former is approximately a mean 10% lower than the latter. This can be explained by lower fertilisation as well as by a lower control of pests and adventices. Available detailed data for organic farms in the Centre region of France (Glachant, 2009) show a crop yield versus fertiliser relationship quite similar to that established for conventional agriculture in our three basins (Figure 4-2), thus supporting our assumption.

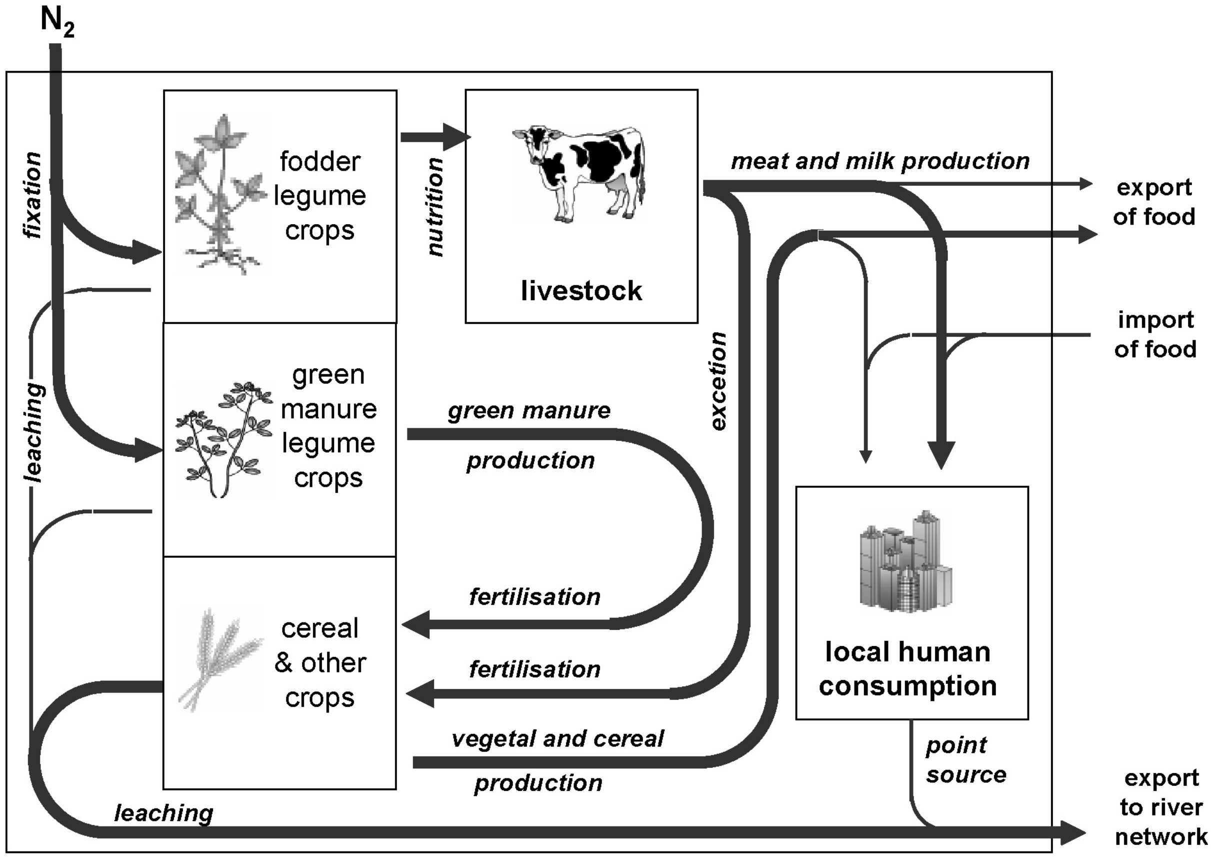

Figure 4-1: Circulation of nitrogen in an idealised organic agriculture scenario.

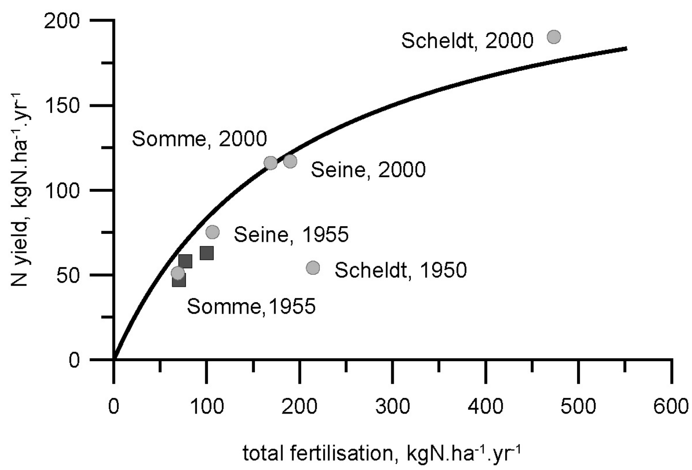

Figure 4-2: Relationship observed between mean cereal crop yield and total (synthetic and organic) N fertilisation observed in the three basins in 2000 and 1955 (grey circles, as calculated from French and Belgian agricultural statistics). The mean yields reported for organic farms in the Centre region of France (Glachant 2009) are also plotted for comparison (dark grey squares).

Figure 4-3a and 4-3b shows the theoretical cereal yield and total cereal production calculated for the three basins as a function of livestock density and percentage of green manure used in crop successions, taking into account the constrained land allocation for fodder, green manure and cereal crops. With increasing livestock density, cereal yield increases rapidly, both because the absolute amount of fertilisation increases but also because this amount is concentrated on a decreasing cropland area, which also explains why the total cereal production decreases after reaching a maximum. By comparison, the cereal yield actually observed in organic farming systems in France and Belgium ranges between 45 and 100 kgN/ha/yr (FranceAgriMer, 2009; Glachant, 2009), thus probably restricting the realistic part of the graphs in Figure 4-3a to the lower half of the cattle density range.

4.3.2. N-leaching under OA

For the sake of evaluating the diffuse nutrient emissions of organic agriculture, we have examined the data reported in the literature concerning nitrogen concentrations in surface or sub-root water from organic farms.

Drinkwater et al. (1998) showed that substituting synthetic fertiliser with either animal or green manure to maize-soybean rotations in the north-eastern US reduced nitrate leaching by 30% and increased the soil organic matter pool. In the Bavarian Tertiary Hills region, with drained loamy soils, Honish et al. (2002) reported mean sub-root water nitrate concentration of 4.5–6.5 mgN/l for an organic farming domain, while concentrations twice as high were found in conventional agriculture. Korsaeth and Eltun (2000) compared conventional and ecological farming practices in central south-eastern Norway, including a complex rotation alternating root crops, cereal and ley. Nitrate drainage runoff was reduced from 30–35 to 20–21 kgN/ha/yr, with mean nitrate concentrations in the range 4–6 mgN/l. Hansen et al. (2000) used a modelling approach to evaluate nitrogen leaching from different organic farming systems (including arable crop, pig and dairy-beef production) in Denmark on both loamy and sandy soils. The calculated mean nitrate concentration under the root zone varied in the range of 7.7–14 mgN/l for sandy soils and 5.5–8.6 mgN/l for loamy soils. For the entire experimental INRA domain of Mirecourt (East of Parisian Basin, France), which has been reconverted into organic farming since 1994, Benoît and Larramendy (2003) calculated the mean nitrate concentration of drained water, based on measurements in suction-pipes, to 5.5–6.5 mgN/l, about half being caused by the ploughing of temporary alfalfa meadows used in the crop rotation before cereals. All these data converge toward a range of 3–6 mgN/l as a reasonable value for the mean nitrate concentration in the surface water produced by organic agriculture. This range will be used in our scenario to define the diffuse sources of nitrate. Interestingly, this is exactly the range (3.4–5.8) reported by Sabatier (1890) for the agricultural systems of the Paris basin at the end of the nineteenth century.

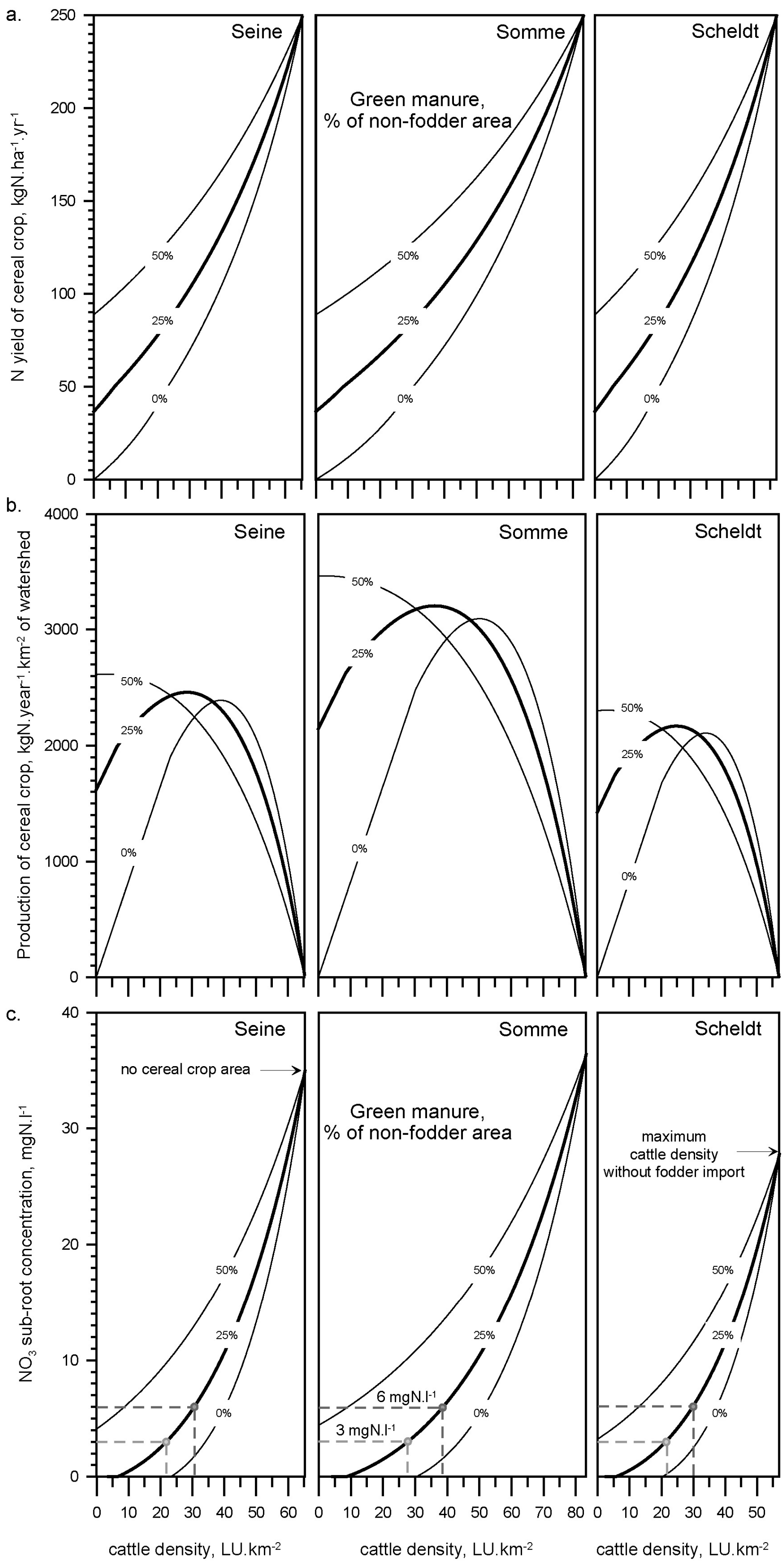

Figure 4-3: N yield of cereal crop (a), Production of cereal crop (b) and Nitrate sub-root concentration as a function of organic fertilsers, here represented by cattle density (manure) and the proportion of non-fodder area devoted to legume crops (green manure) with no change in the total agricultural area of the Seine, Somme and Scheldt basins.

To make this range consistent with the agronomical constraints discussed in the preceding section, we calculated the nitrogen losses from arable land as the excess of N-fertilisation over N crop uptake and converted it into nitrate concentration considering the mean annual runoff values of the respective watersheds, namely 260 mm for the Seine, 240 mm for the Somme and 330 mm for the Scheldt (Thieu et al., 2009). For pastures and green manure areas, a mean nitrogen loss of 5 kgN/ha was considered. The resulting mean nitrate sub-root concentration at the scale of these mixed-agricultural areas calculated for the three basins as a function of livestock density and green manure proportion are shown in Figure 4-3c. Setting the proportion of green manure in succession to a value of 25%, the above-discussed range of 3–6 mgN/l for sub-root nitrogen concentrations defines the allowed range of cattle densities to 22–30, 27–39 and 21–29 LU/km² for the Seine, Somme and Scheldt, respectively.

4.3.3. Other constraints to the OA scenario

As for groundwater concentration, we will assume in this idealised scenario that aquifers have reached equilibrium with the infiltrating sub-root water concentration. As such, equilibrium can require as long as 50 years, our scenario thus represents the long-term effect of organic agricultural practices. The same riparian nitrate retention (expressed as a percentage of nitrate flux emitted from the watershed) was assumed to be the same as in the reference scenario (Thieu et al., 2009). The scenario considered here uses the same hydrological constraints as in the latter study, namely those of a wet, a mean and a dry year (2001, 2000 and 1996, respectively). The point sources of nutrients (inputs of urban wastewater) correspond to the scenarios defined in Thieu et al. (2010) with 90% P removal and 70% N removal in all wastewater treatment plants with capacity over 20,000 inhabitant equivalents.

Data on the role of organic farming practices on the emissions of other nutrients (phosphorous, silica) and suspended matter are too scarce to support any hypothesis about a change with respect to the presently observed situation, so that we have assumed the same diffuse sources for these nutrients as those discussed by Thieu et al. (2009).

4.4. Results

The results of this generalised organic agriculture scenario will be compared with those presented earlier by Thieu et al. (2010). They include i) a pristine scenario, (ii) a present (2000) reference scenario, (iii) a scenario with improved wastewater treatment and implementation of good agricultural practices, and finally (iv) a business-as-usual scenarioresulting from the status quo in wastewater treatment and the pursuit of the present agricultural practices with groundwater nitrate concentrations continuing to increase until they reach equilibrium with infiltrating sub-root water (Ledoux et al., 2007). The assessment of these scenarios is based on the three indicators previously identified to characterise the fate of nitrogen at the basin scale and up to the coastal zone.

4.4.1. Reducing the gap between autotrophy and heterotrophy

Our organic farming scenario involves a significant reduction of agricultural production in both the Seine and the Somme basins, with a simultaneous increase of livestock density and heterotrophy (Table 4-1). The Seine watershed becomes nearly equilibrated in terms of autotrophy (A) and heterotrophy (H), with only a small export of agricultural products, ranging from 1310 to 1350 kgN/km²/yr, respectively, for the 3- and 6-mgN/l sub-root concentration scenarios. The Somme watershed can still export a larger part of its agricultural production (from 2390 to 2650 kgN/km²/yr). In the Scheldt basin, agricultural production remains about the same, but the livestock density is greatly decreased from 70 LU/km² in the current conventional agriculture to less than 30 LU/km² in our organic farming scenarios. However, due to its high human population density, the Scheldt remains a heterotrophic basin, importing about 570 kgN/km²/yr as food and feed. The trends corresponding to our scenario are represented in an A-H diagram (Figure 4-4), which also shows the long-term historical trajectories of the three basins. This organic agriculture scenario corresponds to the reversal of the trends observed during the last 30–50 years, and the return toward a closer balance between autotrophy and heterotrophy.

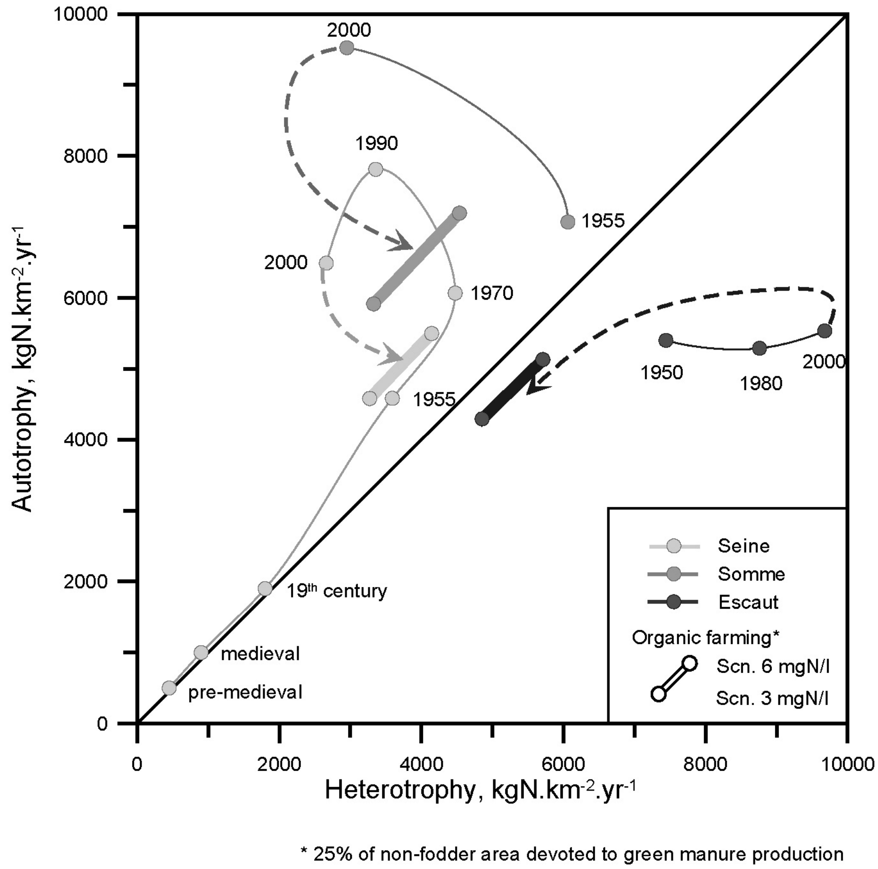

Figure 4-4: Autotrophy and heterotrophy of the three basins based on the assumption of a shift to organic agriculture (see Table 4-1) compared with the present and historical situations (Billen et al., 2009).

Table 4-1: Estimation of autotrophy and heterotrophy of the three basins within the scenario of shifting to organic agriculture of the entire present agricultural area. The corresponding figures for the present situation (as described in Billen et al., 2009) are indicated for comparison.

	Seine	**Somme**	**Scheldt**
Total area, km²	76,370	6190	19,860
Agricultural area, km²	50,487	5237	13,039
Human population density, inhab./km²	202	101	496
Present situation (Billen et al., 2009)			
Autotrophy, kgN/km²/yr	6855	10500	5355
Livestock, LU/km²	15.6	23.3	69.2
Heterotrophy, kgN/km²/yr	2561	2591	9697
Export potential (A-H), kgN/km²/yr	4294	7909	−4342
Generalised OA scenario (6 mgN/l)			
Autotrophy, kgN/km²/yr	5498	7193	5131
Livestock, LU/km²	30.5	38.8	29.8
Heterotrophy, kgN/km²/yr	4144	4536	5710
Export potential (A-H), kgN/km²/yr	1354	2658	−579
Generalised OA scenario (3 mgN/l)			
Autotrophy, kgN/km²/yr	4583	5915	4296
Livestock, LU/km²	21.75	27	21.25
Heterotrophy, kgN/km²/yr	3272	3328	4854
Export potential (A-H), kgN/km²/yr	1311	2588	−559

Table 4-2: Human food production and consumption of the three basins in the organic agriculture scenario. The corresponding figures for the present situation (as deduced from agricultural statistics) are indicated for comparison.

	Seine		**Somme**		**Scheldt**	
	Needs	Prod. (% needs)	Needs	Prod. (% needs)	Needs	Prod. (% needs)
Present situation						
Cereal (and vegetables), kgN/km²/yr	387	6189 *(1676%)*	194	9524 *(4921%)*	950	5535 *(582%)*
Meat and milk, kgN/km²/yr	719	267 *(37%)*	359	*484 (135%)*	1765	1189 *(67%)*
Generalised OA (6 mgN/l)						
Cereal (and vegetables), kgN/km²/yr	387	2460 *(636%)*	194	3211 *(1659%)*	950	2137 *(225%)*
Meat and milk, kgN/km²/yr	719	522 *(73%)*	359	*804 (224%)*	1765	512 *(29%)*
Generalised OA (3 mgN/l)						
Cereal (and vegetables), kgN/km²/yr	387	2417 *(624%)*	194	3141 *(1623%)*	950	2157 *(227%)*
Meat and milk, kgN/km²/yr	719	372 *(52%)*	359	*560 (156%)*	1765	365 *(21%)*

In addition to the overall status of auto-/heterotrophy, the satisfaction of human needs in terms of both animal and plant proteins should be assessed. The human daily N requirement per capita in France and Belgium is estimated at 15 gN/inhab/day, with a share of animal products (meat, milk and eggs) of about 65% (FAOstat, 2006). The share of animal and cereal production implied by the construction of our organic agriculture scenario does not match this pattern. Regarding the cereal production by organic farming, although it would decrease in average by a factor of 3 with respect to their level in the current situation, this production remains in large excess compared with the requirements of the local population (Table 4-2). However, as far as animal products are concerned, the present conventional agriculture system, except within the Somme basin, does not cover the demand of local populations for meat and milk, which is presently largely imported from distant specialised areas. Our organic agriculture scenario slightly improves the self-sufficiency for animal products in the Seine basin (52–73% for the respective organic scenarios, compared with the current 37% value), while for the Scheldt basin the degree of meat and milk needs coverage decreases (21-29% for the organic scenarios, compared with a present level of 67%)

4.4.2. Improving the quality of the drainage network

Thieu et al. (2010) have demonstrated that improvement of urban waste water treatment in the Seine, Somme and Scheldt basins has a much greater effect on the restoration of phosphorus quality than on nitrogen because of the dominant part of diffuse sources in the total N loading. The short-term outcomes of the implementation of good agricultural practices would be a decrease of nitrogen surface runoff fluxes with respect to the reference 2000 scenario by 32%, 40% and 35%, respectively, for the Seine, Somme and Scheldt (Thieu et al., 2010). In our organic agriculture scenario, this reduction over the short term reaches 54–73%, 63–78% and 49–70% for the three basins, respectively, when testing the range of 3–6 mgN/l for the mean nitrate concentration in the sub-root water produced by organic agriculture. Under the same assumptions, the groundwater concentrations would decrease of 24–58%, 39–67% and 13–51% for the three basins, respectively, in the OA scenario. By comparison, in the business-as-usual scenario, the groundwater nitrogen fluxes to the drainage network increases (by 82% in the Seine, 78% in the Somme and 77% in the Scheldt) due to the equilibration of groundwater concentrations.

Figure 4-5 shows that the impact of good agricultural practices is essentially significant in the more damaged region (Brie-Beauce for the Seine, Upper Scheldt) restoring medium water quality, except in small residual poor-quality areas. However, the (long-term) assumption of a 6-mg/l mean nitrate concentration with organic agriculture restores all poor-quality areas and results in a shift from medium to good water quality (0.45–2.25 mgN/l) for the Somme and the upstream basins of the Seine, which mixing forest and agricultural use. By considering a lower sub-root nitrate concentration (3 mg/l), these areas become very good (<0.45 mg N-NO_3/l), and most of the three drainage networks become good quality. Such an organic agriculture scenario is the only one able to achieve the good quality status in the downstream part of the three rivers. The pristine state scenario produces a very good quality to the entire drainage networks, although this does not represent a realistic target, as it ignores any human life. At the other extreme of this prospective exercise, the business-as-usual scenario shows severe worsening of freshwater contamination, characterised by an overall poor quality in the lower Seine river and several of its tributaries. All these results support the urgent need to mitigate nitrogen emissions.

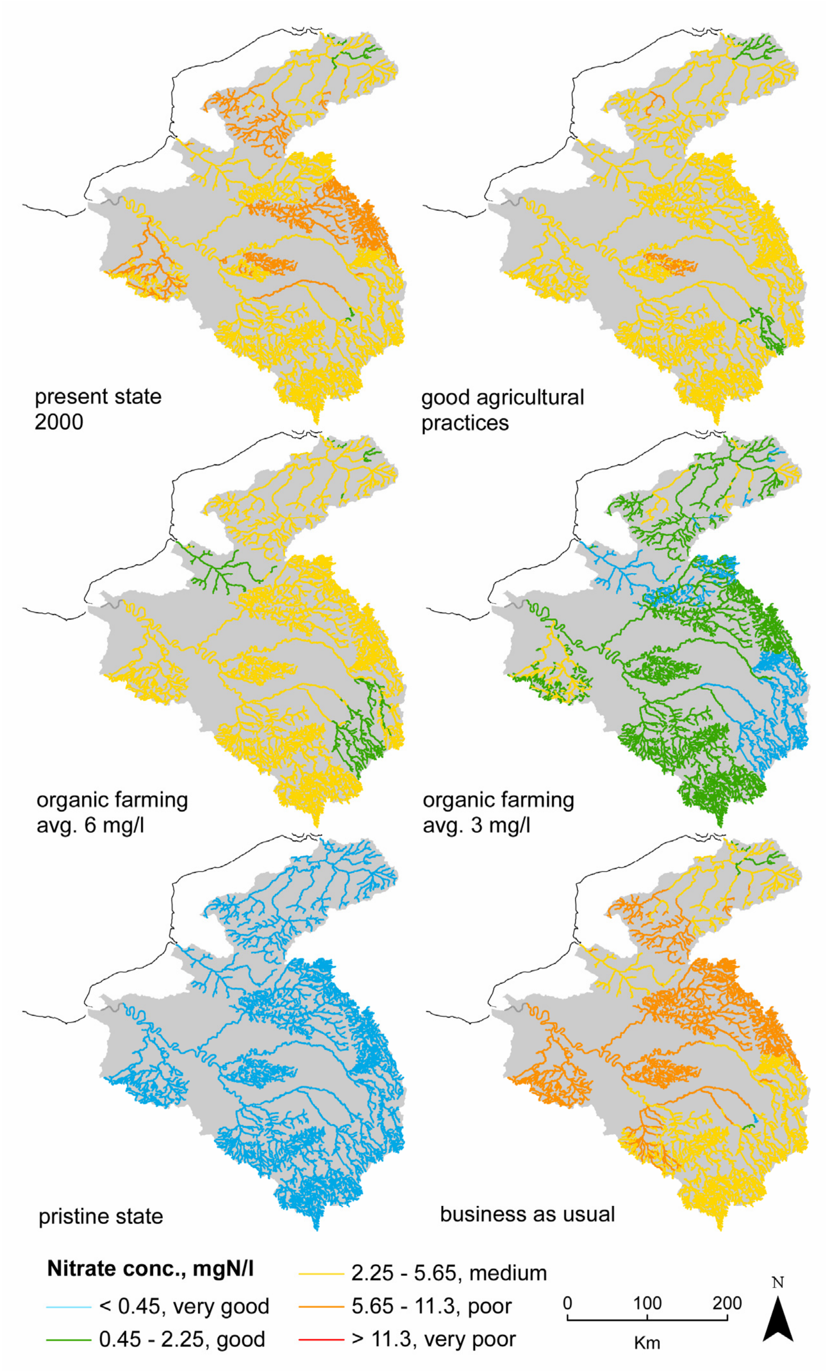

Figure 4-5: Distribution of yearly mean concentration of nitrogen over the three drainage networks for different scenarios, as calculated by the Riverstrahler model.

4.4.3. Limiting the potential for coastal eutrophication

For mean hydrological conditions (based on the year 2000), present reference nitrogen fluxes exported from the Seine to the sea are estimated at 1975 kgN/km²/yr (Figure 4-6). This amount of nitrogen delivered decreases to 1462, 1084 and 768 kgN/km²/yr, respectively, for the short-term good agricultural practices scenario and the two long-term organic agriculture scenarios, while the business-as-usual simulation provides a flux of 2245 kgN/km²/yr. For the Somme, characterised by a reference flux of 1333 kgN/km²/yr, these changes in agricultural practices reduce nitrogen export values to 987, 663 and 436 kgN/km²/yr, respectively, while 1623 kgN/km²/yr is reached for the business-as-usual scenario. In the Scheldt, the smaller gap between surface and groundwater contaminations makes the business-as-usual simulation (1570 kgN/km²/yr) closer to the current situation (1464 kgN/km²/yr). The impact of good agricultural practices is comparable to the effect of organic agriculture, with fluxes estimated respectively at 1071 and 974 kgN/km²/yr. A further decrease, to 737 kgN/km²/yr, is simulated assuming a mean nitrate concentration of 3 mgN/l on organic agricultural soils. Thus, whatever the hydrological conditions are, organic agriculture appears as the most efficient scenario. It significantly reduces nitrogen fluxes by a factor of 2–3 at the outlet of the three basins. In comparison, the implementation of good agricultural practices reduces it by a factor of only 1.4.

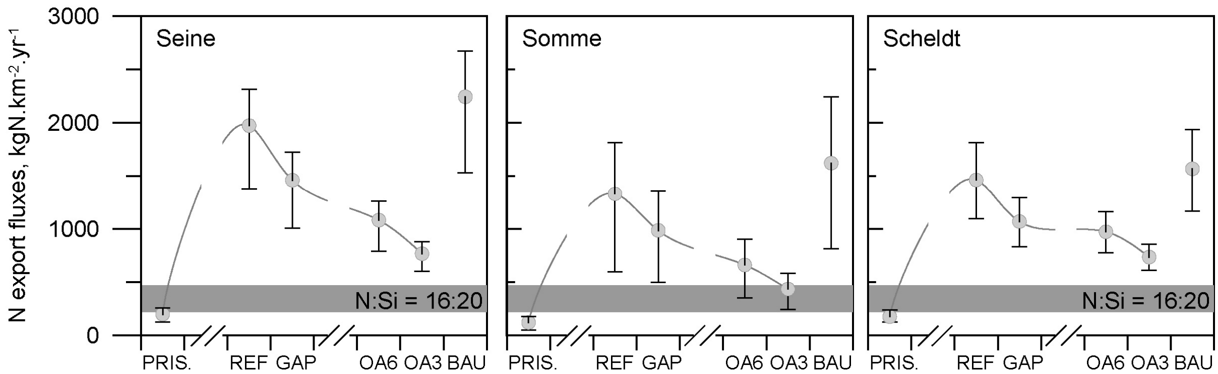

Figure 4-6**:** Mitigation of the Seine, Somme and Scheldt export fluxes. Calculation made by the Seneque-Riverstrahler model for mean (central symbol), dry (negative bar plot) and wet (positive bar plot) hydrological conditions. PRIS: pristine scenario, REF: reference year 2000 scenario, GAP: good agricultural practices scenario, BAU: business-as-usual scenario.

Negative values of the N-ICEP (as observed for Pristine condition, see Table 4-3) indicate that silica is present in excess over nitrogen, i.e. that there is "potentially" no eutrophication problem. On the contrary, positive values indicate that the amount of nitrogen is in excess of the requirement of siliceous algae growth, possibility leading to the development of nonsiliceous (undesirable) algae.

Table 4-3: Calculated values of the indicator of coastal eutrophication potential (N-ICEP) for the different scenarios, under dry and wet conditions.

ICEP, kgC/km²/d	**Seine**		**Somme**		**Schledt**	
Hydrology	dry	wet	dry	wet	dry	wet
Pristine	−1.8	−4.1	−1.4	−4.5	−2.8	−5.3
Reference 2000	18.2	28.9	7.1	21.2	13.1	21.0
Good agricultural practices	12.4	19.8	5.5	14.1	8.9	12.9
Organic farming 6mg/l	8.6	12.2	3.0	6.8	7.8	10.5
Organic farming 3mg/l	5.7	6.2	1.4	1.9	5.2	5.8
Business as usual	20.2	34.1	10.2	27.7	13.9	22.6

The variability of nitrogen-diffuse sources depending on the hydrological conditions explains greater N-ICEP values during wet years for all scenarios. The receiving coastal waters of the three basins are currently characterised by a high potential for eutrophication, with the N-ICEP value ranging for 'present reference' from 21 to 29 kgC/km²/day under wet hydrological conditions, while the business-as-usual scenario increases these values to 23–34 kgC/km²/day. Good agricultural practices would decrease the N-ICEP by wet conditions to 13–20 kgC/km²/day, while the two organic agriculture scenarios would reach N-ICEP values as low as 6-12 , 2-6 and 6-10 kgC/km²/day for the Seine, Somme, Scheldt respectively, thus representing a quite significant reduction of the potential to sustain harmful algae development in the coastal area.

4.5. Discussion

The scenario discussed in this paper was developed after having concluded that realistic mitigation measures (improvement of wastewater treatment and so-called good agricultural practices) are not radical enough to obviate the environmental problems caused by nitrogen contamination of ground-, surface and coastal waters. By constructing a scenario based on generalised organic farming practices at the scale of these three highly populated European watersheds, we were willing to test whether agriculture, by essence, can conciliate (i) the demand for food and feed by local populations, (ii) a good ecological functioning of aquatic ecosystems and (iii) a balanced nutrient status for the adjacent coastal area.

For the first point, it clearly appears that organic farming scenarios hold a closer equilibrium between autotrophy and heterotrophy, even though it preserves the original status of the three basins, with the Somme remaining the most autotrophic territory, while the Scheldt stands with a shortfall in production with respect to its consumption needs. Our results are valuable in the context of the recent controversy about the capability of organic agriculture to "feed the world" (Badgley and Perfecto, 2007; Connor, 2008). As a whole, the territory covered by the three basins – Seine, Somme and Scheldt – covering one of the most populated areas of the world, can largely produce the proteins required for human and animal consumption through organic farming, leaving even an exportable surplus of over 1000 $kgN/km^2/yr$, i.e. more than one-third the amount currently exported by the three basins combined. When looking into the details, however, one can see that the present requirements in animal proteins for human consumption would not be covered by our organic farming scenario, which would still require the importation of 40–58% of the current needs of meat and milk, compared with 50% in the current situation. This persisting requirement of long-distance importation of meat and milk clearly emphasises the unsustainability of the increasing share of animal proteins in the modern human diet, presently representing as much as 65% of total protein consumption in France and Belgium. Results of our organic scenarios are highly sensitive to any hypothetic change in human diet. We calculate that the two OA scenarios considered (respectively 3 and 6 mg/l) might became self-sufficient for both meat and cereal production if the percentage of animal protein consumed by human decrease to 35-50 % for the seine and 15-20% the Scheldt. Such ratios are currently observed across the world (FAOstat, 2006) : India (17%); Egypt (20%); Vietnam (27%); Senegal (31%); South Africa (35%); Bolivia (41%) Brazil and Russian Federation (50%)

Regarding the ecological functioning of the three river systems, our results clearly show that a shift to organic agriculture allows reaching much lower nitrate concentrations in surface water than any other mitigation measures. It is the only scenario able to bring the nitrogen concentration in most rivers from the three basins below the level of 2.25 mgN/l (10 $mgNO_3/l$), which is often considered a threshold for a good ecological status (Camargo and Alonso, 2006).

At the coastal zones, this scenario also more than halves the nitrogen delivery with respect to the good agriculture practice scenario. Judging from the N-ICEP calculation, this is not enough, unfortunately, to eliminate all risk of harmful algal blooms in the receiving coastal waters; the risk is, however, very significantly reduced. Application of the MIRO model (Lancelot et al., subm.) predicts that the maximum cell density of *Phaeocystis* blooms in the Belgian coastal zones would be reduced from 46 10^6 cells/l in the present situation to 21–30 10^6 cells/l in our organic agriculture scenario, while the duration of the blooms would be reduced from 33 days to 18 and 25 days, respectively, for the two OA scenarios and for similar hydrological conditions (those of the year 2000). By comparison, the corresponding figures for the scenario of good agricultural practices are 35 10^6 cells/l for 27 days.

As a whole, our generalised organic farming scenario is certainly not a realistic short-term objective. Presently, organic farming only concerns 3.9% of the total agricultural area in Europe, with, however, large disparities between countries, from 11% in Austria to only 2% in France (Bonny, 2006). However, on the consumer side, the demand for food produced from organic farming is rapidly growing, by 10–15% per year in recent periods (Bonny, 2006). This also might plead in favor of extending this mode of agriculture in the future, which over the long term appears to be the only way to preserve water resources while meeting the population's food requirements.

Acknowledgements

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CHAPITRE

5 - Application des scénarios tendanciels du Millenium Ecosystem Assessment

Sub-Regional and Downscaled-Global Scenarios of Nutrient Transfer in River Basins: the Seine-Scheldt-Somme Case Study

Thieu, V.[1], Mayorga, E.[2], Billen, G.[1,3,], and Garnier, J.[1,3] (2009)
Global Biogeochemical Cycle, in press, doi:10.1029/2009GB003583

1: UPMC Univ Paris 06, UMR 7619 Sisyphe, F-75005, Paris, France
2: Applied Physics Laboratory, University of Washington, Seattle, WA 98105-6698, USA
3: CNRS, UMR 7619 Sisyphe, F-75005, Paris, France

Abstract

In an attempt to downscale the global prospective scenarios established by the Millennium Ecosystem Assessment to the level of three individual watersheds (the Seine, Somme, and Scheldt Rivers), we examined the application of the regional RIVERSTRAHLER-model, based on a mechanistic representation of in-stream processes, in tandem with the semi-empirical Global*NEWS*-model, by downscaling the input data of the latter into information required by the former. Overall, the model simulates the major trends of the changes that occurred in 1970-2000, although with some discrepancies revealing the weakness of certain hypothesis in the global approach. For the future, the prediction is a significant decrease in total nitrogen and phosphorus fluxes into the sea compared to those of 2000. We showed the benefits of combining a process-based approach of nutrient transfer at the local scale with the use of global-scale models for integrating the development of socio-economic driving forces acting at the global level.

5.1. Introduction

In many places in the world, eutrophication has become a major problem affecting large estuaries (Cugier et al., 2005; Turner and Rabalais, 1994), coastal bays (e.g., Chesapeake (Jaworski et al., 1992), San Francisco bay (Cloern, 1996)), and open coastal areas (e.g., the continental coastal water of the North Sea (2007; Lancelot et al., 2005)). One of the primary controls of eutrophication is the nutrient levels, especially nitrogen (N), phosphorus (P), and silica (Si), as the amount and the ratio of these elements are key factors determining algal development (Billen and Garnier, 2007; Officer and Ryther, 1980). Together with a high concentration of human activity along the seashore, the nutrient load issued from human activity within the watershed and discharged by large river systems into coastal bodies of water, is also at the origin of environmental deterioration of marine ecosystems Consequently, the concerns of policymakers involved in integrated water management have evolved from local analyses of human activities and their proximate impact on river systems to a more consistent regional approach at the basin scale (Kronvang et al., 1999; Wolf et al., 2005).

A significant amount of research has been devoted to simulations of nutrient delivery to coastal zones. One of these approaches is the use of empirical models (Alexander et al., 2002; Boyer et al., 2002; Galloway et al., 2004; Green et al., 2004; Howarth et al., 1996; Seitzinger et al., 2005; Seitzinger et al., 2002a) that express nutrient fluvial transport as a function of several explanatory variables, including morphological and hydrological information, indicators of human activities in the watersheds, and including sometimes quantitative components of the nutrient land mass balance. A limitation of this type of model is the availability of sufficient data sets to calibrate the models parameters, although recent improvements in world database accessibility (e.g., Meybeck and Ragu, 1996) make this approach widely transferable to an increasing number of well-documented rivers systems or allows globally applicable calibration based on global river data. Alternatively, mechanistic/deterministic models (Billen et al., 1994; de Wit and Bendoricchio, 2001; Everbecq et al., 2001; Whitehead et al., 1998) evaluate the transfer and retention properties of river networks by describing the kinetics of the main processes involved in nutrient dynamics. The robustness of these models relies on their capacity to reproduce observed trends for the variable considered, without the need for a calibration stage.

Both the empirical and the mechanistic model have become more complex in terms of the need to explain (mechanistic approach) or describe (empirical approach) recent changes affecting the ecological functioning of the world's river systems. Indeed, continental aquatic systems originally driven by natural factors have been modified by human actions during the last several decades (Vitousek et al., 1997). Some of these anthropogenic forcing act across the physical boundaries of watersheds and must therefore be analyzed at a larger scale. This is, by definition, the case of societal drivers or even economic globalization but it also includes more global changes affecting any other compartment of the Earth (atmosphere, lithosphere) that interacts with river systems (Meybeck, 2003).

As an alternative to highly complex and uncertain predictions of how the future will evolve, a scenario approach integrates several projections by gathering multiple assumptions within consistent storylines (Verburg et al., 2006). Along the same lines initiated by the IPCC assessment (IPCC, 2000), the Millennium Ecosystem Assessment (MEA, 2005) proposed four scenarios, using storylines that explore aspects of plausible global futures and their implications for ecosystem services (defined as the benefits that people obtain from the environment (MEA, 2003)). These storylines represent a qualitative approach for describing the continental and worldwide dynamics controlling economic, demographic, and even sanitary development. However, the assessment of changes in human activities and their related impact on river basins and coastal ecosystems must be quantitative and include the use of global models. Accordingly, such models have been developed by the Global *NEWS* (Global Nutrient Export from Watersheds) working group (Seitzinger et al., 2009) by adapting previous models (Fekete et al., 2002; MNP, 2006; Seitzinger et al., 2005) with the aim of i) translating the MEA storylines into quantitative scenarios, ii) computing nutrient loading of the landscape and of river systems, and iii) using semi-empirical models to assess future nutrient river export to coastal ecosystems.

Due to the complexity of the global economic system, prospective scenarios of human activities in watersheds should be conceived at the global level and include regional to global scale interactions. However, there is also a need to increase the spatial resolution of the simulated scenarios, in order to examine them at a smaller scale, more adapted for management. This can be feasible with sub-regional, mechanistic models, better able to represent local dynamic processes, providing that they are able to make use of "large-scale-scenario" constraints as a background and to improve them by integrating sub-regional dynamics. Among the several models that describe the ecological functioning of river systems, this study examines the outcome of coupling of the Riverstrahler model (Billen et al., 1994) with global models. Beyond a simple comparison between "global-empirical" and "local-mechanistic" models, our aim was to explore their potential cooperation in order to achieve a refined prediction of the amount of nutrients delivered to the sea at the local to regional scale.

As an extension of the work by Sferratore et al. (2005), this work focuses on of the case study of the continental coastal waters of the North Sea which are severely affected by eutrophication, and examines the contribution of the Seine, Somme, and Scheldt Rivers nutrient fluxes. First, the heterogeneity of the three basins is discussed through a downscaled description of their landscapes and human activities. In the context of a joint modeling approach, the conceptual river system representation by the two models is compared and the sensitivity of the Riverstrahler model to be upscaled into a single basin is tested. The consistency and accuracy of global inputs is then analyzed with respect to a sub-regional-scale high-resolution database, and a methodology for downscaling the former is proposed. On the basis of the downscaled global input, a mechanistic assessment of nutrient exported to the sea is first validated for the recent period (1970–2000) and then extended to integrate the four scenarios of the MEA. Finally, this approach is used to test the potential impact on river retention of plausible future "inner-basin" changes.

5.2. Study area

The basins of the Seine, Somme, and Scheldt Rivers spreading across France, Belgium, and The Netherlands (Figure 5-1) are a major source of input for the continental coastal waters of the North Sea (Lacroix et al., 2007). The three individual watersheds, 6,000 km² for the Somme, 19,800 km² for the Scheldt, and up to 76,000 km² for the Seine, have quite contrasting characteristics. The Somme supports the lowest population density (101 inhabitants km^{-2}) and 81% of its area is devoted to farming activities. The Seine, by contrast, flows through large urban areas such that the population density is two-fold higher than that along the Somme (200 inhabitants km^{-2} on average) whereas agricultural land covers only 63% of the catchment area. With an average population density of 496 inhabitants km^{-2}, the Scheldt is the most populated system, with less than 50% of its area used for agricultural activities.

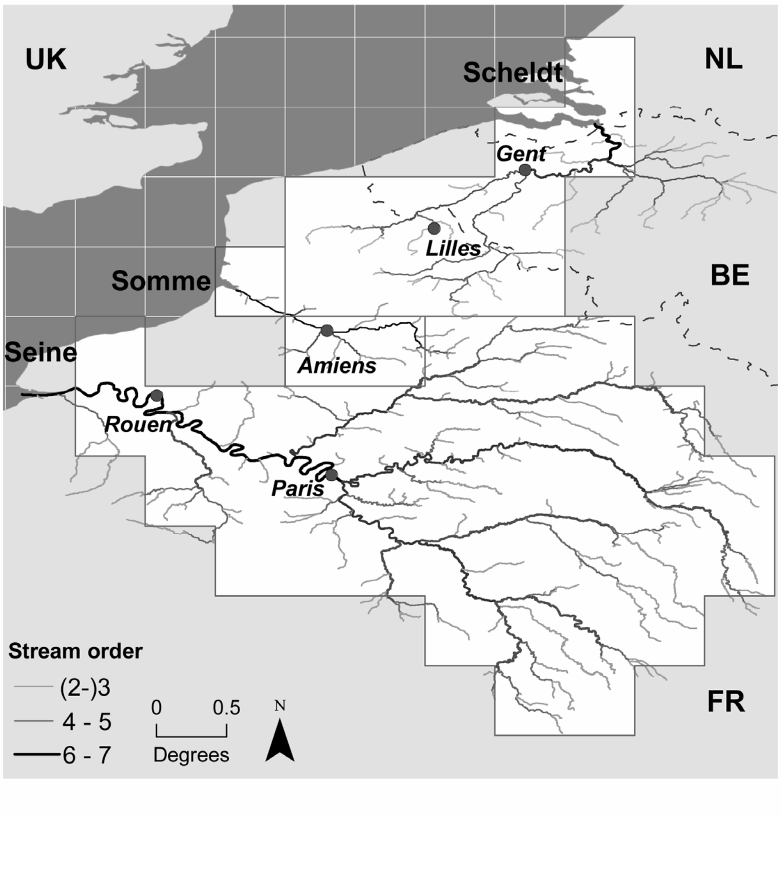


Figure 5-1: Map of the Seine, Somme, and Scheldt continental aquatic systems, as viewed by the Riverstrahler model (drainage network) and NEWS 2 models (basin scale). The grid size shown (0.5° × 0.5°) represents the elemental unit of the NEWS 2 models.

Beyond this clear general anthropogenic gradient, the spatial distribution of land use and human settlements as well as the morphological properties within each basin greatly differ. The central Paris conurbation accounts for the majority of the Seine population. Around it, a region of intensive cereal crop is followed at the periphery of the basin by an area of mixed cattle farming. The main sub-catchments join in the central part of the basin, where the Seine becomes a seventh-order river. Three large reservoirs (total volume of 750 million m^3) have been built to regulate the discharge upstream of the overcrowded Paris conurbation.

The fourth and last order of the Somme is deeply embanked by a chalky zone that crosses almost the entire basin. Dominated by areas of intensive cereal crop cultivation, the river comprises only three major urban centers that act as three main point sources scattered from upstream to downstream.

The Scheldt does not have a similar clear organization; rather, urban areas are spread throughout the basin resulting in a landscape of mixed urban and agricultural activities (Billen et al., 2005). Two systems that drain important cities (including Lille, in France, and Brussels, in Belgium) join in the very downstream part of the river (30 km from the outlet) to form its sixth order. Low-resolution databases, such as the Simulated Topological Network, with a spatial resolution of 30 minutes (STN-30, (Vörösmarty et al., 2000)), used by *NEWS* 2 models, do not depict this scheme of stream confluence in flat downstream areas. Accepting that such local-scale errors are inherent to the use of global dataset (unless time consuming local-scale manual corrections are made everywhere) our analysis of the Scheldt River system is therefore limited to its southern part, i.e., the upper-Scheldt and Leie Rivers and does not include the area drained by the Rupel affluent.

5.3. Comparison of the Global NEWS and Riverstrahler models

5.3.1. *NEWS* 2 watershed models

The Global *NEWS* group has elaborated several sub-models applied to more than 5000 river basins that simulate annual exports to the sea of different nutrient forms, including dissolved, particulate, organic, and inorganic forms of nitrogen and phosphorus (Beusen et al., 2005; Dumont et al., 2005; Harrison et al., 2005a; Harrison et al., 2005b). For dissolved forms, two main sources of nutrients are distinguished: point sources and diffuse sources. Point sources are related to wastewater flows, primarily in urban areas. Diffuse sources are primarily related to agricultural activities, such as livestock production and fertilized cropland, and to disturbances of natural ecosystems (e.g., atmospheric deposition of anthropogenic nitrogen). The models of dissolved forms, originally independently formulated, have been unified in *NEWS* 2 (Mayorga et al., 2009) in order to provide a coherent analysis of nutrient sources and exports from watersheds according to the following general-yield equation (1):

$$Yld = FE_{riv} . (RS_{dif} + RS_{pnt})$$

$$Yld = FE_{riv} . [(FE_{land} . WS_{dif}) + (FE_{wwt} . WS_{pnt})] \qquad (1)$$

Where WS_{dif} and WS_{pnt} are respectively diffuse and point watershed sources, partially emitted to river by the use of export-coefficients FE_{land} and FE_{wwt} (with "land" refering to landscape and "wwt" to wastewater treatment). The resulting river sources (RS_{dif} and RS_{pnt}) are exported to the basin's outlet, using the FE_{riv} export fraction. The latter expresses aquatic or in-stream retention and is determined by basin-scale calibrations and empirical parameterizations based on syntheses from the literature.

Nutrient sources are derived from empirical relationships that take into account population density, per capita gross domestic product and regional sanitation information for point sources, and on the basis of the IMAGE 2.4 (MNP, 2006) model output and regional agricultural information for diffuse sources. The gross watershed sources (WS_{dif} and WS_{pnt}) of nitrogen and phosphorus are assumed to be partially retained in the "landscape" before reaching the river network. This retention affects both diffuse (1- FE_{land}) and point (1- FE_{wwt}) sources, and account for the "terrestrial" retention of each nutrient form.

By distinguishing between terrestrial and aquatic retention, *NEWS* 2 models provide an intermediate level with which to assess the net emission of nutrients (RS_{dif} and RS_{pnt}) after their retention within wastewater treatment, soil, groundwater, and riparian areas, and are thus highly compatible with a drainage-network approach to river systems.

5.3.2. Riverstrahler drainage-network model

The Riverstrahler model (Billen et al., 1994; Garnier et al., 1999a) is based on a comprehensive description of processes occurring within the water column and involved in the transfer and retention of nutrients (Table 5-1). The model is extended to the entire drainage network, with in-stream transformation and retention processes explicitly calculated at the seasonal scale. The model assumes the system to be controlled by hydrological variations (with a 10-days resolution), morphological, and all land-based constraints, while the process kinetics involved in ecological functioning are assumed to be basically the same along the river continuum (see Garnier et al., 2002a for a detailed description). Implementation of the model thus relies on databases that include physical information on the drainage network and an accurate description of point and non-point sources as they are geographically distributed within the watersheds.

In contrast to *NEWS* 2 models, the Riverstrahler model only describes aquatic retention of nutrients, and not the processes occurring in watershed landscapes and soils. However, the role played by riparian areas is explicitly taken into account. Also, the consideration of a lower contamination of aquifers than sub-root water when defining the contributions of surface and base flows, allows to integrate groundwater retention. However, these terms are not mechanistically described and they can easily be by-passed, enabling a strict "drainage network description" with an aquatic retention term equivalent to the one considered by *NEWS* 2 model (1 - FE_{riv}) after terrestrial retention.

Table 5-1: State variables and processes taken into account by the Riverstrahler model.

State variables	**Processes**
Suspended matter	Sedimentation, resuspension
Phytoplankton (diatoms, non siliceous algae)	Photosynthesis, growth, respiration, lysis, sedimentation, erosion
Zooplankton	Grazing, growth, respiration, remineralization and excretion, mortality
Heterotrophic bacteria	Respiration, growth, mortality
Dissolved and particulate organic matter	Degradation (rapid or slow hydrolysis), remineralization
Dissolved oxygen	Photosynthesis, respirations, nitrification, benthic consumption
Ammonium, nitrate	Nitrification, denitrification, uptake, benthic recycling
Nitrifying bacteria	Planktonic nitrification
Organic P and adsorbed inorganic P	Algal uptake, benthic recycling, adsorption-desorption
Dissolved and biogenic silica	Diatoms uptake, biogenic silica dissolution
Fecal bacteria	Mortality

Another difference between the *NEWS* 2 and Riverstrahler models is their spatial resolution. While *NEWS* 2 semi-empirical models consider the watershed in its entirety (as a single feature), the elemental spatial unit of the Riverstrahler model is the incremental watershed area drained by a river reach between two confluences. Accordingly, these units can be described as a set of river branches with a spatial resolution of 1 km, or they can be aggregated to form upstream basins that are idealized as a regular scheme of tributary confluences (Strahler, 1957). This ability to adapt the spatial resolution of the drainage network is an advantage of the Riverstrahler model over the *NEWS* 2 approach to whole-river systems, especially when non-point and diffuse sources are not evenly distributed over the river basins under study. A further difference is the temporal scale. *NEWS* 2 models have an annual time scale, while the Riverstrahler model describes seasonal nutrient variability based on a 10-day resolution.

5.3.3. Upscaling the spatial resolution of the drainage network

As previously shown for sub-catchments of the Seine River (Oise River: (Ruelland et al., 2007)), in this study the sensitivity of Riverstrahler model simulations to spatial resolution is extensively assessed at the outlet of the Seine, Somme, and Scheldt drainage networks. Three different spatial resolutions were chosen. In the finer representation, all second-order sub-catchments were considered as individual basins (i.e., 645 sub-basins for the Seine, 30 for the Scheldt, and 24 for the

Somme), whereas stream orders greater than three were represented as branches (i.e., 6188 km of streams for the Seine, 530 km for the Scheldt, and 229 km for the Somme). Then, the resolution of the drainage network representation was decreased by analysing every fourth order as an individual sub-basin and by considering the different branches for orders 5 (Scheldt), 6, and 7 (Seine). In the third representation, each of the three river systems was treated as a single basin, as in the *NEWS* approach.

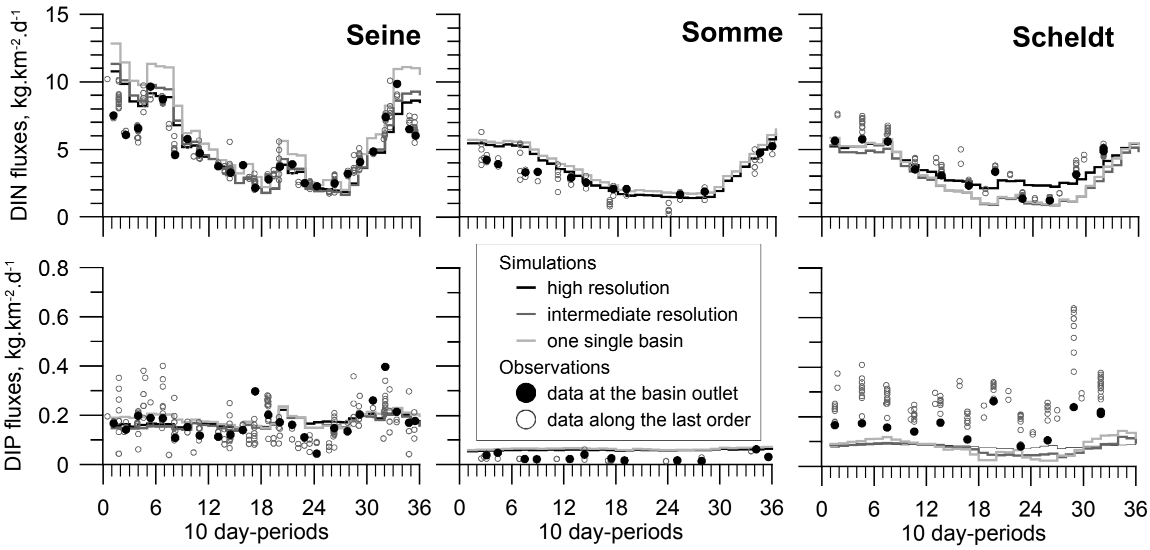


Figure 5-2: DIN and DIP flux exports to the sea, observed (dots) and calculated (line) as determined by the Riverstrahler model according to several representations of the drainage network for the year 2000: i) high, with the representation of each order-2 sub-basins, ii) intermediate, with the representation of each order-4 sub-basins, and iii) low, with the representation of the entire drainage network as a single basin.

Table 5-2: Detail of the high-resolution database used for the sub-regional assessment of the Seine, Somme, and Scheldt basins.

Model input	Spatial resolution	Data sources
Hydrology	8-km (PET and precipitation)	SAFRAN Grid, MétéoFrance
	5 meteorological stations	Belgian Royal Institute of meteorology
	X,Y location (for gauging stations)	Banque hydrologique
		VMM, Vlaasme Milieu Maatschappij
Morphology	90m grid cell	SRTM, Shuttle Radar Topographic Mission (NGA, NASA)
Land use	25 hectares (minimum surface)	CLC 2000 database (Corine Land Cover, EEA)
	Agricultural district (500 km² in average)	Farming practices (Mignolet et al., 2007)
		INS, Institut National Statistique (Belgian)
		RGA, Recensement Général Agricole
Population	X,Y location (of sewage release)	AESN, Agence de l'Eau Seine Normandie
		AEAP, Agence de l'Eau Artois Picardie
		VMM, Vlaasme Milieu Maatschappij
		SPGE, Société Publique Gestion de l'Eau

Dissolved inorganic fluxes of nitrogen (DIN), and phosphorus (DIP) were calculated and compared to observed data for the year 2000 (Figure 5-2) using high-resolution input data (Table 5-2). Good agreement between the different simulations with respect to the observed fluxes was found, despite an underestimation of DIP fluxes in the Scheldt. The observed seasonal trends were also correctly simulated. Within the Seine, the higher values of the DIN fluxes for a lower-resolution representation of the drainage network can be explained by the fact that reservoirs (conceptually connected to branches) could not be integrated in a single-basin representation of the drainage network so that their role in nutrient retention cannot be taken into account. Also, both surface and base flows were spatially averaged in the framework of a single-basin representation, thereby eliminating the local disparities and the seasonal variability of the total simulated flows, as observed for DIN and DIP fluxes in the Scheldt.

The reduction in the Riverstrahler model ability to simulate nutrient fluxes with decreasing resolution is relatively small and mainly related to the loss of information subsequent to the generalisation of the constraints by order. This degree of resolution is not appropriate to a detailed exploration of network contamination, but the invariance of the exported fluxes makes this upscaling process robust enough to allow comparison with the annual values provided by the global *NEWS 2* models.

5.4. Consistency of high- and low-resolution input data

A prerequisite for the sub-regional use of global input data is the consistency of low- and high-resolution sets of data (Table 5-2). This aspect was assessed for the three rivers by comparing nutrient exports from the terrestrial part of the watershed to the three river drainage networks, as calculated on the basis of similar hydrological constraints (see Section 5.1). For *NEWS* 2 models, these fluxes separate diffuse and point sources contribution, and represent the gross nutrient input to the river basin multiplied by a calibrated watershed-export coefficient (see Eq.1). For the Riverstrahler model they correspond to the net quantity reaching the river bank, once both riparian and groundwater retention are subtracted, while point sources contribution is directly accounted.

The results for nitrogen and phosphorus are remarkably comparable (Table 5-3). They reproduce the gradient of anthropogenic inputs from the Scheldt to the Seine and finally the less populated Somme river basin and depict the major changes (increase of N input and decrease of P input) that occurred from 1970 to 2000. Moreover the respective contributions of point and non-point sources are correctly represented for nitrogen. The apportionment of phosphorus source is however not kept with the global data in the 2000's, especially for the Scheldt basin where point emissions of P have rapidly decreased in the last three decade, resulting in a balance of point and diffuse sources (Table 5-3). For the *NEWS* 2 budget, it has to be noted that models of particulate forms do not allow attribution to point or non-point sources (Mayorga et al., 2009), so that particulate forms of nutrient have been entirely attributed to diffuse sources. This is a strong assumption in the assessment of phosphorus sources contribution only, as particulate nitrogen represents a small part of TN.

Table 5-3: Comparison of nutrient input to the Seine, Somme, and Scheldt Rivers on the basis of the high-resolution database integrated into the Riverstrahler models, and as provided by the NEWS 2 models after landscape retention for the years 1970 and 2000. 1970 inputs are derived from Billen et al., (2007b) for the Seine and from Billen et al., [2005] for the Schledt. For these two rivers, computation of diffuse sources is based on hydrological conditions derived from NEW2 (Scheldt) or highly similar (Seine: 173 mm/yr for the dry scenario in Billen et al., (2007b)).

			***NEWS* 2**			**Riverstrahler**		
			total sources (kg/km²/yr)	*point (%)*	*diffuse (%)*	*total sources (kg/km²/yr)*	*point (%)*	*diffuse (%)*
Seine								
	N	1970	990	47%	53%	1195	53%	47%
		2000	1272	35%	65%	1402	41%	59%
	P	1970	223	90%	10%	171	83%	17%
		2000	140	81%	19%	121	61%	39%
Somme								
	N	1970	-	-	-	-	-	-
		2000	917	26%	74%	798	23%	77%
	P	1970	-	-	-	-	-	-
		2000	82	74%	26%	77	31%	69%
Scheldt								
	N	1970	1815	43%	57%	1448	41%	59%
		2000	2365	37%	63%	1976	44%	56%
	P	1970	350	95%	5%	615	96%	4%
		2000	236	92%	8%	184	57%	43%

These results suggested that, despite different assumptions about the terrestrial retention of nutrients, and the use of a low-resolution database, *NEWS* 2 models are able to describe consistent inputs to the river system, that are highly comparable with those derived from high-resolution databases gathered for their integration into the Riverstrahler model. This finding justifies further work to downscale these values and to mechanistically assess the nutrients transferred and exported from the river.

5.5. Downscaling global input data

The translation of the global-scale constraints provided by the Global *NEWS* data into ones that are appropriate for the Riverstrahler model raises several methodological issues: (i) the state variables themselves differ between the two models; (ii) the basin-integrated constraints provided by Global *NEWS* have to be distributed according to stream orders to be used by the Riverstrahler model implemented to the whole basin (minimum resolution); (iii) the hydrological and climatic (temperature) constraints need to be spatially and temporally distributed. Table 5-4 provides a synthesis of the downscaling methodologies that concern hydrological and climatic constraints, point sources, and diffuse sources.

5.5.1. Hydrology and temperature

NEWS 2 considers annual runoff data derived from worldwide and long-term simulation of the Water Balanced Model, corrected with observed river discharges (Fekete et al., 2002; Feteke et al., 2009). For the year 2000, the values provided at global scale underestimate the hydrological regimes of the three rivers, e.g. 133,120 and 179 mm yr^{-1} respectively for the Seine, Somme and Scheldt river basin against 259, 240 and 328 mm yr^{-1} as obtained from observations or finer scale simulations (Thieu et al., 2009). Nevertheless, these values of global runoff stay in the range of values currently observed during dry years and reproduce the differences between the three basins, with higher runoff values for the Scheldt basin.

The Riverstrahler model uses the simulations provided by a simple rainfall-discharge model (see detailed description in (Le et al., 2007)) calibrated over several years of observed daily discharge enabling the partitioning of surface water and groundwater runoffs to be correctly reproduced, as an average fraction of the annual total runoff (Figure 5-3).

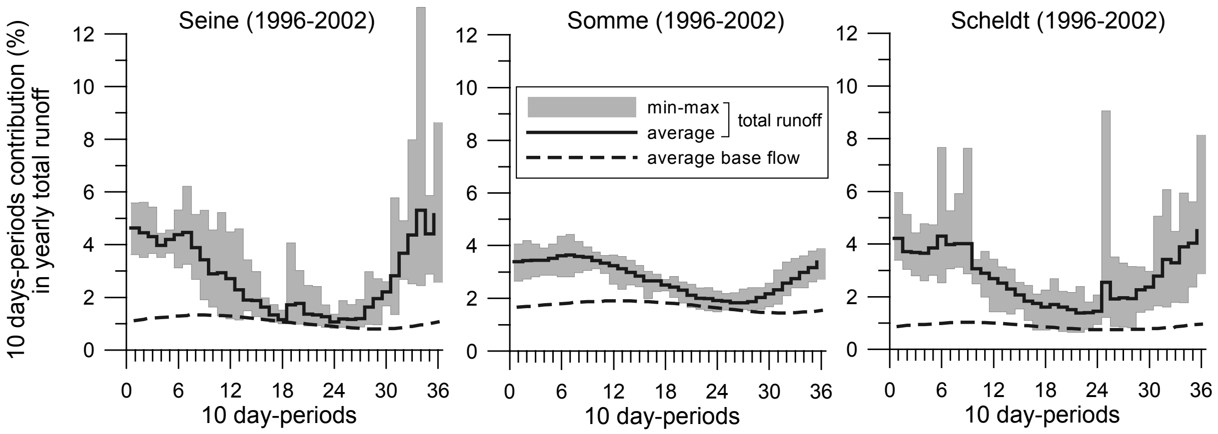


Figure 5-3: Seasonal distribution of the annual runoff, and partitioning between surface and groundwater flow, based on hydrological modeling (rainfall-discharge) simulation, calibrated for the period 1996–2002.

Also, a net water abstraction term is considered in the *NEWS* 2 approach. It is expressed as a fraction of the natural runoff and includes intake for irrigation and other types of human water consumption, both of which imply water loss from the river system. To ensure the consistency of hydrological inputs, this total water withdrawal was distributed by stream order in the Riverstrahler model. The Riverstrahler model uses a sinusoidal seasonal variation of temperature specifically described by stream order. For the different scenarios which differ in air-temperature increase with respect to 1970 (+1.3 to +1.4 in 2030 and from +1.5 to +2.0 °C in 2050 (Alcamo et al., 2006; Bouwman et al., 2009)) the same increase in mean annual water temperature has been imposed (see Table 5-4).

5.5.2. Diffuse sources to the drainage network

The diffuse nutrient sources included in *NEWS* 2 models (Bouwman et al., 2009) are here considered as net river inputs (i.e., after landscape retention), and the simulated dissolved forms of nutrients are inorganic and organic nitrogen (DIN, DON) and phosphorus (DIP, DOP), and organic carbon (DOC). Here, the particulate forms of nitrogen and phosphorus are not directly retrieved, as they are not a true input of the Riverstrahler model that derived particulate nutrient from suspended solid variable (see Table 5-4). The Riverstrahler model assesses diffuse sources by considering a constant mean composition, assigned to surface and groundwater flows respectively, according to land-use distribution within the watersheds. Accordingly, the variability of nutrient fluxes is entirely supported by the seasonal change in runoff, and the *NEWS* 2 annual nutrient loads can be easily converted to mean concentrations. Surface and groundwater contamination levels considered by the Riverstrahler model are assumed to be similar in order to avoid any apparent retention of nutrients in the aquifers, while riparian-retention terms are ignored.

The spatial apportionment of diffuse sources of nutrients by order is based on the analysis of the high-resolution constraints for the reference year 2000. An analysis of nutrient-flux distributions (in their *NEWS* 2 forms) revealed that nutrient proportions by stream order were very similar, regardless of the nutrient form studied, and strongly correlated with the proportion of watersheds drained by each order, without a clear influence of land-use type (R^2=0.99). Therefore, the surface area of watersheds drained by stream order (Figure 5-4) was used as a simple descriptor that enabled the downscaling of global diffuse-source values.

5.5.3. Point sources to the drainage network

The *NEWS* 2 and Riverstrahler models and databases consider human-waste emissions starting from their collection in sewage systems and disregard non-collected rural emissions, which are assumed not to reach the river network (Van Drecht et al., 2009). Based on an approach similar to the one used for diffuse sources, the distribution of point nutrient sources by stream order was analyzed on the basis of the high-resolution constraints for the year 2000. Population equivalents (Figure 5-4) were used as indicators to distribute points sources provided by the *NEWS* 2 models. Indeed, the point sources of nutrient distributed by stream order are highly correlated to population equivalent (R^2=0.88 to 0.99 for DIN and DIP respectively).

Table 5-4: Transposition of the NEWS model variables following the requirement of the Riverstrahler model. The downscaling methodology is primarily based on the distribution by order observed throughout the high-resolution database. For (2), the transition from nutrient loads (tons/yr) to nutrient concentration, the annual runoff value (1) was used. Point sources (3) depend of population densities connected to sewage system, this driver is commonly described by the both models.

NEWS forms	Riverstrahler requirement	Allocation rules/downscaling methodology
Hydrology (1) and temperature		
Annual runoff, natural value for the basin (mm/yr)	Superficial runoff by 10-day periods Groundwater runoff by 10-day periods	Runoff partitioning and seasonal distribution are based on hydrological model (Feteke et al., 2009) output calibrated on observed data (1996-2000)
Water consumption	Outflow daily discharge of water	Proportionally distributed as a mean withdrawal on each order
Mean temperature (°C)	T°C : mean temperature (°C)	$T°C = T_{mean} - T_{amplitude} . \cos(\omega . (t - 30))$ with $T_{amplitude}$= 6 to 9°C according to stream order and unchanged across scenarios; $\omega = 2 \pi / 365$; *t* in Julian day
River diffuse sources (2)		
DIN: dissolved inorganic nitrogen (load in tons/yr)	NO_3: nitrate concentration NH_4: ammonium concentration	Partitioning of DIN between NO_3^- and NH_4^+ based on a mean ratio by order (derived from high resolution database, see also(Thieu et al., 2009))
DIP: dissolved inorganic phosphorus (load in tons/yr)	TIP: total inorganic phosphorus concentration	TIP = DIP + cPIP . SM (Némery et al., 2005) cPIP=Pac . DIP / (DIP + KPads) exchangeable P content of soil Pac = 0.0055 gP.kg^{-1} (the saturation level) KPads = 0.7 mgP/l (adsorption half-saturation constant)
-	DSi : dissolved silica concentration	Constant value: 3.64 mgSi/l^{-1} (130 µmol/l) in agreement with observed values (Billen et al., 2007b; Meybeck, 1986)
-	BSi : biogenic silica concentration	Use of a mean content of 4.9 mgSi.g^{-1} of SM (Sferratore et al., 2006b)
DOC: dissolved organic carbon (load in tons/yr)	$DOC_{1,2,3}$: dissolved organic carbon (following 3 classes of biodegradability) concentration	Assuming an average partition of the DOC leached from soil: rapidly degradable (2%), slowly degradable (4%) and refractory (94%) (Servais et al., 1987)
TSS: total suspended solids concentration (no trapping)	SM :suspended matter concentration $POC_{1,2,3}$: Particulate organic carbon (following 3 classes of biodegradability) concentration	Variable common between the two models Assuming a mean carbon content of 10 g C kg-1 with a distribution of 3%, 12% and 85% respectively for rapidly degradable, slowly degradable and refractory
River point sources (3)		
DIN: dissolved inorganic nitrogen (load in tons/yr)	NO3: nitrate NH4: ammonium	Partitioning of DIN between NO_3^- and NH_4^+ based on a mean ratio by order (derived from the high-resolution database, see also Thieu et al. (Thieu et al., 2009))
DIP: dissolved inorganic phosphorus (load in tons/yr)	PO4: phosphate	Variable common between the two models
-	SM: suspended matter	Based on a theoretical release of 10 g. inhabitant^{-1}.day^{-1}
-	TOC :total organic carbon	Theoretical release of 4g C. inhabitant^{-1}.day^{-1} (Servais et al., 1999)
- -	DSi: dissolved silica BSi: biogenic silica	Use of a theoretical release 0.3 mg Si. inhabitant^{-1}.day^{-1} for dissolved silica and 0.5 mg Si.inhabitant^{-1}.day^{-1} from biogenic silica (Sferratore et al., 2006b)

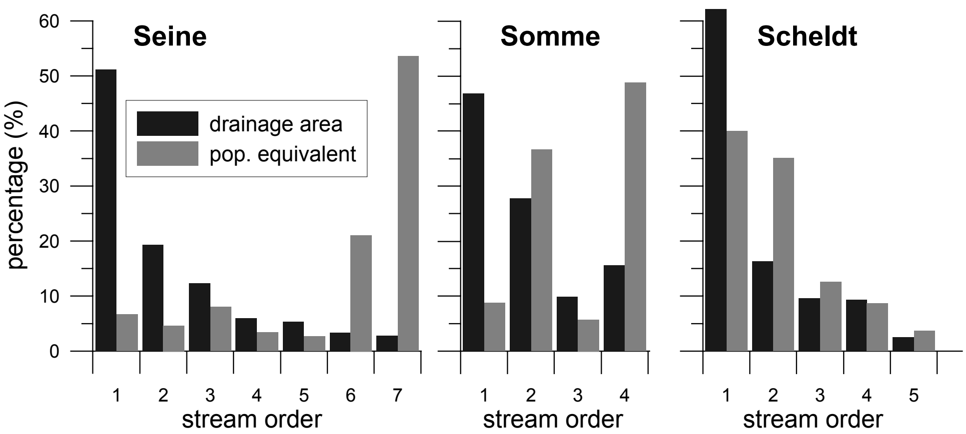


Figure 5-4: Distribution of drainage area and population equivalents by strahler order, as two synthetic indicators to describe the distribution of diffuse source and point source nutrient loading. This corresponds to the assessment of high-resolution information available for the year 2000.

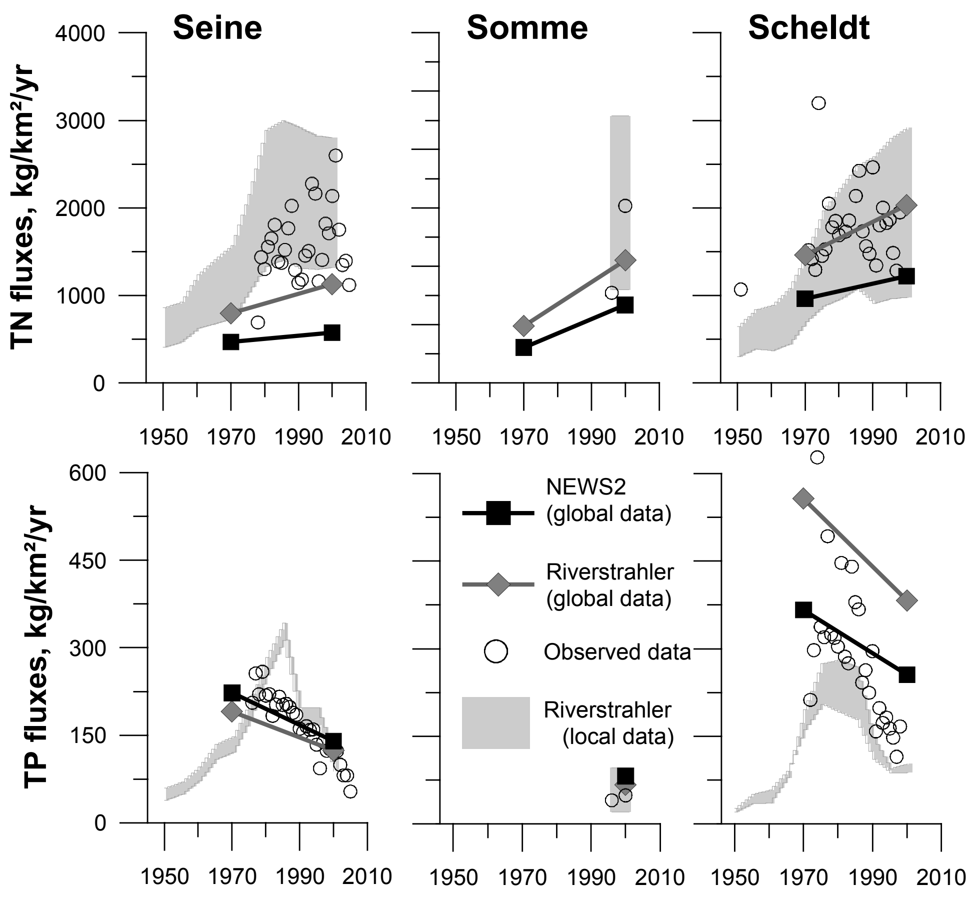


Figure 5-5: Nutrient (TN and TP) fluxes exported to the North Sea by the Seine, Somme, and Scheldt Rivers systems, as observed and simulated by i) the NEWS 2 models on the basis of global inputs (black line); ii) the Riverstrahler model on the basis of downscaled global inputs (gray line); iii) the Riverstrahler model on the basis of a high-resolution database for two extreme hydrological conditions (Billen et al., 2007b; Billen et al., 2005) (gray area). TP fluxes could not be calculated for the year 1970 for the Somme (see also Table 5-3) due to the lack of correct global scale point sources data for this period.

5.6. Deterministic approach of nutrient transfer based on downscaled global data

5.6.1. Validation for the period 1970–2000

The full *NEWS* 2 dataset was downscaled for the recent simulated period (1970–2000), following the previously described methodology, in order to implement the Riverstrahler model and then compare the nutrient transfers and exports simulated by the two models constrained by the same inputs.

The beginning of the period 1970–2000 was characterized by a rapid increase in nitrogen loads in response to the intensification of agricultural production and the construction of sewage systems collecting household effluents. At the end of this period, improvements in wastewater treatment contributed to the slower increase in nitrogen and phosphorus effluents. Of particular relevance was the prohibition of polyphosphates in washing powder, which led to the substantial decrease in phosphorus loading (Table 5-3).

The simulations by the two models are compared with observations (Figure 5-5). Finer scale simulations performed by the Riverstrahler model (Figure 5-5: gray area) and observed nutrient fluxes gathered over this period are provided to stress the amplitude of change in nutrient export fluxes. Although the two models provide the right trends (e.g. a clear increase in nitrogen and a rapid decrease in phosphorus and flux levels in the same order of magnitude between 1970 and 2000 (Figure 5-5), the NEW2 model tends to underestimate both N and P for the Seine and the Scheldt. This can be explained by the fact that hydrological conditions considered in these flux estimations are generally low for the global input data compared to those provided by the local ones.

For the Seine, Somme and Scheldt rivers, total phosphorous fluxes exported for the year 2000 were similar, respectively, 124, 68 and 383 kg $km^{-2}yr^{-1}$ according to Riverstrahler and 140, 82 and 255 kg $km^{-2}yr^{-1}$ according to *NEWS* 2. Although greater differences appear for the Scheldt with an impressive decrease in the amount of phosphorous exported to the sea, the two models runs enclose the large variability of observed data over this period.

For total nitrogen, Riverstrahler provided a better fit with the observed data, presuming that *NEWS* 2 models overestimate aquatic retention. For *NEWS* 2 models, aquatic retention only concerned the DIN variable (Dumont et al., 2005), as water consumption removed less than 1% of the respective runoff and no reservoir was considered. *NEWS* 2 estimates for nitrogen retention in the year 2000 were 697 kg $km^{-2}yr^{-1}$ for the Seine, 509 kg $km^{-2}yr^{-1}$ for the Somme and 1152 kg $km^{-2}yr^{-1}$ for the Scheldt i.e. half the total nitrogen inputs. The Riverstrahler process-based approach provided much lower aquatic retention values for nitrogen: 126 kg $km^{-2}yr^{-1}$ for the Seine, 50 kg $km^{-2}yr^{-1}$ for the Somme and 504 kg

$km^{-2}yr^{-1}$ for the Scheldt. Most of this retention (from 47 to 78%) was due to benthic denitrification, while losses caused by burial to deeper sediment played only a minor role.

The consistency of the Riverstrahler simulations supports i) the relevance of using such global input data in assessments of radical changes in nutrient loads, and ii) the benefit of the process-based high-resolution approach for calculating the fraction of nutrients exported to the sea.

5.6.2. Sub-regional analysis of the Millennium Ecosystem Assessment scenarios

The MEA proposed four scenarios of the world future that are structured around theoretical schemes of development, with various degrees of international integration and environmental concerns (Alcamo et al., 2006). The Global Orchestration (GO) scenario is characterized by high-level globalization, rapid economic growth, and a reactive rather than proactive approach to environmental issues. On the opposite end, the Adapting Mosaic (AM) scenario is based less on the international integration of economies; instead, it actively addresses environmental management at the regional scale, mostly through simple and inexpensive solutions. The Techno Garden (TG) scenario is, likewise, deeply involved in environmental issues but it is supported by global improvements in environmental technologies. Lastly, the Order by Strength (OS) scenario is less concerned with ecosystem management, focusing primarily on security and regional markets.

The three neighbouring basins, those of the Seine, Somme and Scheldt Rivers, are sources of consistent change with respect to the global future dynamics of industrialized countries (Bouwman et al., 2009; Van Drecht et al., 2009). As an example, population growth in the period 2000–2050 is predicted to increase similarly (14–14.8%) in the GO scenario, to slow down (1–4.3%) in the TG scenario, but decrease slightly (-6 to -10%) in the AM scenario and even more significantly (-14 to -18%) in the OS scenario.

Changes in diffuse nutrient sources can be appraised through the evolution of soil nutrient surpluses resulting from the balance achieved between gross sources (fertilizer, manure, crop fixation, atmospheric deposition) and withdrawal (crop export and animal grazing). These changes reflect the development of human activities and can be linked with the amount of nutrients exported to rivers (Riverstrahler model input), as terrestrial retention remains constant across the four scenarios.

Nitrogen surplus rapidly decreases within the more environmentally concerned scenarios (TG and AM), with an average of -57% in 2050, supported by an increase in fertilizer efficiency and an important decrease in atmospheric deposition; by contrast, the surplus is reduced by only -26% in the GO and OS scenarios (Table 5-5). Gross nitrogen export by agriculture increases in all scenarios; the only exception is the Scheldt, because of the predominance of livestock farming. Phosphorus diffuse sources, which represent only a small part of total phosphorus input to the rivers, follows the same line, with important decreases in the P surplus (-36 to -85%) in the TG and AM scenarios.

Table 5-5: Synthesis of the evolution of the main land-based drivers and sources according to the four Millennium Ecosystem Assessment scenarios: percentage values assessing change between 2000 and 2050. (*Diffuse-source values consider anthropogenic areas only; the "export" term includes crop export and animal grazing.)

			World development					
			Globalization			Regionalization		
Environmental management	Reactive		GO			OS		
			Seine	Somme	Scheldt	Seine	Somme	Scheldt
		Socioeconomic						
		Population (inhab km^{-2})	242 (15%)	114 (15%)	399 (14%)	180 (-14%)	85 (-14%)	284 (-19%)
		GDP (1995 US$ inh^{-1}yr^{-1})	65047 (180%)	65047 (180%)	66664 (181%)	53309 (130%)	53309 (130%)	50940 (115%)
		Urban pop. (%, ± change)	93.3 (+6%)	100 (+23%)	98.8 (0%)	93.3 (+6%)	100 (+23%)	98.8 (0%)
		Sewage connect. (%, ± change)	96.1 (+7%)	100 (0%)	96.9 (+2%)	96.7 (+8%)	100 (0%)	95.9 (+1%)
		Point sources						
		Raw N emission (kg km^{-2}yr^{-1})	1864 (46%)	882 (46%)	3263 (52%)	1327 (4%)	628 (4%)	2068 (-3%)
		N removed by sewage (%)	71.8	71.8	61.2	62.5	62.5	53.0
		Raw P emission (kg km^{-2}yr^{-1})	386 (24%)	182 (24%)	686 (50%)	270 (-13%)	128 (-13%)	423 (-8%)
		P removed by sewage (%)	81.8	81.8	69.8	72.3	72.3	61.1
		Diffuse sources(*)						
		Gross N source (kg km^{-2}yr^{-1})	9756 (-12%)	12170 (-5%)	15631 (-8%)	9845 (-12%)	11638 (-9%)	14210 (-16%)
		N export (kg km^{-2}yr^{-1}yr)	5988 (2%)	8253 (10%)	9489 (3%)	6226 (6%)	7841 (4%)	8337 (-10%)
		N surplus (kg km^{-2}yr^{-1})	3768 (-29%)	3917 (-26%)	6142 (-21%)	3619 (-31%)	3797 (-28%)	5873 (-24%)
		Gross P source (kg km^{-2}yr^{-1})	1404 (-4%)	1950 (13%)	2488 (5%)	1474 (1%)	1793 (3%)	2220 (-6%)
		P export (kg km^{-2}yr^{-1})	1069 (0%)	1497 (9%)	1655 (13%)	1126 (5%)	1413 (3%)	1452 (-1%)
		P surplus (kg km^{-2}yr^{-1})	335 (-13%)	453 (27%)	833 (-8%)	348 (-9%)	380 (6%)	768 (-15%)
	Proactive		TG			AM		
			Seine	Somme	Scheldt	Seine	Somme	Scheldt
		Socioeconomic						
		Population (inhab km^{-2})	214 (2%)	101 (2%)	354 (1%)	197 (-6%)	93 (-6%)	315 (-10%)
		GDP (1995US$ inh^{-1}yr^{-1})	58064 (150%)	58064 (150%)	59544 (151%)	52642 (127%)	52642 (127%)	53107 (124%)
		Urban pop. (%, ± change)	93.3 (+6%)	100 (+23%)	98.8 (0%)	93.3 (+6%)	100 (+23%)	98.8 (0%)
		Sewage connect. (%, ± change)	96.1 (+7%)	100 (0%)	96.9 (+2%)	96.1 (+7%)	100 (0%)	95.9 (+1%)
		Point sources						
		Raw N emission (kg km^{-2}yr^{-1})	1608 (26%)	761 (26%)	2816 (32%)	1449 (14%)	685 (14%)	2348 (10%)
		N removed by sewage (%)	71.8	71.8	61.2	62.5	62.5	53.0
		Raw P emission (kg km^{-2}yr^{-1})	332 (7%)	157 (7%)	591 (29%)	295 (-5%)	140 (-5%)	486 (6%)
		P removed by sewage (%)	81.8	81.8	69.8	72.3	72.3	61.1
		Diffuse sources(*)						
		Gross N source (kg km^{-2}yr^{-1})	8433 (-24%)	9979 (-22%)	13760 (-19%)	8004 (-28%)	10015 (-22%)	12567 (-26%)
		N export (kg km^{-2}yr^{-1}yr)	6454 (10%)	8163 (9%)	9352 (1%)	5821 (-1%)	7957 (6%)	8817 (-5%)
		N surplus (kg km^{-2}yr^{-1})	1979 (-62%)	1817 (-66%)	4408 (-43%)	2184 (-59%)	2058 (-61%)	3750 (-52%)
		Gross P source (kg km^{-2}yr^{-1})	1328 (-9%)	1611 (-7%)	2203 (-7%)	1182 (-19%)	1487 (-14%)	1909 (-19%)
		P export (kg km^{-2}yr^{-1})	1166 (9%)	1472 (7%)	1626 (11%)	1056 (-2%)	1434 (4%)	1536 (5%)
		P surplus (kg km^{-2}yr^{-1})	162 (-58%)	139 (-61%)	576 (-36%)	126 (-67%)	53 (-85%)	373 (-59%)

Nonetheless, future changes in point sources will deeply modify phosphorous releases as well as those of nitrogen, albeit to a lesser extent. The main differences between the scenarios are related to population and economic growth, both of which are rapid in the TG and GO scenarios. For our three industrialized watersheds, there is no significant difference between the scenarios regarding the level of connection to sewage treatment. The fraction of population connected increased slowly (1–2%) after the major improvements in sanitation that were made between 1970 and 2000. At the 2050 horizons of the TG and GO scenarios, wastewater treatment is improved, thereby removing a mean 70–82% of phosphorus and 61–72% of nitrogen. These efficiencies are lower within the OS and AM scenarios (61–73% for phosphorus removal and 53–63% for nitrogen removal). However while higher economic growth supports greater improvement of sewage treatment, the concomitant higher population growth is associated with an increase in raw emissions, which ultimately limits the differences between the scenarios. The decrease in nitrogen and phosphorous inputs to rivers is important compared to the levels of these nutrients exported in the year 2000, but there is little variability across the scenarios. Phosphorus exports to the river are 47–64 kg km^{-2}yr^{-1} for the Seine; 23–32 kg km^{-2}yr^{-1} for the Somme, and 129–163 kg km^{-2}yr^{-1} for the Scheldt.

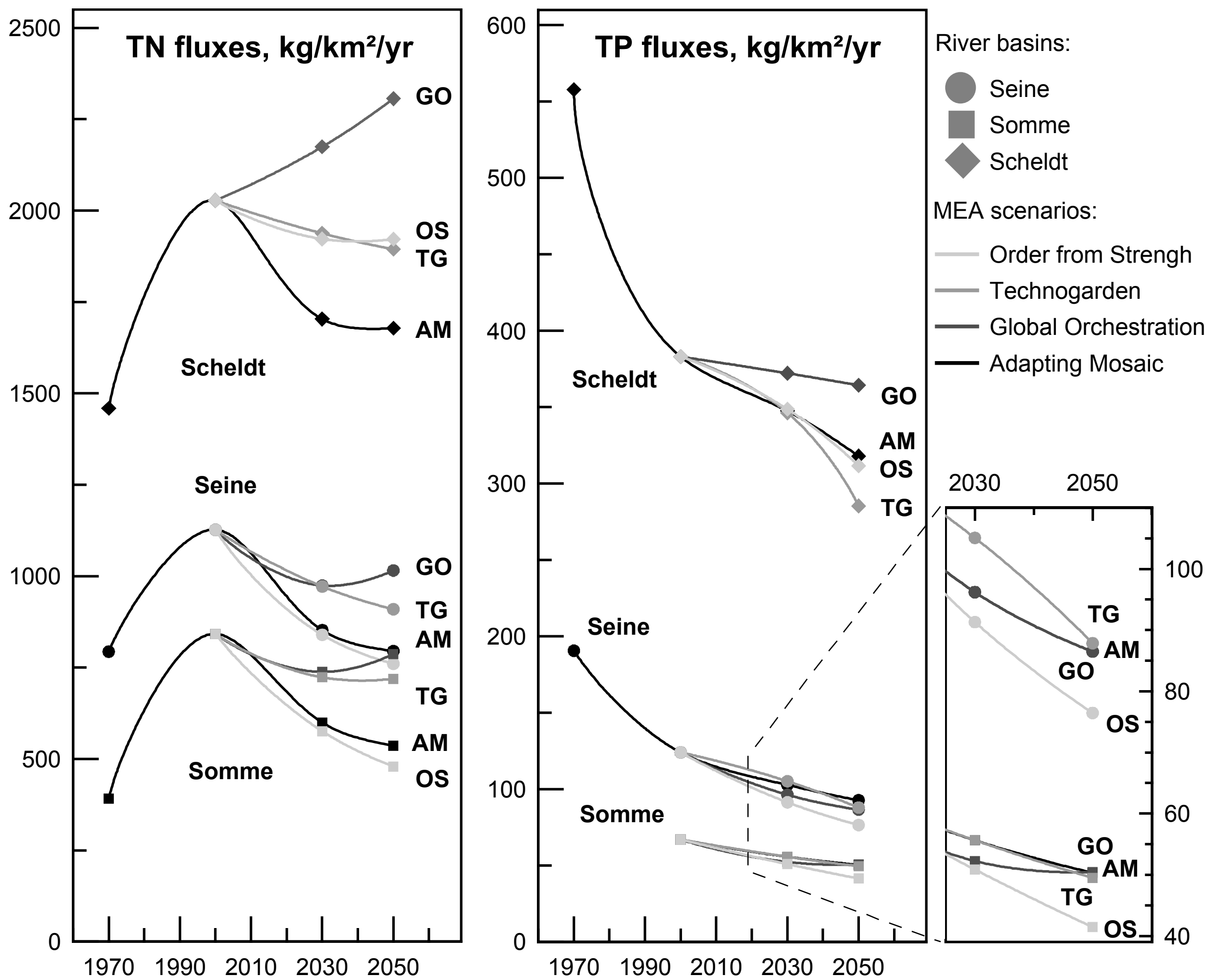


Figure 5-6: Total N and total P deliveries calculated by the Riverstrahler model according to the MEA global inputs (GO: Global Orchestration; OS: Order of Strength; TG: Techno Garden; AM: Adapting Mosaic) downscaled to the Seine, Somme, and Scheldt drainage network.

Despite a higher aquatic retention in the Scheldt, estimated to 52 % by the Riverstrahler model (compared to about 30% each for the Seine and the Somme), phosphorus levels at the outlet of the Scheldt remain two- to three-fold higher than the Seine and Somme deliveries (Figures 5-5 and 5-6). Indeed, water treatment is less advanced in the Scheldt, in agreement with the treatment efficiencies reported for the current period (Billen et al., 2005; Thieu et al., 2009). For example, phosphorus removal by sewage treatment increases from 50 to 61% between 2000 and 2050 in the Scheldt versus 59–72% for the other two basins during the same period. Also, the processes involved in aquatic retention and simulated by the model are not significantly affected across the scenarios, such that variations of fluxes at the outlets of the three basins reflect the changes influencing river inputs.

Table 5-5 provides an overview of the MEA's perspective of the impact of changes in "driving forces," and show that these changes do not seem to be followed by a spatial re-arrangement of human activities. For example, the fractions of urban population remain unchanged across the scenarios, there is no evident modification of the total areas of grassland or wetland, and even agricultural areas remain rather stable. Indeed, these sub-regional MEA storylines are translated into changes in the intensity of human disturbances, while spatial distributions within the catchment areas are similar across the scenarios.

5.6.3. Integrating the potential impact of the inner basin dynamics

The downscaling of the *NEWS* 2 constraints relies on the spatial distribution by stream order of synthetic indicators, namely, the watershed surface for diffuse sources, and the population for point sources. Both were defined for the year 2000, and the distribution rules were assumed to remain constant for future scenarios. One benefit of a mechanistic, spatially distributed approach lies in its ability to take into account changes in the spatial organization of human activities.

The example of population distribution is used here to illustrate the contrasts between the GO and AM scenarios. In accordance with their storylines, we assumed a redistribution of population between small and large towns. Thus, in the AM scenario, 25% of the population living in large agglomerations (over 100,000 inhabitants) moved to medium (between 20,000 and 100,000 inhabitants) and small (below 20,000 inhabitants) towns. In the GO scenario, the populations of large agglomerations increased by 25% at the expense of medium and small towns.

When transposed to the different orders of each of the three basins, these new distributions are determined by the proportion and the size of the respective urban centers, as well as their location along the stream order. In the AM scenario, the population is less concentrated and is relocated downstream along the Scheldt and to the upstream parts of the Seine. By contrast, in the GO scenario, there are greater disparities in population distribution. The case of the Seine is particularly impressive with respect to the growth of the Paris conurbation along the last order of the basin.

The responses of the three river systems to these changes in the distribution of point sources in the AM and GO scenarios were assessed with the Riverstrahler model. The trend to more-uniform distribution of point sources in the AM scenario translated into an increase in the aquatic retention of nitrogen and phosphorous. Phosphorous was more sensitive, with the calculated retention increasing from 22 to 26 kg $km^{-2}yr^{-1}$ for the Seine and from 5.7 to 6.2 kg $km^{-2}yr^{-1}$ for the Somme. In the GO scenario for the Seine and Somme basins, our assumption resulted in an increase in the population in the downstream part of the basins, thus decreasing aquatic retention with respect to the similar scenario with no spatial redistribution. For example, phosphorous retention decreased from 22 to 19 kg $km^{-2}yr^{-1}$ in the Seine and from 6 to 5.5 kg $km^{-2}yr^{-1}$ in the Somme. As the reference distribution of population within the Scheldt is opposite to that of the other two rivers, with a higher population upstream, the effect of the population re-allocation was also different from that of the two other basins: aquatic retention of both N and P increased by 2% in the GO scenario and decreased by 1% in the AM scenario.

5.7. Discussion and conclusions

Despite their four highly contrasting views of how the world will evolve over the next few decades, the scenarios provided in the framework of the MEA are based on consistent and plausible assumptions. Sub-regional use of global input data has provided a useful assessment of nutrient sources and successfully reproduced the contrasts observed between the Seine, Somme, and Scheldt Rivers. These rivers differ strongly in their population densities and agricultural orientations. However, the proximity and similar level of development of these three industrialized basins contributed to limit the differences in the final assessment of the scenarios in terms of nutrient delivery at the outlets. In addition, the low water discharge values used from global dataset also tend to minimize these differences.

The methodology proposed here for downscaling global inputs does not introduce further assumptions about regional changes in the main driving forces, thus allowing a comparison of the results by the two models. However, in the present work, the impact of future climate change on hydrology was possibly largely underestimated. Changing annual runoff values in the MEA scenarios (-2% and 2%) were not accompanied by changes in the seasonal distribution of the surface and groundwater contributions (Figure 5-3). These potential changes in the seasonal variability of rivers discharge, include an increase in diffuse sources contribution during winter and a decrease of precipitation in summertime that might concentrate point sources inputs and also influence water residence times. For the Seine River, *Ducharne et al.* (Ducharne et al., 2007) have demonstrated that these seasonal changes in discharge might have a negligible impact under climate change scenario, while higher impact should be expected from the warming of the water column. The Riverstrahler model includes temperature as an important process driver acting mainly on the dynamics of bacteria, phytoplankton and zooplankton communities (nutrient uptake, growth, and mortality) (Garnier et al., 1995). This controlling factor is not considered in the application of the global *NEWS*2 models to the different prospective scenarios; it is taken into account in the Riverstrahler application although the sensitivity of the annual total nitrogen and phosphorus delivery is very low (less 2%).

Another key factor affecting the impact of the MEA scenarios on river nutrient export is the internal changes expected to occur within the basins and their impact on nutrient retention. We have demonstrated the necessity of a spatially distributed approach to describe the transfer of human wastewater release within a basin. The change in river nutrient retention is directly linked with the increased residence time of the water mass (Seitzinger et al., 2002b). However, several other aspects of proactive environmental management, such as restoration of natural stream morphology (enlargement of the river bed, connection of lateral arms) and flow regime (Muhar et al., 1995; Poudevigne et al., 2002), could supply the AM or TG scenarios with additional features.

The Riverstrahler mechanistic approach is better adapted than the *NEWS* 2 models to integrate the link between biogeochemical processes and morphological constraints, or the spatial organization of the landscape at the basin scale. However, the use of global-scale models remains essential for integrating the development of socio-economic driving forces acting at the global scale and the major dynamics transcending the limits of river basins.

Starting from the ability of the Riverstrahler model to be upscaled to a single-basin representation, we demonstrated that the *NEWS* 2 estimates of nutrient loads transferred to the river network are consistent and that the data can be downscaled on the basis of simple descriptors (population distribution and watershed area by stream order). A comparison of the simulated fluxes over the period 1970–2000 emphasized the benefit of an approach linking "global-empirical" modeling of nutrient transfer from the source to the river with a "sub-regional, spatially distributed and process-based" approach of in-stream retention.

Here, the purpose was not only to provide a sub-regional assessment of the MEA or to compare the models' performances for the three sample watersheds; rather, a further aim was to analyze the suitability of this global information with respect to the requirements of modeling approaches at more-detailed scale levels. The scalable Riverstrahler model has already been successfully applied to the analysis of several river systems across the world: the Red River (Vietnam: (Le et al., 2005; Le et al., 2010)), the Kalix sub-arctic basin (Swedish: (Sferratore et al., 2008)), and the Danube (Garnier et al., 2002a). The methodology presented here could be transposed to these or other basins in the world, supporting the idea that a mechanistic approach can be applied at the global scale provided that adequate information is available.

Acknowledgments

We thank the Global *NEWS* network for providing widely open access to their results at the global scale. In particular, we thank Carolien Kroeze and Lex Bouwman for their comments on earlier versions of the paper. This work was supported by the PIREN-Seine (CNRS, UMR Sisyphe), the Thresholds (European integrated project) and TIMOTHY (Belgian Science Policy) programs.

Troisième partie

Approche intégrée du continuum rivière – zone côtière

Les zones côtières disposent naturellement d'apports importants en sels nutritifs leur permettant de soutenir une production primaire dite 'nouvelle', essentielle au développement des niveaux trophiques supérieurs. Cette richesse confère aux eaux côtières une dimension écologique importante. Elles constituent des zones de reproduction privilégiées et leur fonction de nourricerie permet d'accueillir la croissance des juvéniles de nombreuses espèces (Le Pape et al., 2003). La dimension sociétale de ces zones est également significative de par les usages récréatifs (tourisme bleu) qu'elles proposent.

Cependant, les zones côtières sont des milieux de transition complexe à la fois soumis à l'influence des eaux marines et des eaux douces d'origine terrestre. Les dimensions (largeur, profondeur) et la morphologie (encaissement, découpage de la côte, bathymétrie) conditionnent l'interaction entre les eaux du large appauvries en sels nutritifs et les apports fluviaux enrichis, ainsi que le temps de résidence de ces masses d'eau dans la zone côtière. L'éclairement est un autre facteur contrôlant la productivité des écosystèmes côtiers. La fin de la période hivernale se marque par l'installation de la thermocline, qui supporte dans sa partie superficielle (épilimnion) une grande partie des activités autotrophes. Cette stratification est souvent le facteur déclenchant des « blooms » phytoplanctoniques. Ces efflorescences algales sont dans un premier temps siliceuses (diatomées) et prélèvent les nutriments azote, phosphore et silice dans les proportions moyennes définies par Redfield et al., (1963) C :N :P :Si = 106 :16 :1 :20. La prédominance des diatomées s'explique notamment par une physiologie plus adaptée aux faibles températures (Garnier et al., 1995). La régénération des nutriments ainsi prélevés est assurée par la dégradation de la matière organique produite aux différents niveaux de la chaine trophique. Cependant, la re-dissolution de la silice biogénique est bien plus lente que la re-minéralisation de l'azote et du phosphore. Ainsi, la production algale utilisant ces nutriments régénérés est essentiellement non-siliceuse.

A cela vient s'ajouter le déséquilibre des apports terrestres qui sont particulièrement appauvris en silice, durant la période estivale, du fait de la faiblesse du régime hydrologique limitant les apports diffus des bassins (principale source de silice). L'étude théorique menée par Billen et Garnier (1997) illustre parfaitement l'installation de ces épisodes de carence, favorables à la croissance d'espèces non-siliceuses (Figure III-1).

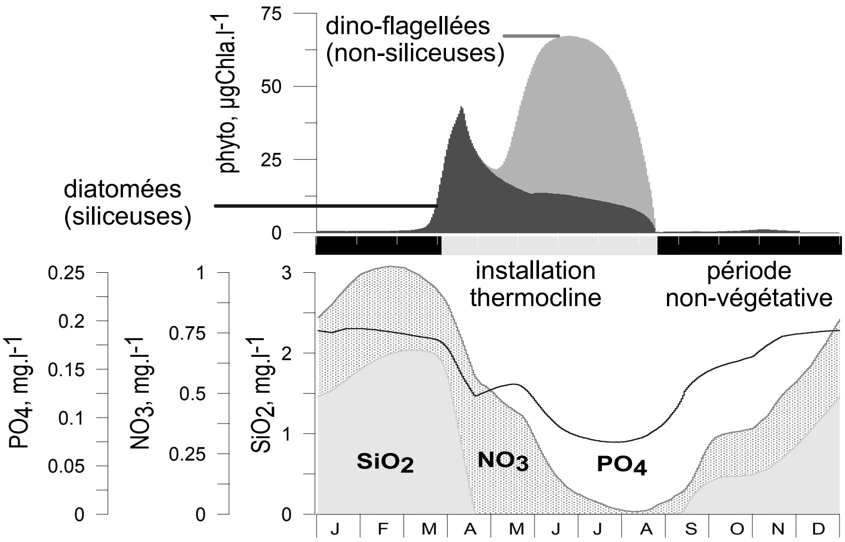


Figure III-1 : Succession des productions diatomées et dino-flagellées en lien avec la disponibilité des sels nutritifs. Données extraites de (Billen and Garnier, 1997)

Dans la Baie Sud de la Mer du Nord, ces blooms sont principalement composés de Phaeocystis qui se développent sous forme de colonies, dont la taille importante (>400µm), empêche leur consommation par les copépodes (zooplancton). Sous l'effet des vagues, cette espèce protégée par un mucus gélatineux, peut former des accumulations de mousse sur le littoral (Lancelot, 1995). Le fait que cette production primaire, ne soit pas directement consommable par le méso-zooplancton, entraine une perte d'efficacité trophique dans le milieu. Ces blooms sont observés depuis plus d'un siècle, avec une augmentation nette de leur durée depuis les années 1970 (Cadée and Hegeman, 1991). Leur intensité est par ailleurs soumise à des variations interannuelles importantes (Lancelot et al.). L'apparition d'un premier bloom diatomique, appauvrissant le milieu en silice dissoute, permet l'efflorescence de Phaeocystis dans un milieu enrichi en azote et phosphore (Rousseau et al., 2002).

Les précédentes parties ont permis de caractériser le fonctionnement des trois principaux hydrosystèmes contribuant à l'enrichissement de la Baie Sud de la Mer du Nord, mais aussi d'apprécier l'effet d'une modification des usages terrestres sur les flux de nutriments exportés à la côte. L'analyse des rapports molaires de ces flux terrigènes a été considérée dans un premier temps comme un indicateur potentiel du déséquilibre s'opérant au niveau de la zone côtière.

Cependant, la saisonnalité des développements algaux, dans ces espaces intermédiaires à l'hydrodynamisme complexe, nécessite une approche de modélisation couplée représentant plus finement les dynamiques saisonnières en jeu.

L'approche de modélisation MIRO (Lancelot et al., 2005) permet, en simulant les cycles biogéochimiques du carbone, de l'azote, du phosphore et de la silice, d'étudier la dynamique des efflorescences de Phaeocystis. Dans sa configuration multi-box (0D), les apports anthropiques terrigènes sont intégrés directement dans deux sections côtières considérées homogènes. La première (FCZ, French coastal zone) reçoit la contribution de la Seine et de la Somme. La seconde (BCZ, Belgian coastal zone) est soumise aux apports directs de l'Escaut et aux apports indirects de la FCZ.

La similarité des approches de modélisation de MIRO et de Riverstrahler, leur résolution (échelle des interactions micro- et macroscopique), ainsi que le partage des variables d'état, permet un couplage opérationnel des deux modèles. Différentes études ont déjà considéré ce couplage (Lancelot et al., 2007; Lancelot et al., 2009). Toutefois, elles n'avaient pas intègré la contribution de la Somme et ne portaient que sur des travaux de modélisation rétrospectifs et actuels. Le passage à une description plus fine et spatialisée des activités (cf. première partie) sur les bassins, justifie d'actualiser ces travaux, en intégrant l'ensemble des trois bassins. Il est alors possible pour le modèle Rivestrahler-MIRO de simuler i) la mise en œuvre prospective d'une réduction des émissions anthropique, ii) la modification des flux de nutriments exportés à la zone côtière, et iii) l'impact sur les niveaux de développement des diatomées et non-diatomées (Phaeocystis).

Ce sixième chapitre présente l'aboutissement du travail réalisé dans le cadre du projet européen Thresholds, qui vise à caractériser l'état des écosystèmes aquatiques et leur résistance aux perturbations. Dans notre cas d'étude, il s'agit notamment de déterminer les seuils de nutriments à ne pas dépasser à l'exutoire des bassins Seine-Somme-Escaut, pour prévenir et/ou réduire l'eutrophisation côtière. Cette recherche interdisciplinaire a été réalisée grâce à une collaboration avec les équipes ESA (Ecologie des Systèmes Aquatiques) et CEESE (Centre d'Etudes Economiques et Sociales de l'Environnement) de l'Université Libre de Bruxelles.

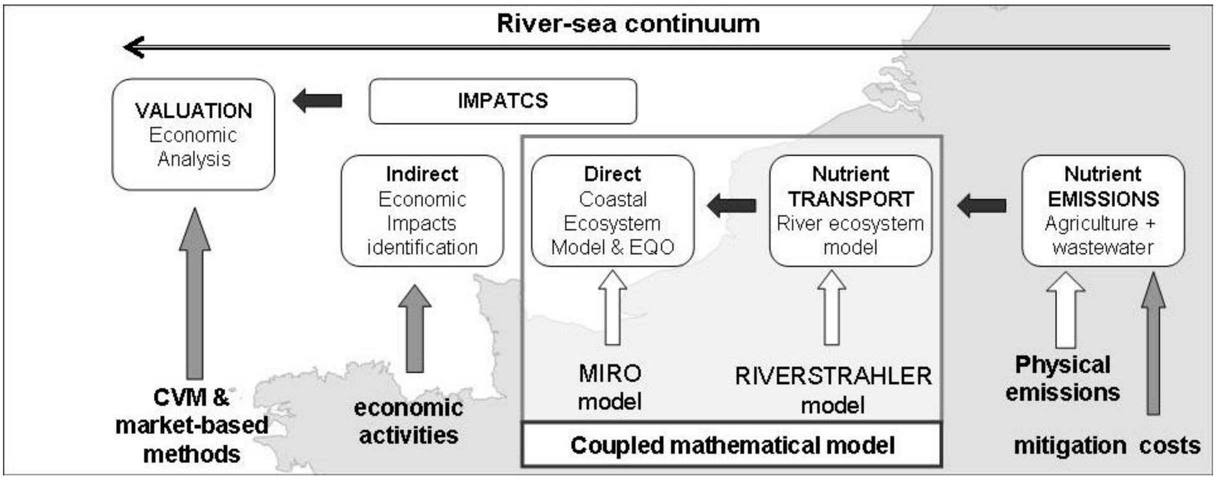


Figure III.2 : approche intégrée du continuum rivière – zone côtière, établie en collaboration avec les équipes de l'ULB-ESA et –CEESE.

La démarche intégrée (Fig. III-2) propose, au travers du couplage Riverstrahler-MIRO la simulation du transfert des nutriments depuis leurs émissions jusqu'à leur export à la zone côtière, où l'impact direct de ces émissions est évalué en terme d'abondance et de durée d'efflorescence (réponse écologique). Des scénarios visant à réduire les émissions de nutriments sont implémentés à l'échelle des bassins versants et évalués économiquement. L'objectif a été de rapprocher l'efficacité écologique et économique des différentes mesures de remédiation envisagées.

CHAPITRE

6 - Evaluation écologique et économique d'une réduction des développements algaux en Mer du Nord

Ecological and economic effectiveness of nutrient reduction policies on coastal Phaeocystis colony blooms in the Southern North Sea: an integrated modeling approach

Lancelot, C.[1], Thieu, V.[2], Polard, A.[3], Gypens, N.[1], Garnier, J.[2,4], Hecq, W.[3],Billen, G.[2,4]
Submitted to Regional Environmental Change

[1] Université Libre de Bruxelles, Ecologie des Systèmes Aquatiques, CP-221, Bd du Triomphe, B-1050 Brussels, Belgium
[2] Université Pierre et Marie Curie, Sisyphe, 4 Place Jussieu, F-75005 Paris, France
[3] Université Libre de Bruxelles, Centre d'Etudes Economiques et Sociales de l'Environnement, CP 124, Avenue Jeanne 44, B-1050 Brussels, Belgium
[4] CNRS, UMR 7619 Sisyphe, 4 Place Jussieu, F-75005 Paris, France

Abstract

Phaeocystis blooms occur every spring in the Eastern Channel and Southern North Sea as a result of nutrients discharged by the Seine, Somme and Scheldt rivers. These blooms affect the structure and functioning of the coastal ecosystem and cause environmental damage visible as foam accumulation on beaches. In spite of intense research effort, little is known on the present-day magnitude of the phenomenon and how it might change in response to future nutrient reduction policies. In this paper we apply an integrated modeling approach that links the undesirable *Phaeocystis* phenomenon to changing human activity in the three watersheds. The tool results of the coupling between the SENEQUE-RIVERSTRAHLER model, an idealized biogeochemical GIS-based model of the river system implemented in the Seine, Somme and Scheldt watersheds and the biogeochemical MIRO model describing diatom and *Phaeocystis* blooms in the marine coastal domain. Model simulations explore how nutrient reduction options regarding diffuse and point sources would impact on the *Phaeocystis* colony spreading in the coastal area. The reference and prospective simulations are performed for the year 2000 characterized by mean meteorological conditions, and nutrient reduction scenarios include and compare upgrading of wastewater treatment plants and changes in land use and agricultural practices. A cost-effectiveness analysis is performed for each nutrient reduction scenario. Further the reduction obtained for *Phaeocystis* blooms is assessed by comparison with ecological indicators (bloom magnitude and duration) and the cost for reducing foam events on the beaches is estimated. The added effect of meteorological conditions on *Phaeocystis* reduction is discussed.

6.1. Introduction

Impressive foam layers attesting for coastal ecosystem imbalance cover every beach along the eutrophied Eastern Channel and Southern North Sea in springtime. This well-known event is due to the accumulation of ungrazable forms of *Phaeocystis globosa* colonies that succeed to silicon-limited spring diatoms (Rousseau et al., 2002) while responding to nitrogen-excess anthropogenic river loads (Lancelot, 1995), and spread over the whole area along a SW-NE gradient (Lancelot et al., 1987). In agreement, *P. globosa,* here reported as *Phaeocystis* is considered as an indicator species of marine ecosystem disturbance (Tett et al., 2007) and recommendations for decreasing its abundance to numbers of non-problem areas and good ecological status have been made in the scope of the implementation of the OSPAR Strategy to combat eutrophication (OSPAR, 2005), the Water Framework (WFD, 2000/60/EC), and Marine Strategy (MSD, 2008/56/EC) Directives of the European Union. This implies the definition of a *Phaeocystis* reference condition from which the foam event could be scaled as well as the understanding of the link between *Phaeocystis* colony blooms and anthropogenic loads (Lancelot et al., 2009). These are two necessary conditions for implementing nutrient reduction strategies to mitigate *Phaeocystis*-derived foam events. They are however not sufficient as the needed nutrient reductions to be obtained by upgrading wastewater treatment plants and/or changing agricultural practices have to be technically feasible and economically sustainable.

In this paper we present an original integrated impact assessment methodology to valuate the ecological and economic effectiveness of realistic nutrient mitigation scenarios applied in the North Sea watershed over a short-term scale. The tool is the coupled river-ocean model SENEQUE/RIVERSTRAHLER-MIRO (hereafter referred as SR-MIRO) that calculates on the one hand nutrient export at the river outlet as a function of nutrient inputs to the drainage network (Billen and Garnier, 1999; Garnier et al., 2002a; Thieu et al., 2009) and on the other hand the ecological response of the *Phaeocystis*-dominated coastal sea to the simulated river loads (Lancelot et al., 2007; Lancelot et al., 2005). This tool has been successfully applied in the Belgian coastal zone (BCZ) for the reconstruction of diatom-*Phaeocystis* blooms over the 1950-1998 period (Lancelot et al., 2007; 2009).

Here, as a further implementation of the SR-MIRO mathematical tool, our goal is to test and to estimate the monetary cost of several realistic nutrient mitigation scenarios (upgrading of wastewater treatment plants, use of different agricultural practices and their combination; Thieu et al., 2010) on the nutrient and ecological quality status of the BCZ. To our best knowledge this is the first study that assess at the same time the technical, ecological and cost effectiveness of possible short-term nutrient reduction policies implemented at a regional scale of an aquatic continuum, from the headwater streams to the coastal zone. Most existing papers either focus on the cost effectiveness of nutrient reduction at the river outlet without exploring the benefits for the coastal ecosystems (e.g. Gren et al., 1997; Wustenberghs et al., 2008) or ignore the technical feasibility of nutrient reduction to the sea (Atkins and Burdon, 2006; Lise, 2004; Nunneri et al., 2007).

Nutrient enrichment of the BCZ results from the input of transboundary (SW-Atlantic waters enriched by the rivers Seine and Somme) and local (mainly the Scheldt river) sources of land-based nutrients (Fig. 6-1) and is influenced by human activity on the three river basins (3S for Seine, Somme and Scheldt). The simulated domain therefore includes the 3S watershed (about 100 000 km²; Thieu et al., 2009) and the coastal eastern Channel and Southern North Sea up to the northern limit of the Belgian exclusive economic zone EEZ (Fig. 6-1).

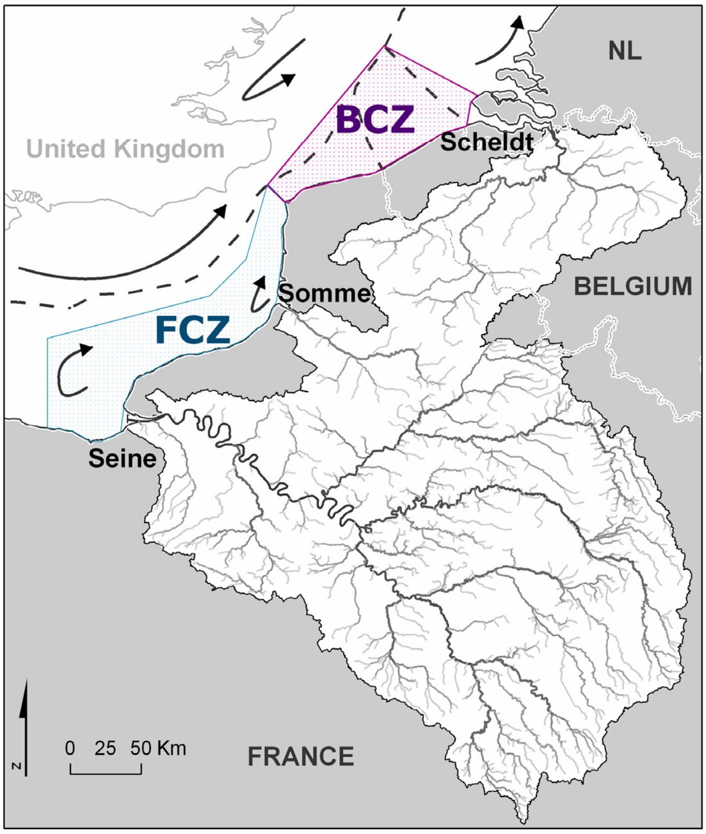


Figure 6-1 RS-MIRO implementation geographical domain: the 3S watersheds, the coastal eastern Channel and Southern North Sea. Dotted line indicates the boundaries of the Exclusive Economic Zone (EEZ)

The ecological effectiveness of each nutrient mitigation scenario is assessed in terms of nutrient enrichment and *Phaeocystis* colony blooms reduction. The eutrophication status of the BCZ is appraised based on a comparison between simulated winter nutrients and OSPAR recommendations for the BCZ (winter $NO_3 \leq 15$ mmol m^{-3}; winter $PO_4 \leq 0.8$ mmol m^{-3}) while the threshold of 4 10^6 cell L^{-1} is used as ecological reference for scaling *Phaeocystis* disturbance and hence foam events. Based on field observations and model simulations, this colonial cell density corresponds to a healthy ecosystem characterized by an efficient transfer of *Phaeocystis* production to higher trophic levels (Lancelot et al., 2009). Besides the integrated modeling approach, a major issue to the present study is to estimate the cost effectiveness of nutrient mitigation in terms of nutrient reduction to the sea as well as in terms of reduction of the ecological impact here expressed by the resulting change in *Phaeocystis* magnitude and duration. The simulated maximum colony cell density determines bloom magnitude while the duration corresponds to the number of days during which *Phaeocystis* cell abundance is higher than 4 10^6 L^{-1}.

6.2. Material and methods

6.2.1. The SENEQUE/RIVERSTRAHLER-MIRO (SR-MIRO) model

The mathematical tool results of the coupling between RIVERSTRAHLER - a deterministic biogeochemical model calculating nutrient transformations in the river system and their export to the coastal sea as a function of hydrological, morphological and anthropogenic constraints (Billen and Garnier, 1999; Garnier et al., 2002a; Thieu et al., 2009) – and MIRO – a deterministic biogeochemical model calculating carbon and nutrient cycling and diatom/*Phaeocystis* colony successions in the coastal sea (Lancelot et al., 2007; Lancelot et al., 2005).

For this application the RIVERSTRAHLER model has been spatially implemented in the three river basins (Seine: 76370 km^2; Somme: 6190 km^2 and Scheldt: 19860 km^2) as a combination of 46 sub-basins and 14 main axes (see Thieu et al., 2010 for details), through the GIS-based SENEQUE tool (Ruelland et al., 2007). This suite of tool and model relies on a general spatial database covering the 3S watersheds and describing nutrient emission as point and non point sources. Point sources of nutrients i.e. wastewater discharges, are calculated from information on the effective treatment capacity for each plant, the type of treatment and the geographical position of the release. Diffuse sources of nutrients are transferred to the river system as decadal surface and base flows respectively characterized by sub-root and groundwater constant concentrations for each nutrient. RIVERSTRAHLER includes a deterministic module (RIVE) of biogeochemical processes in the water column and at the sediment- water interface (a total of 30 state variables) describing eutrophication processes and nutrient transformations in the river system (equations and parameters in Garnier et al., 2002a; and updated in Thieu et al., 2009).

MIRO, also a high-resolution trophic model including 38 state variables, was developed to assess and understand eutrophication problems associated to *Phaeocystis* blooms in coastal seas (Lancelot et al., 2005). Details on model state variables and parameterization are available on www.int-res.com/journals/suppl/appendix_lancelot.pdf. Some parameterization of diatom growth was however updated based on the exhaustive review of Sarthou et al. (2005). These concern the half-saturation of nutrient uptake now at 0.6, 0.5 and 0.03 mmole m^{-3} for Si, N and P respectively as well as the Si summer diatom stoichiometry with Si:C now at 0.02 mmolSi:mgC.

RIVE and MIRO models are based on similar chemical and biological principles making relevant and easy their coupling (Lancelot et al., 2007). In order to consider the indirect and direct input of nutrient river loads to the BCZ, MIRO is implemented in a multi-box frame delineated on the basis of the hydrological regime and geographical position of river outlets. Two boxes are therefore considered: the French coastal zone (FCZ; Fig. 6-1) receiving inputs from the inflowing Atlantic waters and the Seine and Somme rivers and the BCZ receiving inputs from the FCZ and the Scheldt river (Fig. 6-1). Ocean boundary conditions at the Southern French border of the simulated domain are provided by MIRO simulations obtained in a closed system (Lancelot et al., 2005). River nutrient (C, N, P, Si) inputs are generated by the SENEQUE/RIVERSTRAHLER model on a 10-daily basis and are linked to MIRO state variables as described in Lancelot et al. (2007).

6.2.2. SR-MIRO simulations

6.2.2.1. References

Year 2000 has been chosen as present-day reference (RF) in view of its average meteorological conditions (global solar radiation, temperature, rainfall), the best documentation of human pressures for the three watersheds (Thieu et al., 2009) and the availability of nutrient and phytoplankton data in the central BCZ (Breton et al., 2006) for validation of the SR-MIRO tool. In addition a pristine reference state (PR) has been constructed to characterize natural features of the BCZ in terms of nutrient conditions and ecological functioning. The pristine state was established by erasing all human alterations, including anthropogenic nutrient sources and hydraulic management. Forest covers the whole landscape and leaf litter along the river bank supplies the only allochthonous organic input and associated nutrients.

6.2.2.2. Nutrient reduction short-term scenarios

Three series of nutrient reduction scenarios were considered: those testing the upgrading of wastewater treatment plants (WWT), the implementation of different agro-environmental measures (AGR) and the combination of the two types of measures (MIX). The choice of the different nutrient reduction scenarios was based on different guidance documents (Interwies et al., 2004; Jacobsen, 2007; Kranz et al., 2004; MVW, 2005; Wateco, 2003). For the agricultural scenarios, the focus was given to nitrogen management, because this element is determining the magnitude of *Phaeocystis* colony blooms in the studied area (Lancelot, 1995). Furthermore, while nitrogen-control practices tend also to decrease phosphorus release, phosphorus mitigation methods often have little effect on nitrogen (Howarth, 2005).

The different WWT scenarios were implemented in compliance with the EU WFD recommending the improvement of N and P removal by the implementation of nitrification, denitrification and dephosphatation processes. Such an upgrade of wastewater treatment corresponds to a N and P reduction of respectively 70 and 90% (EU directive on urban wastewater treatment 91/271/EEC). Due to this, WWT scenarios include the progressive implementation of the largest plants (treatment capacity >500,000 inhabitant equivalent (IE); WWT1) followed by plants with lower treatment capacity (>100,000 IE: WWT2; >20,000 IE: WWT3) up to the application of N and P removal improvement to all identified urban effluents (WWT4). These scenarios were established on the basis of the actual sanitation situation of the three river basins for the year 2000 (RF).

Agro-environmental scenarios (AGR) respond to the Council Directive 91/676/EEC (nitrate directive). The first scenario tested for decreasing nutrient leaching from agricultural land (AGR1) involves the introduction of nitrogen trapping winter crops (NTWC) and the decrease of fertilizer use. AGR1 implementation is based on a homogeneous typology of the 3S agricultural landscape, making use of agronomical census and results (model simulation and local experiments) available for N trapping efficiency of NTWC (Thieu et al., 2010). The second agro-environmental scenario (AGR2) introduces a modification of the agricultural landscape by substituting fodder corn crops with grazed meadow, and no modification of the cattle number. Nitrogen leaching from grazed meadows is then estimated from the new cattle density according to Vertes et al. (2002). Due to the inertia of groundwater reservoirs, the short-term assessment of AGR1 and AGR2 is based on change in surface runoff composition only. The resulting nitrate concentration estimated for the surface waters of the concerned 3S area are detailed in Thieu et al. (2010).

6.2.2.3. Nutrient reduction long-term scenarios

Three additional scenarios were conducted to assess the long-term effect of the implementation of agro-environmental measures on the quality of the water base flux. In these scenarios, the nitrate contamination of the aquifer is obtained after a delay that corresponds to the infiltration time, assuming a long-term equilibrium between the surface and groundwater concentrations. Assuming that drastic change of agricultural practices is required to significantly reduce N excess, BEST scenarios mimic the shift to organic agriculture of all agricultural fields of the 3S watershed. Based on existing literature, two probable values were tested for characterizing diffuse sources of NO_3 from biological agriculture: 6 (BEST1) and 3 (BEST2) mg N L^{-1} (Thieu et al., subm.). BEST scenarios also assume improvement of N and P removal in WWT plants (capacity >20,000 IE) as described above in the WWT2 scenario.

Results obtained here are appraised by comparison with a 'Business As Usual' scenario (BAU) that maintains the status of year 2000 for land base sources of nutrients over the long-term, with groundwater nitrate concentrations assumed to have reached equilibrium with infiltrating surface agricultural water. BEST scenarios will not be economically valuated as its implementation involves a complete change of the agricultural landscape towards an integrated farming-breeding system fully satisfying the needs of local crops with no use of mineral fertilizers (Thieu et al., subm.).

6.2.3. Cost valuation of the nutrient reduction scenarios

The costs valuation methodology follows the guidelines of Marlowe et al. (1999). All costs are established for 2008, in relation to a projection of the existing situation, i.e. the situation in which the environmental protection measure has not been implemented (the "baseline case"; Marlowe et al., 1999). Therefore, only additional costs actually incurred, relative to the baseline case are taken into account. This cost assessment focuses on direct costs with no consideration of indirect costs. Direct costs are established based on the sector (i.e. the driving force in the Driver-Pressure-State-Impact-Response (DPSIR) model) where the nutrient reduction scenario is implemented; indirect costs are supported by the other sectors (Wateco, 2003). The predicted increase of energy and fuel costs is not considered in the cost assessment of WWT and AGRI scenarios, following the guidelines of Marlowe et al., (1999) for comparing pollution protection measures in "real price" terms.

The cost valuation methodology depends on the type of nutrient mitigation scenario and applies to technical measures. The upgrade WWT scenarios correspond to end-of-pipe technical measures while the agricultural scenarios are process-integrated ones. The only purpose of end-of-pipe measures is to reduce environmental damage, so the entire investment expenditure, operating and maintenance costs can be regarded as pollution control scenario costs (environmental costs). Process-integrated measures concern the whole production process and if savings arise which are greater than the environmental component then the environmental costs will be taken into account if the payback time of the measure is longer than 3 years (VROM, 1998).

6.2.3.1. WWT improvement

Two types of cost have to be distinguished for the assessment of WWT scenario costs: (i) the financial costs for retrofitting the plants to tertiary treatment facilities and (ii) the costs for operating a tertiary treatment plant (as compared to a secondary treatment plant). The financial cost has been annualized with the following expression:

$AEC = [NPV * a] / [1 – (1 + a)-D]$ (European Communities, 2003)

In which AEC is the annual equivalent cost; NPV is the net investment value; a is the discount rate (5 %); D corresponds to the lifetime of the capital equipment (20 years) (Santo et al., 2006 and VITO, SPGE experts advices).

The added operating costs due to the introduction of tertiary treatment were considered identical for the 3 S watersheds. This figure varies between 0 and 4 € per IE, according to the treatment already existing in the WWTP. For instance, a full nitrogen treatment increases operating costs by 1 € per IE (C. Didy, pers. com. 2008) while the physico-chemical treatment of phosphorus corresponds to an increase of 2.5 € per IE (J.-P. Karpinsky, pers. com. 2008).

The investment costs are highly variable, depending on several criteria such as the WWTP localization, the chosen treatment technology and the type of material used (for the same technology).

Moreover, some WWTPs of the 3S watershed had to be fully upgraded to tertiary treatment while other just managed to reach the standards of the EU directive on urban wastewater treatment. A cost range of 115- 425 € per IE was applied, taking into account three main criteria for each WWTP: the construction year, the capacity of the plant and the wastewater treatment in process in 2000. These costs only concern improvements in the plants but not in the collection and sewage network.

6.2.3.2. Agro-environmental measures

The total annual cost of the agricultural scenario AGR1 has been estimated by summing those related to the reduction of nitrogen fertilization and to the use of a NTWC.

The unit cost of the nitrogen fertilization reduction measure includes the market value of yield loss and the savings on the purchase of nitrogen fertilizer:

$$\text{Unit Cost } (€\ ha^{-1}) = ((Ybs * PM) – (Ys * PM)) – (CFbs – CFs)$$

In this equation, Y is the crop yield (t $ha^{-1}yr^{-1}$), PM is the crop market value (€ t^{-1}) and CF is the fertilizer cost (€ ha^{-1}), bs and s refer respectively to the baseline (RF) and the nitrogen mitigation scenario. The crop-specific unit costs were taken from Lacroix et al. (2005) and are based on the results of the STICS model (Brisson et al., 2003). The total annual cost of the nitrogen fertilization reduction has been then estimated from unit costs given for wheat, winter barley and sugar beet and an average cost for the other crops, multiplied by the corresponding agricultural crop surfaces of the 3S basins (Thieu et al., 2009).

The further introduction of NTWC into the usual crop rotation involves different cost items such as the farming cost of sowing, growing, cropping, burying and seed and machinery cost, all based on a literature review. The average seed cost was assessed at 20 € ha^{-1} (Colot et al., 2007; Labreuche et al., 2007; Préault, 2004) to which 10 € ha^{-1} were added to consider the additional farming costs of the introduction of NTWC (Labreuche et al., 2007). The annual cost of NTWC was then estimated on this base of 30 € ha^{-1} and considering the spring crop surfaces in the 3S basins (Thieu et al., 2009).

For the economic valuation of extensive farming (AGR2), we compared gross margins between two types of cattle breeding farms: farms with grasslands as single fodder crops (AGR2), and farms with corn silage crops and grasslands (RF scenario). Gross margin corresponds to the production value minus variable costs, i.e. operating costs. These costs concern for instance seeds, fertilization, pesticides products and labor force; they represent variable inputs needed for production.

As a first hypothesis we assumed identical fixed costs (equipments, land…) but a variable cattle production between the two types of farming. It has been previously shown that under such conditions the replacement of maize by mowed and grazed grassland for cattle feeding is more cost effective (Alard et al., 2002; Béranger and Journet, 2003; Deprez et al., 2005; Institut de l'élevage, 2007; Marsin, 2008; RAD, 2004; Rohellec and Mouchet, 2008). We therefore conclude that this scenario has no annual cost, even though the first stages of extensive farming implementation generate a surplus of work and also punctual costs (i.e. fencing grazed meadows). It is expected that short-term disadvantages of such changes will result in benefit in the medium term.

6.3. Results

6.3.1. SR-MIRO validation

Figure 6-2 compares SR-MIRO simulations of nutrients and phytoplankton obtained in the BCZ in 2000 with time-series data collected the same year in the central BCZ at St 330 (51°26.05 N; 002°48.50 E). As a general trend, the coupled SR-MIRO model describes remarkably well the observed diatom-*Phaeocystis*-diatom succession, in time and magnitude (Fig. 6-2e,f). The corresponding seasonal cycle of nutrients is simulated as well (Fig 6-.2a,b,c). Yet the NO_3 spring depletion is not fully captured by the model (Fig. 6-2a), also reflected in the simulation of chlorophyll a (Chl *a*, an indicator of phytoplankton biomass) that fails to reproduce the magnitude of the observed spring maximum (Fig. 6-2d).

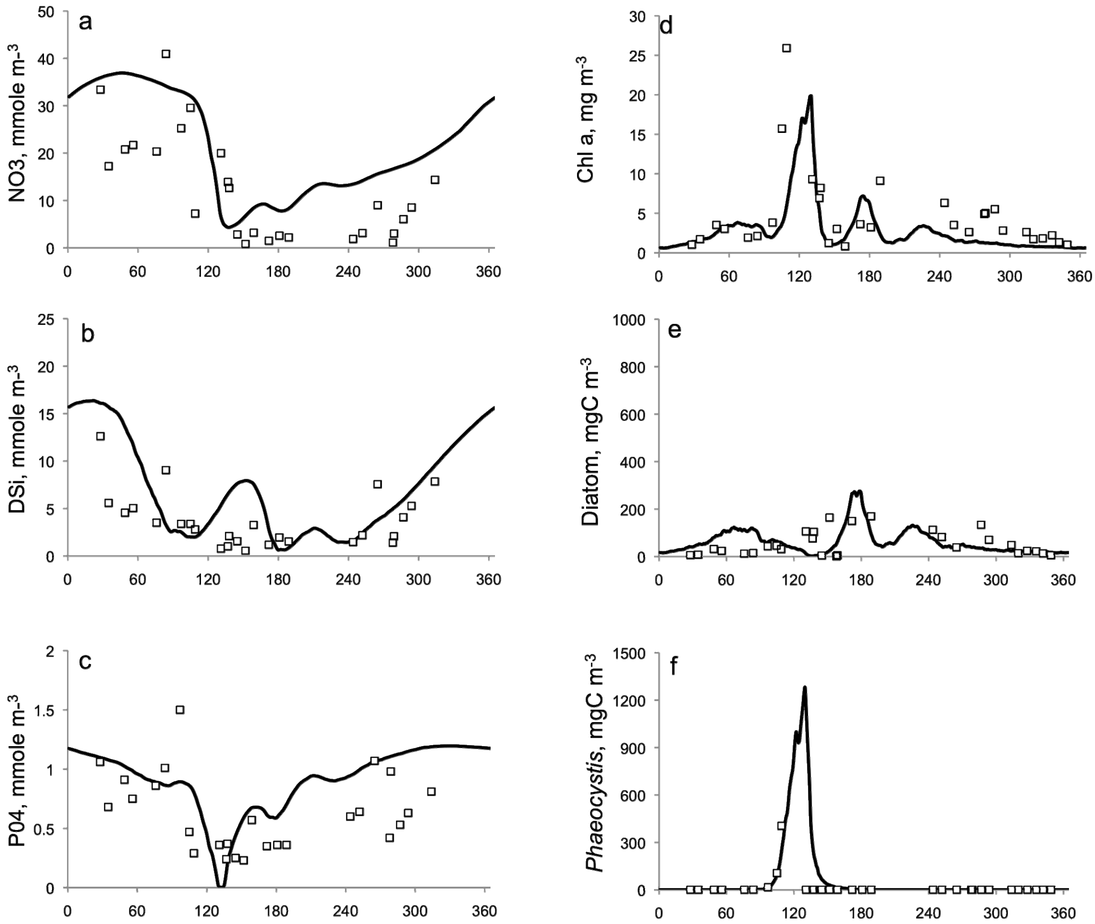


Figure 6-2 SR-MIRO simulations and observations of NO_3 (a), DSi (b), PO_4(c), Chl *a* (d) diatoms (e) and *Phaeocystis* colonies (f) in the BCZ for year 2000

A close examination of the SR-MIRO seasonal cycle of Chl *a* and of diatoms (Fig. 6-2d,e) suggests that the lack of NO_3 simulated depletion might be explained by a delay in the build up of the second diatom bloom (Fig. 6-2e) also reflected in a parallel slight overestimation of DSi remineralization (Fig. 6-2b). The ten-day resolution of the SENEQUE/RIVERSTRAHLER model results might well explain such a short delay. Another plausible explanation comes from the better competition of *Phaeocystis* colonies for low PO_4 (Fig. 6-2c,f) which is the limiting nutrient of phytoplankton growth in 2000 (Fig. 6-2c,d).

With this exception, we consider the SR-MIRO tool well validated for the purpose of our work, considering its ability to correctly simulating the DSi-controlled diatom growth (Fig. 6-2b,e) and the PO_4-controlled *Phaeocystis* growth (Fig. 6-2c,f),

6.3.2. Nutrient reduction at the 3S river outlets

SENEQUE/RIVERSTRAHLER (SR)-simulated nutrient loads obtained for different short- and long-term mitigation scenarios are compared to pristine (PR) and present-day (RF) simulations (Table 6-1). A particular attention is given to N and P loads as, Si input to the river system and at the outlet is not significantly modified by human activity (Garnier et al., 2002b; Sferratore, 2006). Clearly, for all scenarios, the river Seine is the major contributor of land-based N and P (78-80%) to the simulated marine domain, followed by the Scheldt (15-18%) and the Somme (3-4%). The current anthropogenic N and P inputs (RF) for 2000 are respectively one order of magnitude higher and three times higher than in the pristine (PR) reference (Table 6-1).

The improvement of wastewater treatment allows a significant decrease of P loads (19-47 %; Table 6-1) by all rivers, the maximum reduction being reached for the Seine (Table 6-1). N loads simulated for the different WWT scenarios show however little reduction (1 to 8%).

In contrary, agro-environmental scenarios are predicting a substantial reduction of N loads while P loads are slightly positively affected if not (Table 6-1). Yet AGR scenarios are decreasing diffuse particulate P but the concomitant reduction of suspended solid inputs results in a lower adsorption of PO_4 that affects positively the total P loads. The general application of the NTWC scenario (AGR1) in combination with extensive farming (AGR2) gives a significant reduction of N loads for the 3 basins (AGR3, about 20%; Table 6-1).

As expected, the combination of WWT upgrading and of agro-environmental practices (MIX) allows the best reduction of both N and P on the short-term. A similar N reduction of 26-27% is simulated at the outlet of the 3S basins (Table 6-1). The P load reduction varies between basins, being the highest at the Seine outlet (43%), the lowest for the Somme (23%) and intermediate for the Scheldt (33%). Interestingly enough, MIX scenarios which mimic the best technically feasible nutrient reduction measures to be implemented, are not able of achieving -on the short-term- the 50% reduction of N loads to the Greater North Sea, as arbitrarily recommended by OSPAR already in 1988

(PARCOM 88/2, 1988) making use of 1985 loads as reference (Lancelot et al., 2007). On the contrary the recommended 50% reduction of P loads is already achieved by the solely WWT3 scenario as P originates mostly from point sources.

Table 6-1: The integrated SR-MIRO approach: SENEQUE-RIVERSTRAHLER simulations of nutrient loads at the 3S river outlets: references and mitigations scenarios as used for the MIRO model. PR: Pristine, RF: 2000, WWT1: WWTP>500,000 IE; WWT2: WWTP>100,000 IE; WWT3: WWTP>20,000 IE; WWT4: WWTP > 0 IE; AGR1: NTWC; AGR2: Extensive farming; AGR3: AGR1+AGR2; MIX: AGR3+WWT3; BEST: full biological agriculture + WWT3 with NO_3 diffuse at 6 (BEST1) and 3 (BEST2) mg N L^{-1}.

Scenarios	**Seine**		**Somme**		**Scheldt**	
	N, kt y^{-1} (%RF)	P, kt y^{-1} (%RF)	N, kt y^{-1} (%RF)	P, kt y^{-1} (%RF)	N, kt y^{-1} (%RF)	P, kt y^{-1} (%RF)
References						
PR	15	2.1	0.7	0.11	3.5	0.46
RF	149	6.7	8.1	0.27	28.7	1.62
Short-term						
WWT1	142 (-4.6)	4.6 (-32)	8.1 (0)	0.27 (0)	28.3 (-1.3)	1.42 (-12)
WWT2	141 (-5.4)	4.2 (-38)	8.0 (-1.4)	0.22 (-19)	27 (-5.8)	1.26 (-22)
WWT3	139 (-6.7)	3.9 (-43)	8.0 (-1.4)	0.21 (-22)	26.6 (-7.2)	1.08 (-33)
WWT4	137 (-7.9)	3.6 (-47)	7.9 (-2)	0.20 (-27)	26.6 (7.5)	1.03 (-36)
AGR1	124 (-17)	6.8 (1)	6.4 (-21)	0.28 (+1)	25.3 (-12)	1.65 (+2)
AGR2	145 (-2.3)	6.7 (0)	7.8 (-3.3)	0.27 (0)	26.4 (-7.2)	1.63 (+1)
AGR3	121 (-19)	6.8 (0)	6.2 (-23.3)	0.28 (+1)	23.6 (-17.6)	1.66 (+3)
MIX	110 (-26)	3.8 (43)	6.0 (-26)	0.21 (-23)	21 (-27)	1.09 (-33)
Long-term						
BAU	169 (+14)	6.8 (0)	9.9 (+22)	0.27 (0)	31 (+7.1)	1.62 (0)
BEST1	82 (-45)	3.8 (43)	4 (-50)	0.21 (-23)	19 (-34)	1.10 (-32)
BEST2	58 (-61)	3.8 (+43)	2.7 (-67)	0.21 (-21)	14 (-50)	1.10 (-32)

The 50 % reduction of present N loads by the Seine, the Somme and the Scheldt to the coastal sea is however achieved on the long-term when implementing organic agriculture on the basins and upgrading WWT with a capacity of >20,000 IE (BEST2; Table 6-1). The achievable N reduction of the Scheldt load is generally lower that in the Seine and Somme (Table 6-1) because of a smaller disparity between surface and ground water concentrations (Fig.3 in Thieu et al., 2010).

The obtained achievement is best appraised when comparing N loads projected at the 3S outlet by the BEST scenarios to the BAU scenario, i.e. for the nutrient loads simulated on the long-term when the actual waste water treatment and agricultural status (RF) are maintained on the 3S watershed. Clearly the lack of further nutrient reduction measures makes the situation worse by increasing significantly the N loads at the outlets of the 3S watershed (14, 22 and 7 % for the Seine, Somme and Scheldt respectively; Table 6-1).

6.3.3. Nutrient status of the eutrophied BCZ

In the absence of human alteration (SR-MIRO scenario PR), the inflowing Atlantic waters weakly enriched by continental nutrients discharged by the Seine and to a less extent by the Somme (Table 6-1), are contributing up to 98% of the N and P loads (indirect nutrient sources) to the BCZ making insignificant the direct contribution by the Scheldt (Fig. 6-3ab).

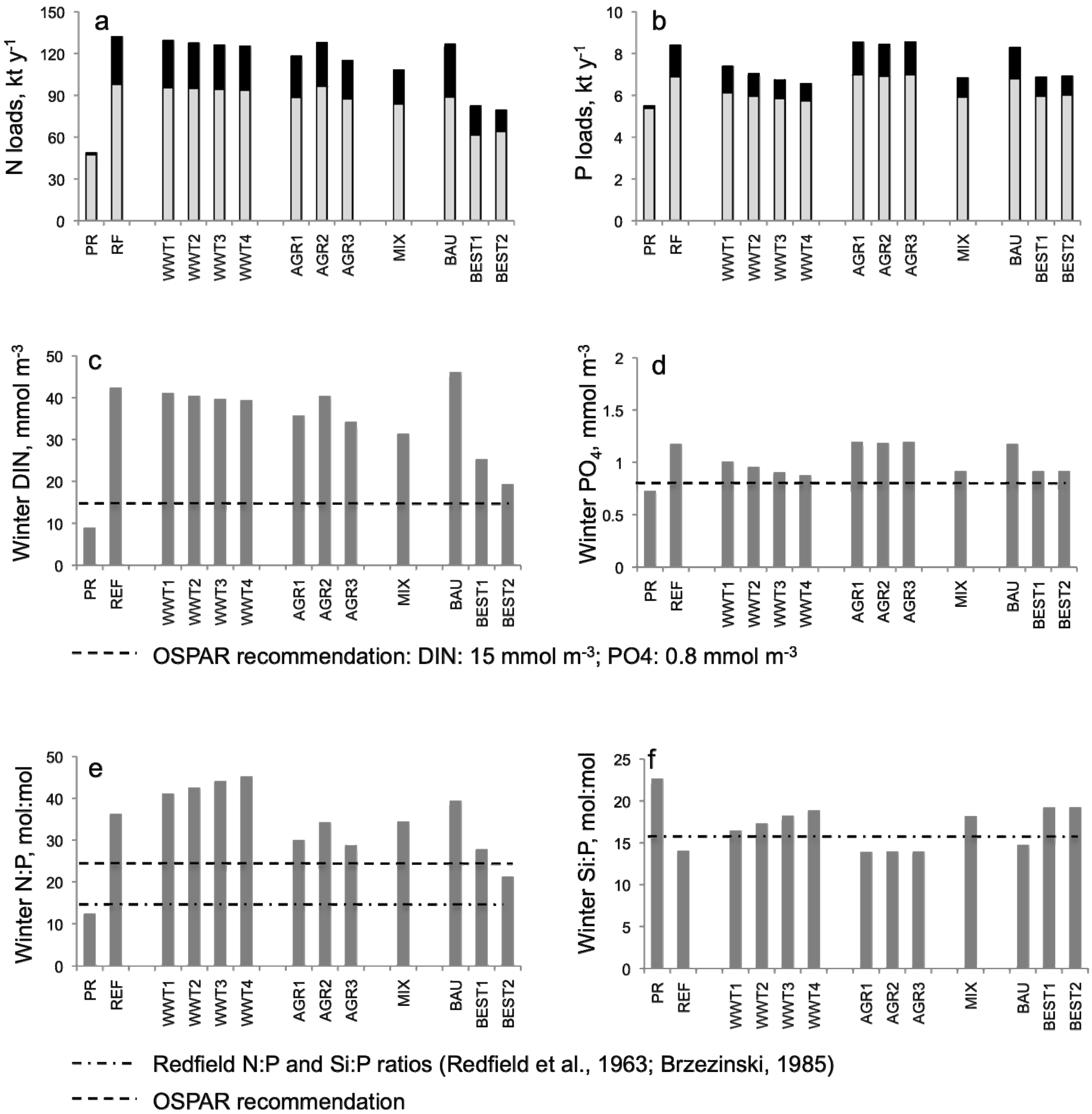


Figure 6-3 SR-MIRO simulations of N (a) and P (b) Atlantic (light grey block) and Scheldt (black block) loads to the BCZ; of winter DIN (c) and PO_4 (d) concentration in the BCZ and their N:P (e) and Si:P (f) ratios: reference and mitigation scenarios. PR: Pristine, RF: 2000, WWT1: WWTP>500,000 IE; WWT2: WWTP>100,000 IE; WWT3: WWTP>20,000 IE; WWT4: WWTP>0 IE; AGR1: NTWC; AGR2: Extensive farming; AGR3: AGR1+AGR2; MIX: AGR3+WWT3; BEST: full biological agriculture + WWT3 with NO_3 diffuse at 6 (BEST1) and 3 (BEST2) mg N L^{-1}

Under the present-day eutrophication (RF), the N and P loads to the BCZ are increased by respectively a factor 2.7 and 1.5 compared to the PR simulation but the relative contribution of direct and indirect nutrient sources is considerably modified (Fig. 6-3ab). According to SR-MIRO simulations, the Scheldt supplies today to the BCZ some 26 and 18 % of the total input of N and P respectively (Fig. 6-3ab). Nutrient reduction scenarios are modifying the total amount received by the BCZ without however changing significantly the relative contribution of the direct and indirect sources (Fig. 6-3ab), due to the implementation of identical nutrient reduction measures on the 3S watershed.

Basically the nutrient reductions simulated at the 3S outlets for the different scenarios are propagating to the BCZ while damped (Table 6-1, Fig. 6-3ab). Shortly, the reduction obtained after WWT upgrading increases with the treatment capacity of plants and shows a significant decrease of P loads to the BCZ (12-22%) while the relative decrease of N inputs remains insignificant (2-5%).

The conclusion is reverted when agro-environmental measures are applied: the reduced N fertilization in combination with the use of NTWC (AGR1) or in combination with extensive farming (AGR3) is decreasing N loads to the BCZ by 10-13% while P loads are slightly increased (2%). The solely extensive farming (AGR2) has little effect with a 3% drop of N loads and no visible effect on P loads (Fig. 6-3ab).

Altogether the most efficient short-term reduction of both N and P inputs to the BCZ is obtained when both tertiary WWT and agriculture measures are implemented (MIX, Fig. 6-3ab). Finally, while a major reduction of P input to the BCZ (>20%) can be achieved by upgrading WWT, a similar relative reduction of N is only obtained with the idealized BEST scenarios (BEST1:38%, BEST2:40%; Fig. 6-3a), thus requiring a drastic change of agricultural practices necessarily supported by new societal practices and environmental policy, e.g. provide incentive measures and regulations to avoid unsustainable agricultural development like biofuel production or further increase in meet consumption etc.

The modification of the BCZ nutrient status due to mitigation scenarios is better approached by comparing the nutrient concentrations simulated in winter, *i.e.* when biological activity is the lowest. Figures 6-3c and d show the winter DIN (Dissolved Inorganic Nitrogen: NO_3+NH_4) and PO_4 concentrations simulated for the reference and mitigation scenarios and compare them to the values of the OSPAR recommendation for a good ecological state of the BCZ (DIN≤15 mmol m-3, PO4≤0.8 mmole m-3; OSPAR, 2005). As expected, the SR-MIRO simulated winter DIN and PO_4 for the pristine reference (PR) are lower than the OSPAR recommendations, although close to (Fig. 6-3cd). All the tested mitigation scenarios lead to higher winter values for both DIN and PO_4 in comparison with the pristine reference (PR) and the OSPAR recommendation. WWT scenarios suggest that values close to OSPAR can be obtained for winter PO_4 after implementation of the EU WFD on the 3S watershed (WWT3; Fig. 6-3d). In contrast, SR-MIRO winter DIN concentrations are all significantly higher than the OSPAR recommended value, whatever the mitigation measure is (winter DIN>15 mmole m^{-3}; Fig. 6-3c). Even when implementing organic agriculture on the 3S basins, the resulting winter DIN in the BCZ i.e. 25 (BEST1) and 19 (BEST2) mmole m^{-3} is still above although closer to the recommended threshold (Fig. 6-3c). Furthermore, the reduction achieved is comparatively much significant than that reached with the MIX scenario.

Under these conditions, the present-day 'excess N compared to P' status of the BCZ (N:P=36; Fig. 6-3e) with respect to the biological needs (N:P=16; Redfield et al., 1963) is maintained along the different scenarios. The imbalance is exacerbated after upgrading of WWT solely (N:P>40; Fig. 6-3e), but relieved when agro-environmental measures are implemented only (N:P close to 30; Fig. 6-3e) or in combination with WWT (N:P=34; Fig. 6-3e).

Again the highest reduction of N obtained after implementation of organic agriculture gives the best nutrient status for the BCZ when both the quantity of N and P ratio are decreased resulting in N:P values close to the OSPAR recommendation (N:P=24) or even lower (N:P=21; BEST 2; Fig. 6-3e). Interestingly when no mitigation measure is taken by now, the SR-MIRO model predicts on the long term a 9% increase of the present-day DIN winter stock and an increase of the N:P ratio from 36 to 39 (BAU; Fig. 6-3e).

Most of human-related eutrophication problems are due to the proliferation of non-siliceous algae (here *Phaeocystis*) that benefit from the excess of N and P loads compared to the unaffected Si that controls diatom growth. Contrarily to *Phaeocystis* diatoms can be considered as desirable, transferring directly their production to copepods. Figure 6-3f then compares for each scenario the simulated winter Si:P with the Si:P requirements of coastal diatoms (Brzezinski, 1985) as an indicator of ecosystem integrity. Clearly, the pristine nutrient status of the BCZ shows a Si excess with respect to P and N while the present-day winter Si to P status is more balanced (Si:P=14; Fig. 6-3f) and is maintained on the long term (BAU scenario; Fig. 6-3f).

All AGR scenarios indicate a potential Si limitation but N excess (Fig. 6-3ef). On the contrary, all WWT scenarios including those in combination with good agricultural pratices (MIX) and organic farming (BEST) result in a winter excess of Si with respect to P (Fig. 6-3f).

6.3.4. *Phaeocystis* colony magnitude and duration

The resulting effect of nutrient reduction scenarios on *Phaeocystis* colony blooms has been appraised based on two indicators: the maximum amplitude reached by the colony bloom and its duration. The former is defined by the maximum cell density reached by the simulated colony bloom (Fig. 6-4a) while the latter corresponds to the number of days with *Phaeocystis* cell density higher than the 'natural' reference (Fig. 6-4b), i.e. 4 10^6 cell L^{-1} (Lancelot et al., 2009). Translated to *Phaeocystis* damage, this indicator gives the possible duration of foam events while the former may reflect the maximum foam density to be reached.

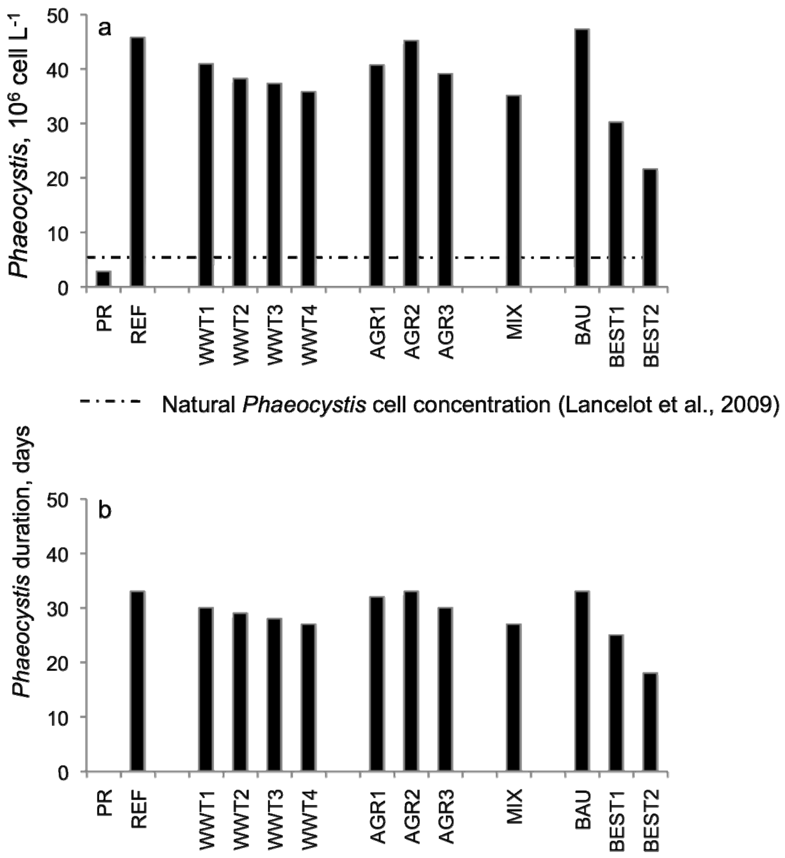


Figure 6-4 SR-MIRO simulations of *Phaeocystis* maximum cells (a) and duration (b) obtained for the reference and mitigation scenarios. PR: Pristine, RF: 2000, WWT1: WWTP>500,000 IE; WWT2: WWTP>100,000 IE; WWT3: WWTP>20,000 IE; WWT4: WWTP>0 IE; AGR1: NTWC; AGR2: Extensive farming; AGR3: AGR1+AGR2; MIX: AGR3+WWT3; BEST: full biological agriculture + WWT3 with NO_3 diffuse at 6 (BEST1) and 3 (BEST2) mg N L^{-1}

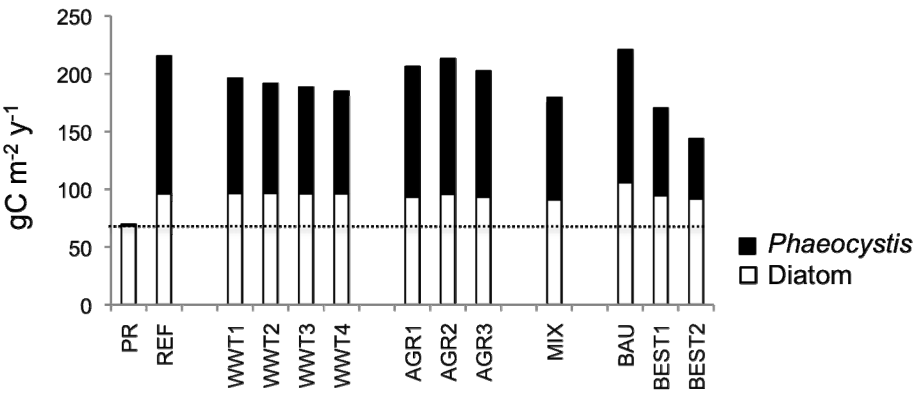


Figure 6-5 SR-MIRO simulations of diatom and *Phaeocystis* colony annual production. All obtained for the reference and mitigation scenarios. PR: Pristine, RF: 2000, WWT1: WWTP>500,000 IE; WWT2: WWTP>100,000 IE; WWT3: WWTP>20,000 IE; WWT4: WWTP>0 IE; AGR1: NTWC; AGR2: Extensive farming; AGR3: AGR1+AGR2; MIX: AGR3+WWT3; BEST: full biological agriculture + WWT3 with NO*3* diffuse at 6 (BEST1) and 3 (BEST2) mg N L^{-1}

A bit disappointing is the low reduction of *Phaeocystis* magnitude obtained for each short-term mitigation scenario (Fig. 6-4a). The highest reduction is achieved after WWT upgrading, i.e. 11 to 22% depending of the number of inhabitants connected to a tertiary wastewater treatment. This reduction is associated with a decrease of the bloom duration (9-18% i.e. 3-6 days with respect to the 33 days simulated for RF; Fig. 6-4b). In comparison a lower reduction of the *Phaeocystis* bloom magnitude and duration is predicted after implementation of the reduced N fertilization in combination with NTWC use only (11, 3%; Fig. 6-4a,b) or in combination with extensive farming (15, 9%; Fig. 6-4a,b). The solely extensive farming (AGR2) has little effect with a 1% drop of the *Phaeocystis* bloom magnitude and no effect on bloom duration (Fig. 6-4ab). It is only when agro-environmental measures are associated to WWT that the magnitude of the *Phaeocystis* colony bloom is substantially decreased (23 %; Fig. 6-4a) while still persisting for 33 days (Fig. 6-4b). Interestingly enough the highest reduction of both the *Phaeocystis* colony bloom magnitude (34-53%) and duration (1-2 weeks) is obtained on the long term after implementation of organic agriculture on the 3S watershed (BEST1 and BEST2, Fig. 6-4ab). On the contrary if no mitigation measure is taken, the BAU scenario predicts on the long-term a 3% increase of the *Phaeocystis* colony bloom magnitude and the persistence of its duration (Fig. 6-4ab), due to the increased DIN supply by groundwater that feeds the surface water discharge in summer. Indeed, due to their ability to grow under low PO_4 (Veldhuis and Admiraal, 1987), any supply of N under condition of low PO_4 is beneficial to *Phaeocystis* colonies (Lancelot et al., 2005). A drastic phosphorus reduction in the watersheds, already ongoing, is in any case required to prevent river eutrophication and associated problems in drinking water production (Garnier et al., 2005a). At the coastal zone, low phosphorus level may be compensated by rapid recycling which makes PO_4 available for the algae by (micro)biological processes (Currie and Kalff, 1984; Labry et al., 2005). This call for further studies to determine to which extent diatoms might compete with *Phaeocystis* in a N:P ratio >> 16, and non limiting Si.

6.3.5. BCZ ecosystem ‘health’

The concept of ecological health was proposed by Costanza and Mageau (1999) to scale the capability of an ecosystem to maintain its vigor and self-organization in response to stressors. Among the different ecosystem attributes used as indicators of ecosystem health or integrity (Nunneri et al., 2007) we choose to analyze changes in ‘vigor’ estimated by primary production (Fig. 6-5) and ‘diversity’ shown by the relative contribution of *Phaeocystis* (Fig. 6-6). As a general trend *Phaeocystis* colonies contribute with diatoms to more than 90% of the annual primary production that sustains biological resources in the BCZ. While diatom production is directly transferred to higher trophic levels through the grazing by copepods, most of the *Phaeocystis* colony production escapes direct grazing and hence decreases the trophic efficiency of the ecosystem (Lancelot et al., 2009).

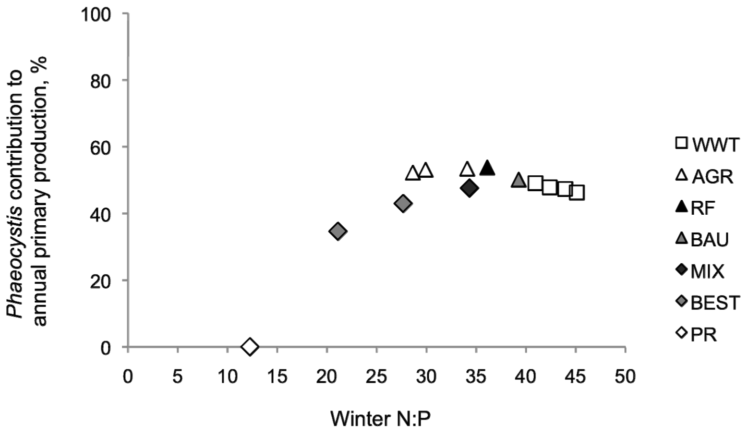
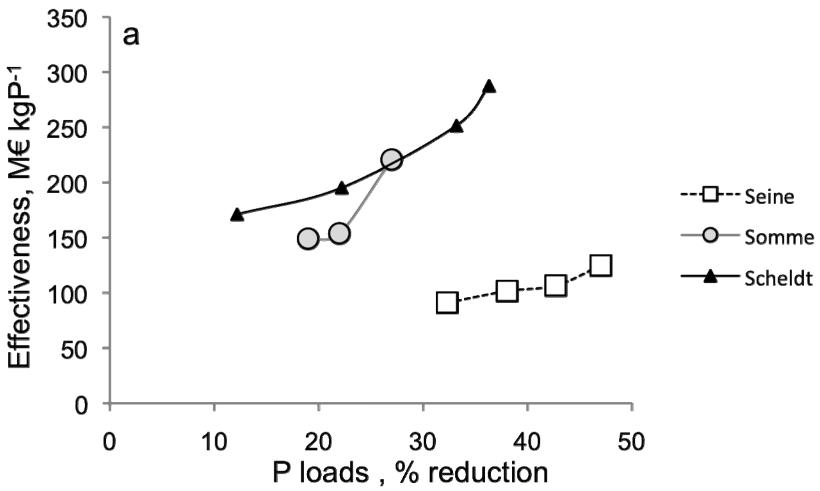


Figure 6-6 Relationship between the contribution of *Phaeocystis* colony production to annual primary production in the BCZ and the winter N:P ratios , all obtained from SR-MIRO simulations under the reference and mitigation scenarios.



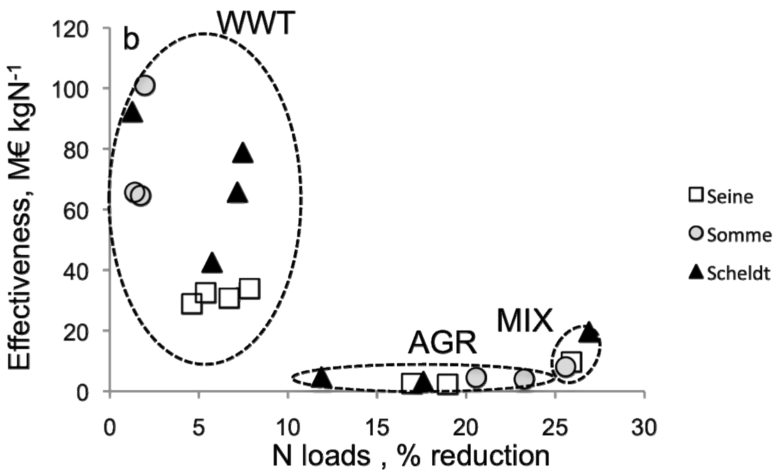


Figure 6-7 Relations ship between cost effectiveness of the nutrient mitigation measure and the achieved nutrient reduction for phosphorus (a) and nitrogen (b)

No significant *Phaeocystis* colony production is estimated for the pristine scenario (Fig. 6-5) when diatoms are producing 89% of the total annual primary production. The present-day supply of imbalanced anthropogenic nutrients (scenario RF) is increasing annual primary production by a factor of about 3 mostly due to the blooming of *Phaeocystis* colonies that now contribute to 54% of the annual primary production (RF, Fig. 6-6). Yet the present-day diatom production is also increased by 38% in comparison with the Si-excess pristine conditions (Fig. 6-5) and is to be related to increased winter stock of PO_4 when compared to PR (Fig. 6-3d). Contrasted results are obtained after the implementation of either WWT or agro-environmental scenarios.

The WWT scenarios are all predicting a substantial decrease of the annual primary production (8-14% depending on level of WWT implementation; Fig. 6-5), at the only expenses of *Phaeocystis* colonies whose annual production decreases by 16-26 % (Fig. 6-5), then contributing to less than 50% (46-49%) of the total annual production (Fig. 6-6). The well-balanced Si:P (Fig. 6-3f) after WWT upgrading explain the lack of impact on diatom production.

A small reduction of primary production is predicted by SR-MIRO when agro-environmental measures AGR1 and AGR3 are implemented (5%; Fig. 6-5) and affects equally the production of *Phaeocystis* colonies and of diatoms now limited by the Si availability (Fig. 6-3f). Insignificant change with respect to RF is predicted when the only extensive farming (AGR2) is implemented (1%; Fig. 6-5). The combination of WWT upgrading and agro-environmental measures MIX shows a significant decrease of both *Phaeocystis* and diatom production (Fig. 6-5). The simulated drop of *Phaeocystis* production is similar as that predicted by WWT3 (Fig. 6-5).

As a whole, the best mitigation of *Phaeocystis* production with the lowest impact on diatom production is obtained when both WWT improvement and biological agriculture are implemented on the long term (Fig. 6-5). Under such conditions the present-day *Phaeocystis* production is reduced by 20-32% and contributes to only 40-35% of the total annual primary production (Fig. 6-6).

Based on these results we conclude that not only the quantity but also the balance between N and P inputs are important in determining the diatom and *Phaeocystis* annual production. In agreement a positive relationship is observed between the SR-MIRO-simulated *Phaeocystis* contribution to annual production and the corresponding winter N:P status of the BCZ up to 28 (Fig. 6-6). Above this threshold, *Phaeocystis* colonies are contributing up to 50 % of the annual production.

6.3.6. Cost assessment and effectiveness

A large number of uncertainties are associated with the economic assessment of nutrient reduction scenarios, particularly for the WWT improvement, which is based on three criteria (age of the WWTP, capacity and existing wastewater treatment). The most accurate economic assessment would have been the attribution of the real cost of each wastewater treatment plant upgrading to tertiary treatment which is impossible due to the lack of available information. In spite of this, the estimated cost range values are comparable with those obtained by Atkins and Burdon (2006).

For the agro-environmental measures, we consider that farmers implement them in an optimized way. Therefore, the cost assessment based on several field studies and on the Farm Accountancy Data Network (FADN), gives minimum costs. However we consider that uncertainty cannot be that high, as farmers tend always to reach the economic optimum.

Table 6-2 reports for each basin, the estimated cost of the implementation of each nutrient reduction measure as well as the cost effectiveness i.e. the cost normalized to the obtained nutrient reduction at the 3S outlet with respect to the present-day discharge (RF scenario; Table 6-1). As a general trend, the cost of nutrient mitigation measures increases with the level of WWT implementation and the size and population of the basin but there is no direct relationship between the cost of the measure and its effectiveness (Table 6-2). For each river basin there is a positive but specific link between cost effectiveness and the percentage of P reduction obtained at the river outlet (Fig. 6-7a). This means that the cost for reducing one unit of P increases with the WWT capacity. Mainly for the Scheldt and Somme basins, the gain obtained by upgrading wastewater treatment facilities for IE <20,000 appears neither ecologically relevant nor cost-effective (Table 6-2). When comparing the three river basins, WWT improvements on the Seine watershed are more cost effective in reducing P inputs at its outlet than the Somme and Scheldt that show a similar trend (Fig. 6-7a). This might result from the fact that wastewater release is more concentrated in the Seine Basin (with 2/3 in Paris agglomeration) than in the other two basins.

Table 6-2: SR-MIRO simulations scenarios: total cost and effectiveness of N and P reduction at the 3S outlet

SR-MIRO SCENARIOS	Cost M €			Cost effectiveness €/kg N			€/kg P		
WWTP-Upgrading NP	Seine	Somme	Scheldt	Seine	Somme	Scheldt	Seine	Somme	Scheldt
WWT1 (>500,000 IE)	198	-	34	29	-	92	91	-	171
WWT2 (>100,000 IE)	260	7.5	70	33	66	43	102	149	195
WWT3 (>20000 IE)	307	9	135	31	65	66	107	154	251
WWT4 (>0 IE)	395	16	169	34	100	79	125	221	287
Ago-environmental	Seine	Somme	Scheldt	Seine	Somme	Scheldt			
AGR1	63	7.5	16	2.5	5	5			
AGR2	-	-	-	-	-	-			
AGR3	63	7.5	16	2.3	3	3			
MIX (WWT3 + AGR3)	370	16.5	151	10	8	20	127	259	287

The upgrade of WWTP to tertiary treatment and the agricultural scenario AGR1 are basic measures, as they respond to EU Directives signed in 1991. Therefore, these scenarios are currently in process. The AGR2 scenario belongs to the supplementary measures category. Its low cost-effectiveness ratio shows that WFD objectives could be reached in a sustainable manner. As AGR2 concerns cattle, it has a much wider effect in the Scheldt basin where breeding production is intensive (cattle density respectively 3 and 5 times higher than in the Somme and Seine watersheds) and is linked with the fodder corn crop. As intensive breeding leads to a high level of nitrogen surplus and therefore constitutes the major source of nutrients losses in agriculture (Choi et al., 2005; Lacroix et al., 2006), AGR2 is clearly relevant (Table 6-2).

In contrast, WWT mitigation measures are not cost-effective in terms of N reduction (30-100 € per kg N; Table 6-2, Fig. 6-7b). In comparison agro-environmental measures are much more cost-effective i.e. ≤ 5€ per eliminated kg N. Other economic studies (e.g. Wustenberghs et al., 2008) report similar conclusions about the higher cost-effectiveness of agricultural measures over improvement of wastewater treatment for nitrogen reduction. Incidentally the cost-effectiveness ratio of the AGR1 scenario in the 3S watershed is quite similar to other studies (Alard et al., 2002; Atkins and Burdon, 2006; Nolte, 2007).

The further comparison between three groups of scenarios, i.e. WWT improvements, AGR1 and AGR2, indicates that the more the measure is implemented upstream of the production or consumption cycle, the more it is efficient, and the more iterative costs are reduced. End-of-pipe measures are effective for P reduction but very costly.

Again the AGR scenarios appear the most technical and cost effective mitigations measures of N loads to the coastal sea (Table 6-2; Fig. 6-7b).

The cost for *Phaeocystis* colony mitigation in the BCZ includes the nutrient reduction cost of all the three basins and the cost-effectiveness is appraised in terms of achieved reduction of *Phaeocystis* maxima and duration (Table 6-3). Clearly, agro-environmental measures even in combination with WWT improvement (MIX) are suggested more cost-effective for reducing both the duration and the maxima reached by *Phaeocystis* bloom in the whole BCZ volume ($3\ 10^{10}\ m^3$). Even if the *Phaeocystis* reduction achieved after WWT improvements is significant and ecologically relevant the cost to attain a given reduction level is significantly higher in comparison with the cost of agro-environmental measures whatever the WWT improvement is (Table 6-3). For a similar 11% reduction of the *Phaeocystis* maxima, the application of good agricultural practices (AGR1) is 3 times more cost effective than the improvement of WWT1 (Table 6-3). Further the highest cost-effectiveness for reducing both *Phaeocystis* magnitude and duration is obtained by the combination of good agricultural practices with extensive farming (AGR3; Table 6-3). Surprisingly the cost for reducing *Phaeocystis* duration by one day is elevated for all nutrient reduction scenarios (77-90 €), excepted for the AGR3 scenario that predicts a 9 days reduction of *Phaeocystis* duration at a cost of 13 M€ per day (Table 6-3).

Finally the combination of WWT improvement and of agro-environmental measures (MIX, Table 6-3) appears the most cost-effective and ecologically-relevant when compared to the cost associated to WWT4 improvement needed to the maximum *Phaeocystis* reduction (WWT4; Table 6-3). Compared to WWT4 the MIX scenario allows reduction of N loads to the coastal sea at low cost (Table 6-2).

Table 6-3: SR-MIRO scenarios: total cost and effectiveness of *Phaecystis* maximum and duration reduction in the BCZ. RF *Phaecystis* maximum= 45.8 10^6 cell L^{-1}; RF 'undesirable' bloom duration: 33 days

SR-MIRO SCENARIOS	*Phaeocystis* colonies		Cost	Cost-effectiveness	
	Maximum 106 cell (% red.)	Duration Day, (% red.)	M €	M €/ 106 cell	M €/ day
WWTP-Upgrading NP					
WWT1 (>500000 IE)	41.9 (11)	30 (9)	232	47	77
WWT2 (>100000 IE)	38.2 (17)	29 (12)	338	44	84
WWT3 (>20000 IE)	37.3 (19)	28 (15)	451	53	90
WWT4 (>0 IE)	35.8 (22)	27 (18)	580	58	96
Ago-environmental					
AGR1	40.7 (11)	32 (3)	88	17	87
AGR3	39.1 (15)	30 (9)	88	13	29
MIX (WWT3 + AGR3)	35.1 (23)	27 (18)	538	50	89

6.4. Discussion

6.4.1. Sensitivity to hydro-climatic factors

NAO-driven hydro-climatic changes in the region have been found to affect the present-day magnitude of *Phaeocystis* blooms in the BCZ by determining the spreading of the Scheldt plume (Breton et al., 2006). In the SR-MIRO domain rainfall is driving the river discharge of freshwater and transport of nutrients while the wind forcing determines the residence time of river nutrients in the BCZ. Part of the uncertainty in the obtained *Phaeocystis* mitigations may thus be approached by comparing SR-MIRO simulations obtained with hydro-climatic conditions characteristics of a dry (1996) and wet (2001) year that respectively predict a lower and higher delivery of nutrients (nitrogen, phosphorus and silicon) at the outlet of the 3S watershed (Thieu et al., 2010) as well as a longer and shorter time residence of water in the BCZ (Gypens et al., 2007).

Table 6-4 compares for chosen short- and long-term nutrient reduction scenarios the winter nutrient stocks and magnitude and duration of *Phaeocystis* bloom obtained for dry and wet years. As a general trend, higher DIN (PO_4) winter concentrations are simulated during wet (dry) years (Table 6-4). A quick examination suggests a similar effect of meteorological conditions on coastal eutrophication assessment with wet years characterized by intense shorter blooms and dry years by *Phaeocystis* blooms of lower magnitude but longer duration except for WWT3 improvement scenario with no change in duration (Table 6-4). The bloom maximum is related to the DIN winter stock which increases during wet years while the duration relies on PO_4 availability, higher during dry years (Table 6-4). The lack of trend obtained for the WWT3 scenario has to be related to the insignificant difference between the dry and wet simulated winter stocks of PO_4 (Table 6-4). As a general trend no difference in bloom duration is predicted between dry and wet years after WWT improvement scenarios (not shown).

The percentage of *Phaeocystis* reduction achieved appears independent of meteorological conditions except for the solely WWT scenarios for which a slightly lower reduction is reached during dry (9%) in comparison with wet (13%) years (Table 6-4).

Absolute difference between dry and wet years depends on the chosen mitigation measures, being noticeably higher for agricultural (AGR3, MIX) than WWT3 scenarios (Table 6-4). On the long-term (BEST1, BAU; Table 6-4) the obtained *Phaeocystis* maxima are comparable while the simulated bloom duration is 4-5 days longer for dry meteorological conditions.

Table 6-4: Sensitivity analysis of RS-MIRO nutrient mitigation scenarios to hydro-climatic conditions

SR-MIRO scenarios	Winter DIN mmole m^{-3}		Winter PO_4 mmole m^{-3}		*Phaeocystis* max. 10^6 cells L^{-1}		*Phaeocystis* duration days	
	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet
REF	40.5	42.7	1.32	1.13	41.6	47.4	40	32
Short term								
WWT3	36.9	40.1	0.93	0.90	38	41.3	28	28
AGR3	33.9	34.3	1.34	1.15	35	40.3	33	30
MIX	30.2	31.6	0.93	0.90	33	38.5	28	27
Long term								
BAU	43.4	46.8	1.32	1.13	47	48	37	32
BEST1	25.2	25.2	0.93	0.90	28.9	29.6	26	22

Altogether this sensitivity analysis shows that meteorological conditions are determining the amplitude and duration of *Phaeocystis* blooms throughout their impact on the winter stock of PO_4 and DIN. An uncertainty of +/- 13% on the maximum of cells and of 3 days can be estimated from the departure between the simulated magnitude and duration of *Phaeocystis* bloom for dry and wet years and those obtained for year 2000 (mean meteorological year).

6.4.2. Relevance of the interdisciplinary approach

This first application of the SR-MIRO mathematical tool including a GIS-based software and two coupled deterministic models has proved to be of great value for testing the short-term ecological and cost effectiveness of future nutrient reduction measures aiming at mitigation of coastal eutrophication in general and of *Phaeocystis* colony blooms in particular.

Considering the BCZ actual eutrophication level established for year 2000, the implementation of the integrated modeling tool in the 3S watershed and adjacent Eastern Channel and Southern North Sea suggests that short-term measures such as the combination of WWT improvement for a plant capacity of 20,000 IE (WWT3) as recommended by the EU WFD and of good agricultural practices (integrated fertilization, use of NTWC and extensive farming; AGR3) here referred as MIX, allows to significantly decrease the winter stock of nitrogen and phosphorus in the BCZ. While the PO_4 threshold of 0.8 mmole m^{-3} recommended by OSPAR is achieved by these measures (Fig. 6-3d), the simulated DIN concentration (30 mmole m^{-3}; Fig. 6-3c) is still two times higher than the OSPAR recommendation of 15 mmole m^{-3} but lower than that obtained when only WWT improvement is considered (i.e. 40 mmole m^{-3}; Fig. 6-3c) highlighting the effectiveness of agro-environmental measures in reducing nitrogen loads. Further the application of the MIX measures allows the reduction of respectively 23 and 18% of the intensity and duration of undesirable *Phaeocystis* colonies in the coastal strip for an annual cost of 538 M€. At a first glance this cost might appear expensive but when considering the extent of the affected coastal zone and the inhabitants of the 3S watershed (about 70 millions), it represents less than 10 € per year and per inhabitant i.e. some 40 € per family. Moreover it has to be noted that the cost concept here developed refers only to eutrophication mitigation at the coastal zone. Subsequently other environmental benefits at the coastal zone but also within the drainage basin such as improved ground- and surface water quality are not included in this valuation.

The *Phaeocystis* mitigation cost analysis shows different cost-effective figures depending on the criterion used to scale the *Phaeocystis* improvement, i.e. the reduction of maximum or the duration. Both have different ecological and economic implications that needed to be further assessed.

In the framework of the EC-funded Thresholds project (www.thresholds-eu.org), an external cost assessment of *Phaeocystis* bloom impacts on recreational activities has been carried out (Longo et al., 2007). The willingness to pay to avoid the blooms in terms of both quantity and duration of the event has been assessed with a conjoint choice analysis. The results demonstrated a complex interrelation between duration, intensity of bloom and externalities. Latest results analysis shows willingness to pay 44 € per person per year for low-level bloom. This is broadly in line with a former study (23 - 41€ per person per year) that also considered health impacts (Stoltel et al., 2003). These externalities compare well with the costs of nutrients mitigation measures estimated in the present study (Table 6-3). It is important to keep in mind however that nutrient mitigation measures allow additional improvements particularly in terms of environmental status of inland surface water and groundwater.

Application of the SR-MIRO tool shows that the reduction obtained by implementing realistic environmental measures on the short-term, is however not sufficient for restoring well-balanced nutrient concentrations for fulfilling N:P:Si needs of coastal phytoplankton. An additional scenario testing the long-term effect of the replacement of conventional agriculture by biological one (organic agriculture) in the whole 3 S-watershed suggests that such a drastic change in agriculture practices, although maybe unrealistic, would be needed for significantly reducing the present-day excess of nitrogen in the coastal sea to level close to OSPAR recommendation (Fig. 6-3c). These long-term scenarios also show that if no nutrient mitigation action is taken, the eutrophication status of the coastal sea will worsen as more N and P will be delivered to the coastal sea (Fig. 6-3c,d) allowing more *Phaeocystis* colonies to bloom (Fig. 6-4).

Acknowledgements

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Conclusion générale et perspectives

Cette étude a permis de caractériser le fonctionnement biogéochimique des trois bassins versants : Seine, Somme, Escaut, d'évaluer la contribution des différentes sources de nutriments, et d'examiner la possibilité de restaurer des apports fluviaux équilibrés à la zone côtière.

Il a d'abord fallu proposer une description suffisante de l'espace couvert par les trois bassins versants. Pour la Seine, les travaux du programme PIREN ont fourni un cadre privilégié, où l'essentiel des données étaient disponibles. La juxtaposition des bassins Somme et Escaut, a conduit à résoudre le problème de l'homogénéité nécessaire à cette description, pour pouvoir non seulement comparer le fonctionnement des trois systèmes, mais également raisonner sur l'application de mesures prospectives transversales à l'échelle de cette espace de plus 100.000 km². Pour l'Escaut, il a fallu réunir des données segmentées entre la France, la Flandre, la Wallonie et la région de Bruxelles, afin d'obtenir une représentation continue de ce bassin. Ce travail a montré qu'il était possible d'aboutir à un portrait décrivant de façon similaire les caractéristiques naturelles (morphologie, hydrologie, usage du sol) et les contraintes anthropiques actuelles (pollutions domestiques, aménagements hydrologiques) s'appliquant aux trois systèmes.

Cette étude nous a conduits à quantifier les sources, le devenir et les transformations des nutriments, et à mettre en évidence les contrastes existant à cet égard entre ces trois systèmes : la Somme tournée exclusivement vers les cultures intensives, la Seine qui accueille une mégapole dans sa partie centrale, et l'Escaut nettement plus peuplé et majoritairement orienté vers des activités intensives d'élevages. La diversité des activités humaines sur les bassins se caractérise par des différences marquées en termes d'apports concentrés par les réseaux de collecte (sources ponctuelles) et ceux transférés par le ruissellement des eaux de pluie (sources diffuses) jusqu'aux grands axes des réseaux hydrographiques. Un bilan spatialisé et validé des transferts de nutriments a été dressé pour les trois hydrosystèmes. Pour cela, différentes approches de modélisation ont été comparées, et l'intérêt d'une représentation fine des processus proposée par le modèle Riverstrahler a été établi.

L'outil spatialisé Seneque-Riverstrahler a montré qu'il était désormais opérationnel pour faire le lien entre l'échelle microscopique des « processus » et celle de la prise de décisions, l'échelle des « bassins versants ». Ce modèle peut donc constituer un outil précieux pour la mise en place de politiques publiques nationales ou européennes puisqu'il constitue un support pour l'analyse prospective et la construction de scénarios. Ceux élaborés et implémentés à l'échelle du continuum Seine-Somme-Escaut - Mer du Nord, nous ont amenés à nous interroger sur l'impact des niveaux d'émissions fluviales vis-à-vis des problèmes d'eutrophisation côtière, mais également sur la capacité des futures réglementations à réguler les émissions de N et P dans les bassins.

En ce qui concerne l'enrichissement des milieux côtiers, il ressort de cette approche de modélisation prospective un bilan assez pessimiste, principalement lié à la persistance de la contamination nitrique. Alors que les niveaux de phosphore sont aujourd'hui considérablement réduits, les émissions d'azote ne font au mieux que se stabiliser depuis quelques années. L'amélioration conjointe des rejets ponctuels azotés et phosphatés, telle qu'elle est prévue, ne permettra pas réduire ce décalage, accentuant dans certains cas le déséquilibre de ces apports (ratio N:P). D'autre part, il apparaît clairement que les efforts actuellement consentis pour améliorer le traitement de l'azote en station d'épuration ne peuvent pas diminuer significativement les flux émis à la zone côtière.

L'analyse de la qualité des eaux dans les réseaux hydrographiques vient compléter ces premières conclusions. Nous avons montré que l'amélioration du traitement du phosphore en station reste nécessaire en aval des grandes agglomérations. Pour l'azote, l'amélioration des rejets après épuration n'a qu'un impact faible sur les quantités totales émises à la zone côtière. Cependant cet impact peut être localement significatif pour la vie des écosystèmes. A titre d'exemple, les travaux réalisés dans le cadre des programmes PIREN-Seine (www.sisyphe.upmc.fr/piren/) et Seine-Aval (http://seine-aval.crihan.fr) sur la nitrification de l'ammonium en stations d'épuration, ont clairement montré les améliorations des niveaux d'oxygénation dans l'estuaire de la Seine.

A l'échelle d'un continuum « rivière – zone côtière », ces apports d'azote sont fortement liés à l'importance des écoulements (lessivage des sols) et les quantités totales exportées à la zone côtière varient entre 1382-2316 kgN/km²/an pour la Seine ; 600-1816 kgN/km²/an pour la Somme et 1099-1813 kgN/km²/an pour l'Escaut, selon les conditions hydrologiques (1996-2001). L'évaluation des scéanrios 'amélioration STEP', 'Pâturage extensif', 'CIPAN' ainsi que la combinaison de ces trois mesures permettre de réduire les quantités d'azote exportées respectivement de 155-138 kgN/km²/an, 26-56 kgN/km²/an, 190-400 kgN/km²/an, 375-590 kgN/km²/an pour la Seine ; 800-553 kgN/km²/an, 798-567 kgN/km²/an, 852-798 kgN/km²/an, 883-960 kgN/km²/an pour la Somme ; et 632-349 kgN/km²/an, 368-660 kgN/km²/an, 396-746 kgN/km²/an, 549-1020 kgN/km²/an pour l'Escaut. D'après ces simulations, la sensibilité aux conditions hydrologiques est estimée entre 65 et 200%, alors que le scénario optimal (combinaison des trois mesures) permet une réduction des flux d'azote n'excédant pas 17 à 39% par rapport à la situation de référence et pour une hydrologie constante. La prise en compte de la variabilité hydrologique de chaque système est ainsi essentielle pour pouvoir apprécier le potentiel des différentes mesures testées. L'ensemble de ces résultats s'accorde sur la nécessité d'un meilleur contrôle des apports diffus d'azote. Notre collaboration avec des équipes d'agronomes et de socio-économistes, a montré que la maîtrise de cette contamination reste difficile à mettre en œuvre, mais n'est pas la plus coûteuse et demeure en revanche la plus efficace. A ce stade, la question est donc de savoir si les politiques sont assez « incitatives » pour espérer des changements significatifs des pratiques agricoles.

Dans cette perspectives, nous avons évalué de façon théorique la possibilité de réguler ces usages en essayant de maintenir une fermeture plus locale des cycles des éléments N, P et Si, remettant en cause l'ouverture croissante des cycles de le matière et des marchés qui a prévalu dans la politique agricole commune des 30-40 dernières années. L'analyse de l'évolution de ces cycles, en lien avec la modernisation de la chaîne agro-alimentaire, est riche d'enseignement et l'agriculture biologique apparaît alors comme une mesure capable de rétablir un fonctionnement biogéochimique plus « équilibré » des territoires et des hydrosystèmes. Toutefois, l'impact de ce mode de culture sur les niveaux de contamination nitrique est encore relativement mal documenté, et notre scénario sur l'agriculture biologique, bien qu'ouvrant des perspectives salutaires, ne représente qu'une approche théorique par rapport aux autres mesures agro-environnementales qui ont été explorées de façon plus réaliste.

Seneque-Riverstrahler a en effet été conçu pour décrire de façon fine l'ensemble des processus se déroulant dans la colonne d'eau et son interface avec le sédiment, mais ceux intervenant dans les parties terrestres et riveraines du bassin versant sont intégrés aux apports diffus qui constituent une condition limite amont. Ces apports doivent donc être documentés par des études expérimentales ou par des sorties de modèles agronomiques.

Pour faire progresser l'approche Seneque-Riverstrahler, une perspective à ce travail, serait de tendre vers une modélisation complète du fonctionnement biogéochimique des bassins versant, au-delà des hydrosystèmes, en reculant plus à l'amont encore la définition des conditions limites. Il serait alors possible d'obtenir une représentation complète des transferts entre les cultures, le sol et les eaux superficielles et phréatiques. Les échanges avec l'atmosphère de gaz à effet de serre comme le N_2O issus de l'agriculture, pourraient également être quantifiés. Des modèles agronomiques tels que STICS (Brisson et al., 2003) ou DNDC (DeNitrification-DeComposition) (Li, 2000; Li et al., 1992; 1994) sont capables de simuler les processus de nitrification, dénitrification et décomposition de la matière organique dans le sol, en relation avec les transferts hydriques et la croissance végétale. La résolution spatiale de ces modèles étant différente (échelle de la parcelle agricole *vs.* ordres de rivière ou distances kilométriques des grands axes), il sera nécessaire de s'interroger sur la possibilité (et la pertinence) de généraliser ces résultats pour les rendre compatibles avec l'approche de modélisation Riverstrahler. Cette ambition est toutefois d'actualité, et les travaux de Ducharne et al., (2007) constituent une étape pour la Seine, avec un premier couplage off-line des modèles Riverstrahler et STICS-MODCOU. La généralisation de cette chaîne de modélisation aux bassins de la Somme et de l'Escaut, qui n'était pas envisageable dans le cadre de cette thèse, reste une perspective de développement, de même que l'intégration d'un module simplifié de fonctionnement des sols, complètement intégré à Riverstrahler, dans le cadre de l'applicatif spatialisé Seneque. Sur ce dernier point, le développement de l'applicatif AIPreshume (Atlas Interactif des Pressions Humaines, Silvestre and Billen, 2007) peut être considéré comme une première étape.

Depuis sa première publication (Billen et al., 1994), le modèle Riverstrahler a vu sa description du fonctionnement biogéochimique des hydrosystèmes évoluer considérablement, en s'appuyant notamment sur des travaux de thèse dédiés à l'analyse de certains processus. On citera à titre d'exemple : le rôle des bactéries nitrifiantes (Cébron, 2004) et dénitrifiantes (Mounier, 2008), le devenir du phosphore (Némery, 2003), et de la Silice (Sferratore, 2006) dans les réseaux hydrographiques. Mon travail de thèse, bien qu'il ne s'attache pas à la modélisation exclusive d'un élément ou d'un processus, s'inscrit dans le prolongement logique de ces travaux, en intégrant l'ensemble de ces acquis dans le cadre d'une démarche de modélisation transversale et appliquée de façon concrète à un espace hautement contrasté.

Cette volonté d' « éprouver » la robustesse du modèle Riverstrahler m'a également conduit à considérer d'autres approches de modélisation, souvent basées sur le calage de paramètres décrivant le fonctionnement écologique de manière plus simplifiée, mais intégrant, au contraire de Riverstrahler, l'ensemble du système bassin versant. Des approches de modélisation plus statistiques, inspirées des modèles *Sparrow* ou *NEWS*-DIN ; ont ainsi été mises en œuvre. De tels modèles restent les seuls à proposer des analyses à l'échelle globale et à pouvoir ainsi intégrer l'influence des dynamiques mondiales, qu'elles soient démographiques, économiques ou politiques. Le travail réalisé en collaboration avec le groupe Global*NEWS*, dans le cadre de l'application des scénarios du Millenium Ecosystem Assessment, a permis d'établir la compatibilité d'une description globale des émissions de nutriment dans les bassins versant avec une modélisation détaillée des processus opérant dans les continuums aquatiques.

Dans le même temps, la constitution de base de données globales, de plus en plus précises et spatialisées, est ouverte à la communauté scientifique. On citera à titre d'exemple : l'accès depuis 2008 à une carte mondiale d'occupation du sol avec une résolution de 300 mètres, réalisée dans le cadre du projet Globcover de l'European Spatial Agency, ou encore, la base de données hydrologiques HydroSHEDS en cours de finalisation par le programme de conservation du WWF (World Wildlife Fund). Dans ce contexte, une perspective à plus long terme pourrait être d'adapter le modèle Riverstrahler à ces bases d'informations descriptives globales, ce qui pourrait permettre de proposer une modélisation déterministe des transferts des nutriments à l'échelle mondiale.

Le travail qui s'achève ici ouvre donc la voie à de très nombreuses applications et à de très riches développements.

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Annexe

Annexe 1 : Cinétiques et paramètres des processus décrits par le modèle RIVE

Process	kinetic expression	Parameters				
phytoplankton dynamics			**Meaning**	**Diatoms**	**Chloro-phyc.**	**Units**
Photosynthesis (phot)	kmax (1-exp-(α I/kmax)) PHY	kmax*	maximal rate of photosynth.	0.2	0.25	h^{-1}
		α	initial slope of P/I curve	0.0012	0.0012	h^{-1} $(\mu E.m^{-2} s^{-1})^{-1}$
reserves synthesis	srmax *M*(S/PHY,Ks) PHY	srmax*	max. rate of reserve synthesis	0.1	0.25	h^{-1}
		Ks	1/2 saturation cst	0.06	0.06	
reserves catabolism	kcr R	kcr.*	rate of R catabolism	0.2	0.2	h^{-1}
growth (phygrwth)	mufmax *M*(S/PHY,Ks) lf PHY	mufmax	max. growth rate*	0.05	0.09	h^{-1}
nutrient limitation factor	with lf = *M*(PO_4,Kpp) or *M*(NO_3 +NH_4, Kpn) *or M*($Si0_2$, KpSi)	Kpp Kpn KpSi	1/2 sat. cst for P uptake 1/2 sat. cst for N uptake 1/2 sat. cst for Si uptake	15 14 0.53	62 14 -	$\mu g\ P\ l^{-1}$ $\mu g\ N\ l^{-1}$ $mgSiO_2\ l^{-1}$
respiration	maint PHY +ecbs phygrwth	maint* ecbs	maintenance coefficient. energetic cost of biosynthesis	0.002 0.5	0.002 0.5	h^{-1} -
excretion (phyex)	exp phot.+ exb PHY	exp exb	"income tax" excretion "property tax" excretion	0.0006 0.001	0.0006 0.001	h^{-1} h^{-1}
lysis (phylys)	kdf + kdf (1+ vf)	kdf* vf +	mortality rate parasitic lysis factor	0.004 0 / 20	0.004 0 / 20	h^{-1} -
phyto sedimentation	(vsphy/depth).PHY	vsphy	sinking rate	.004	.0005	$m\ h^{-1}$
NH_4 uptake	phygrwth /cn NH_4/(NH_4+NO_3)	cn	algal C:N ratio	7	7	$g\ C(g\ N)^{-1}$
NO_3 uptake	phygrwth /cn NO_3/(NH_4+NO_3)					
PO_4 uptake	phygrwth /cp	cp	algal C:P ratio	40	40	$g\ C(g\ P)^{-1}$
$Si0_2$ uptake	phygrwth /cSi	cSi	algal C:Si ratio	2	-	$g\ C(g\ SiO_2)^{-1}$
temperature dependency	p(T) = p(Topt).exp(-(T-Topt)² / dti²)	Topt dti	optimal temperature range of temperature	21 13	37 17	°C °C
zooplankton dynamics				**Total zooplankton.**		
ZOO growth (zoogwth)	μzox.*M*(PHY-PHYo),KPHY*)*.ZOO	μzox KPHY PHYo	max. growth rate 1/2 sat cst to PHY threshold phyto conc.	0.02* 0.4 0.1		h^{-1} $mgC\ l^{-1}$ $mgC\ l^{-1}$
ZOO grazing	grmx.*M((*PHY-PHYo) KPHY*)*.ZOO	grmx	max grazing rate	0.055*		h^{-1}
ZOO mortality	kdz.ZOO	kdz	mortality rate	0.003*		h^{-1}
temperature dependency	p(T) = p(Topt).exp(-(T-Topt)² / dti²)	Topt dti	optimal temperature range of temperature	25 10		°C °C
Lamellibranchs				**Dreissena.**		
Filtration rate		fmax	max filtration rate	0.01*		$m^3\ gDW^{-1}h^{-1}$
temperature dependency	p(T) = p(Topt).exp(-(T-Topt)² / dti²)	Topt dti	optimal temperature range of temperature	25 8		°C °C
bacterioplankton dynamics				**small bac**	**large bac**	
HPi production by lysis	εpi . (phylys+bactlys+zoomort)	εp1 εp2 εp3	HP1 fraction in lysis pducts HP2 fraction in lysis pdcts HP3 fraction in lysis pdcts	0.2 0.2 0.1		- -
enzym. HPi hydrolysis	kib.HPi	k1b k2b	HP1 lysis rate HP2 lysis rate	0.005 0.00025		h^{-1} h^{-1}
HPi sedimentation	(vsm/depth).Hip	Vs	Hip sinking rate	0.05		$m\ h^{-1}$
Hid production by lysis	δe . (phylys+bactlys+zoomort)	εd1 εd2 εd3	HD1 fraction in lysis pdcts HD2 fraction in lysis pdcts HD3 fraction in lysis pdcts	0.2 0.2 0.1		- - -
enzym. HDi hydrolysis	eimax. *M(*HDi,KHi*)*.BAC	e1max e2max KH1 KH2	max. rate of HD1 hydrolysis max. rate of HD2 hydrolysis 1/2 sat cst for HD1 hydrol. 1/2 sat cst for HD1 hydrol.	0.75 0.25 0.25 2.5	0.75 0.25 0.25 2.5	h^{-1} h^{-1} $mgC\ l^{-1}$ $mgC\ l^{-1}$
direct substr. uptake	bmax. *M(*S,Ks*)*.BAC	bmax Ks	max. S uptake rate 1/2 sat cst for S uptake	0.16 0.1	0.6 0.1	h^{-1} $mgC\ l^{-1}$
bact. growth (bgwth)	Y. bmax. *M(*S,Ks*)*.BAC	Y	growth yield	0.25	0.25	-
bact. mortality (bactlys)	kdb.BAC	kdb	bact. lysis rate	0.02	0.05	h^{-1}
bact. sedimentation	(vsb/depth).BAC	vsb	bacteria sinking rate	0	0.02	$m\ h^{-1}$
ammonification	(1-Y)/Y.bgwth/cn	cn	bact. C:N ratio	7		$gC\ (gN)^{-1}$
PO4 production	(1-Y)/Y.bgwth/cp	cp	bact. C:P ratio	40		$gC\ (gP)^{-1}$
temperature dependency	p(T) = p(Topt).exp(-(T-Topt)² / dti²)	Topt dti	optimal temperature range of temperature	20 17	22 12	°C °C

Annexe 1 (suite) : Cinétiques et paramètres des processus décrits par le modèle RIVE

Process	**kinetic expression**	**Parameters**			
nitrification and phosphorus dynamics			**Meaning**	**nitrifying bacteria**	**Units**
NIT growth (nitgwth)	µnix.*M(*NH_4,KNH_4*).M(*O_2,KO_2*)*. NIT	µnix* KNH4 KO2	max growth rate of NIT 1/2 sat cst for NH4 1/2 sat cst for O2	0.046 1.12 0.6	h^{-1} mgN l^{-1} mgO_2 l^{-1}
NH4 oxidation	nitgwth/rdtnit	rdtnit	NIT growth yield NIT	0.1	mgC (mg NH_4)$^{-1}$
NIT mortality	kdnit.NIT	kdnit*	NIT mortality rate	0.01	h^{-1}
PO4 adsorpt/desorpt. (planktonic phase)	Langmuir isotherm	Pac KPads	SM max. adsorpt. capacity 1/2 saturation ads. cst.	0.0056 0.67	mgP (mgMES)$^{-1}$ mgP l^{-1}
temperature dependency	p(T) = p(Topt).exp(-(T-Topt)² / dti²)	Topt dti	optimal temperature range of temperature	23 15	°C °C
benthos recycling					
susp. matter sedim.	(vsm/depth)*MES	vsm	sinking rate		m^{-1}
Diffusion (interstitial ph.)	Fick law	Di	app. diffusion coefficient	8 10^{-5}	cm² s^{-1}
Mixing (solid phase)	Fick law	Ds	mixing coefficient	2 10^{-5}	cm² s^{-1}
orgN mineralis. (maorg)	kib.HPi/cn				
orgP mineralis.	kip.HPi/cp	k1p* k2p*	orgP hydrolysis rate of HP1 orgP hydrolysis rate of HP2	0.05* 0.0025*	h^{-1} h^{-1}
benth. nitrification	kNi*NH4 *(in oxic layer)*	kNi	1st order nitrification cst	0.1	h^{-1}
NH_4 adsorpt/desorpt.	1st order equilibrium	Kam	1st order adsorpt. cst for NH4	9	-
PO_4 adsorpt/desorpt. (in benthos)	1st order equilibrium	Kpa Kpe	PO4 adsorpt. (oxic layer) PO4 adsorpt. (anoxic layer)	100 0	- -
SiO_2 redissolution	kdbSi.SIB	kdbSi	silica redissolution rate	0.0001	h^{-1}
temperature dependency	p(T) = p(Topt).exp(-(T-Topt)² / dti²)	Topt dti	optimal temperature range of temperature	30 15	°C °C

*These parameters depend on temperature according to the relation mentioned.

+*M*(C,Kc) = C/(C+Kc) : hyperbolic Michaelis-Menten function

+vf: parasitic lysis amplification function. It is maintained at zero while algal density of each group remains lower than a threshold value of 65 µg Chl a l^{-1} and temperature is below 15°C.