

Table des matières

Introduction générale	1
Contexte	2
Le projet ABAC	3
Choix méthodologiques	3
Objectifs et organisation des travaux	4
 Partie I. Cadre général de l'étude	 6
Chapitre 1. Etat de l'art	7
1.1. La cascade de l'azote	8
1.2. Les systèmes biologiques en Europe	9
1.2.1. Cahier des charges de l'agriculture biologique	10
1.2.2. Les performances agro-environnementales	10
1.3. La lixiviation des nitrates dans l'hydrosystème	12
1.3.1. Les mécanismes de la lixiviation	12
1.3.2. Les facteurs de régulation	13
1.3.2.1. Les facteurs environnementaux	13
1.3.2.2. Les pratiques agricoles	13
1.4. Les émissions d'oxyde nitreux dans l'atmosphère	16
1.4.1. Les processus responsables des émissions	16
1.4.1.1. La nitrification	16
1.4.1.2. La dénitrification	16
1.4.1.3. La nitrification dénitrifiante	17
1.4.2. Les facteurs de régulation	17
1.4.2.1. Les teneurs en oxygène	17
1.4.2.2. Les apports d'azote et de carbone	17
1.4.2.3. La température et le pH	17
1.5. Les techniques de mesures des fuites d'azote	18
1.5.1. Les méthodes directes	18
1.5.1.1. La lixiviation du nitrate	18
1.5.1.2. Les émissions de protoxyde d'azote	21
1.5.2. Les méthodes indirectes	22
1.5.2.1. Les bilans azotés	22
1.5.2.2. Les facteurs d'émission	23
1.5.2.3. Les méthodes isotopiques (¹⁵ N)	23
1.5.2.4. Les modèles de prédictions des fuites azotées	24
Chapitre 2. Localisation des sites, matériels et méthodes	25
2.1. Localisation et caractéristiques des sites d'études	26
2.1.1. Le bassin de la Seine	26
2.1.1.1. Le contexte climatique	26
2.1.1.2. Un bassin majoritairement céréalier	28
2.1.1.3. Les systèmes biologiques en Ile-de-France	29
2.1.2. Le réseau ABAC	31
2.1.2.1. Localisation des exploitations	31
2.1.2.2. Les systèmes de cultures du réseau ABAC	32
2.1.3. Le bassin de l'Orgeval	33
2.1.3.1. Contextes pédologique et climatique	33
2.1.3.2. Les pratiques agricoles	34
2.2. Matériels et méthode	35
2.2.1. Les propriétés physico-chimiques des sols	35

2.2.2.	La lixiviation du nitrate	35
2.2.2.1.	Protocole des bougies poreuses	35
2.2.2.2.	Calcul de la lixiviation par un modèle hydrique	37
2.2.3.	Les émissions de protoxyde d'azote	37
2.2.3.1.	Protocole des chambres d'accumulation de gaz	37
2.2.3.2.	Expériences en conditions contrôlées	38
2.2.3.3.	Analyse de la phase gazeuse (N ₂ O)	40
Partie II.	Les fuites d'azote à différentes échelles spatiales et temporelles	41
Chapitre 3.	Nitrous oxide emissions and nitrate leaching in a mixed farm	43
3.1.	Abstract	43
3.2.	Introduction	44
3.3.	Materials and methods	45
3.3.1.	Experimental site and design	45
3.3.2.	Soil measurements	49
3.3.3.	Soil solution measurement	49
3.3.4.	Nitrous oxide emissions	50
3.3.5.	Calculations	50
3.3.5.1.	Statistical analysis	50
3.3.5.2.	Percolating water flux	51
3.3.5.3.	Biological nitrogen fixation	51
3.4.	Results	51
3.4.1.	Climate and hydrologic conditions	51
3.4.2.	N ₂ O emissions with manual and continuous sampling	52
3.4.2.1.	Method chambers comparison	52
3.4.2.2.	Factors influencing N ₂ O emission	54
3.4.2.3.	Cumulative N ₂ O fluxes for crops in OF and CF rotations	55
3.4.3.	N leached in organic and conventional systems	56
3.4.3.1.	Agricultural practices impacts	56
3.4.3.2.	Leaching and water inflow	56
3.4.4.	N losses in OF and CF systems	57
3.5.	Discussion	58
3.5.1.	Comparison of method chambers	58
3.5.2.	Mineral and organic fertilisation	58
3.5.3.	Catch crops and legumes	59
3.5.4.	Nitrogen balance over the rotations	59
3.6.	Conclusion	60
Chapitre 4.	Temperature dependence of nitrous oxide production in batchs experiments	61
4.1.	Abstract	61
4.2.	Introduction	62
4.3.	Materials and methods	62
4.3.1.	Study site	63
4.3.2.	Batch experiments: soil denitrifying and nitrifying potential	63
4.3.2.1.	Range of temperatures chosen and soil sampling	63
4.3.2.2.	Incubation conditions	64
4.3.2.3.	Denitrification	64
4.3.2.4.	Nitrification	64
4.3.3.	Chemical analysis	65
4.3.3.1.	Soil properties	65
4.3.3.2.	Dissolved inorganic nitrogen	65
4.3.3.3.	N ₂ O emissions	65
4.3.4.	Calculations	66
4.3.4.1.	Potentials related to temperature variations	66
4.3.4.2.	Determination of optimum temperatures	67

4.4.	Results	67
4.4.1.	Initial soil chemical properties and potentials determinations	67
4.4.2.	Seasonal variation of N ₂ O production at 20°C	68
4.4.3.	Effects of temperatures on N ₂ O production from nitrification and denitrification	69
4.5.	Discussion	72
4.5.1.	Role of experimental conditions on nitrification and denitrification potentials	72
4.5.2.	Influence of temperature on nitrite production and nitrous oxide emissions	73
4.6.	Conclusion	74
Chapitre 5.	Curative vs. preventive management of nitrogen transfers in rural areas	75
5.1.	Abstract	75
5.2.	Introduction	76
5.3.	Site studied and methods	77
5.3.1.	Characteristics of the Orgeval watershed	77
5.3.2.	Sampling and field studies, lab experiments and chemical analysis	78
5.3.3.	Simulating N reduction measures	79
5.4.	Quantifying the N cascade through the Orgeval watershed	80
5.4.1.	N leaching from agricultural soils to sub-root water, tile-drains and aquifers	80
5.4.2.	Denitrification and N ₂ O emissions in soils along a cropped slope	81
5.4.3.	In-stream N elimination processes	84
5.4.4.	A synthetic budget of N transfers in the Orgeval watershed	84
5.5.	Curative management measures to reduce NO ₃ ⁻ contamination	86
5.5.1.	NO ₃ ⁻ and N ₂ O concentrations in an artificial pond	86
5.5.2.	Simulation of the effect of pond creation at the scale of the Orgeval watershed	87
5.6.	Preventive management measures to reduce nitrogen contamination	88
5.6.1.	Good Agricultural Practices	88
5.6.2.	Modelling NO ₃ ⁻ contamination resulting from GAP and organic farming	89
5.7.	Discussion and conclusions	90
Partie III.	La lixiviation de l'azote dans des exploitations du bassin de la Seine	93
Chapitre 6.	Nitrate leaching from organic and conventional arable crop farms	95
6.1.	Abstract	95
6.2.	Introduction	96
6.3.	Materials and methods	97
6.3.1.	Localisation and characteristics of the main areas	97
6.3.2.	Agricultural practices studied	100
6.3.3.	Field measurements	100
6.3.3.1.	Ceramic cups	100
6.3.3.2.	Soil samples	100
6.3.4.	Analyses	100
6.3.4.1.	Soil analysis	101
6.3.4.2.	Soil solution analysis	101
6.3.5.	Calculations	101
6.3.5.1.	Percolating water flow	101
6.3.5.2.	N inputs: fertilisers and biological nitrogen fixation	102
6.3.5.3.	Statistical analysis	102
6.4.	Results	102
6.4.1.	Soil properties in OF and CF	102
6.4.1.1.	Textures	102
6.4.1.2.	Nutrients	102

6.4.2	Soil solution concentrations for the typical OF and CF crop successions	103
6.4.2.1.	Variations in organic rotations	104
6.4.2.2.	Variations in conventional rotations	105
6.4.2.3.	Impact of N inputs on organic crops	105
6.4.3.	Influence of crop management, soil and climate on nitrate losses	106
6.4.3.1.	Relation between N inputs and soil solution concentrations	107
6.4.3.2.	Hydrological conditions and leaching	107
6.4.3.3.	Relation between N total inputs, harvest and N leaching	107
6.5.	Discussion	108
6.5.1.	Experimental design and advantages of studying commercial farms	108
6.5.2.	Effect of climate and soil on leaching variability	108
6.5.3.	Key management practices controlling N leaching	109
6.5.4.	The importance of catch crops	109
6.5.5.	Room for improvement in conventional and organic systems	110
6.6.	Conclusion	110
Chapitre 7.	A large variability of leaching in organic and conventional systems	111
7.1.	Abstract	111
7.2.	Introduction	112
7.3.	Materials and methods	112
7.3.1.	Localisation and characteristics of the main areas	112
7.3.2.	Description of the organic and conventional systems	114
7.3.3.	Field measurements	115
7.3.3.1.	Ceramic cups	115
7.3.3.2.	Soils samples	115
7.3.4.	Analyses	115
7.3.4.1.	Soils analysis	115
7.3.4.2.	Water analysis	115
7.3.5.	Percolating water flux	116
7.3.6.	Statistical analysis	116
7.4.	Results	117
7.4.1.	Climate and soil conditions	117
7.4.2.	Variations of sub-root concentrations	117
7.4.2.1.	Variations according to crops	118
7.4.2.2.	The role of catch-crops	121
7.4.2.3.	Influence of fall fertilisation	122
7.4.2.4.	Variations among CF and OF systems	122
7.5.	N Leached in conventional and organic systems	123
7.6.	Discussion	124
7.6.1.	Farmers strategies	124
7.6.2.	Representativeness of the network	125
7.6.3.	Organic and conventional farming leaching	125
Conclusion générale et perspectives		127
Les choix méthodologiques et leurs limites		128
Les acquis		129
Les perspectives		130
Bibliographie		132
Annexes		

Table des figures

Figure 1-1. Schéma de la cascade de l'azote dans les bassins hydrographiques d'après Billen et al. (2013)	8
Figure 1-2. Représentation temporelle de la minéralisation de la matière organique (a), de l'assimilation de l'azote par une culture semée au printemps (b) et par une prairie (c) au Royaume-Uni (Addiscott et al. 1991).	12
Figure 1-3. Synthèse bibliographique reliant la fertilisation et les concentrations en NO_3^- dans les eaux de drainage dans Billen et al. (2007a).	14
Figure 1-4. Les processus biologiques principaux de la transformation de l'azote dans le sol. En noir, les molécules en phase liquide et en gris, les molécules en phase gazeuse.	16
Figure 1-5. Relation empirique observée entre le surplus et la lixiviation en dessous de la zone racinaire dans les terres arables et les prairies permanentes pâturées (Billen et al. 2013).	22
Figure 2-1. Pluviométrie moyenne annuelle (a), ETP (b) et lame d'eau potentiellement infiltrée (c) dans le bassin de la Seine .	27
Figure 2-2. Typologie des petites régions agricoles (PRA) du bassin de la Seine en fonction des principaux triplets de cultures observés entre 1992 et 2003.	29
Figure 2-3. Temporalité des principaux travaux dans les champs pour les cultures de printemps ou les cultures d'hiver dans les systèmes de grandes cultures du bassin de la Seine (France).	30
Figure 2-4. Carte lithologique du bassin de la Seine avec les exploitations agricoles équipées en 2012-13 (gauche) et 2013-14 (droite)	31
Figure 2-5. Proportion des cultures dans les six départements (Recensement Agreste 2012) et au sein du réseau ABAC.	33
Figure 2-6. Localisation du bassin versant de l'Orgeval dans le bassin de la Seine.	34
Figure 2-7. Schéma des instruments nécessaire pour le prélèvement d'eau sous-racinaire dans une bougie poreuse.	36
Figure 2-8. Photos des prélèvements de GES dans les chambres manuelles (a) et en chambres automatique avec l'analyseur par spectrométrie infra-rouge.	38
Figure 2-9. Schéma descriptif de mesure du potentiel de dénitrification. Pour la nitrification, le principe est le même, les seuls différences sont : l'étape de flush qui disparaît, et la solution de KNO_3 qui est remplacée par une solution (5 ml) de NH_4Cl .	39
Figure 3-1. Location of experimental equipment in the five organic (OF) and two conventional farming (CF) fields. The sub catchment of two equipped drain collectors are also indicated.	45
Figure 3-2. Evolution of rainfall (a), N_2O fluxes measured in automatic and manual chambers for OF1 (b) and CF1 (c), soil mineral N in the layer 0-10 cm (d), and WFPS (e) measured between OF1 and CF1 from March 2013 and June 2014.	53
Figure 3-3. Relationship of soil N_2O emissions measured with continuous sampling in OF1 (a) and CF1 (b) and soil WFPS from March 2013 to June 2014.	54
Figure 3-4. N_2O emission mean fluxes versus soil nitrate contents (0-10 cm) measured (between 5 and 10 cm) during the studied periods in four fields (OF1, OF2, OF3 and CF1) in 2012-2013 and two fields (OF1, CF1) in 2013-2014 (in italics -bold).	55

Figure 3-5. Sub root concentrations at 90 cm depth for OF rotation (a) and CF rotation (b), with median and SD. Spring crops following catch-crops or bare soil are mentioned with (*).	56
Figure 4-1. Nitrification (a) and denitrification (b) potential rates, i.e., nitrate production from nitrite oxidation during nitrification and nitrate reduction into nitrite from denitrification, with means and standard deviation (SD). The curve corresponds to the best fit of a sigmoid function based on all data observed, with $R^2 = 0.4$ for nitrification and $R^2=0.7$ for denitrification.	70
Figure 4-2. Nitrite net productions from ammonium oxidation during nitrification (a), i.e., nT-mT, and from nitrate reduction during denitrification (b), i.e., aT-bT, with means and SD. The curve corresponds to the best fit of a sigmoid function based on all data observed, with $R^2 = 0.3$ for nitrification (a) and $R^2=0.5$ for denitrification (b). Scale for denitrification is ten times greater than for nitrification	70
Figure 4-3. Nitrous oxide (N ₂ O) potential production rates from nitrification (a), i.e., pT, and denitrification (b), i.e., bT-cT, with means and SD. The curve corresponds to the best fit of a sigmoid function, based on all data observed, with $R^2=0.7$ for nitrification (a) and $R^2=0.9$ for denitrification (b). Scale for N ₂ O produced by denitrification is ten times greater than N ₂ O produced from nitrification.	71
Figure 4-4. Mean ($\pm$ SD) of nitrification and denitrification N ₂ O / NO ₃ ⁻ ratios (%) for various temperatures from 5 to 45°C.	72
Figure 5-1. Location of the Orgeval Basin in the Seine Basin and the two sites studied.	78
Figure 5-2. Representation of the Seneque/Riverstrahler model.	80
Figure 5-3. Concentrations of nitrate cascading within the Orgeval basin (see text for explanations, unit in mg N L ⁻¹).	81
Figure 5-4 a. Seasonal average of N ₂ O emission from soils in a forest area and an agricultural slope, redrawn from Vilain et al, (2010). b. Variations of $\delta^{15}\text{N}$ of nitrogen organic matter average over a 1-m soil profile (b), recalculated from Billy et al, (2010). c. Seasonal averages of NO ₃ -N concentrations in the water of the brie aquifer as sampled in the piezometers along the slope, modified from Vilain et al. (2012a).	83
Figure 5-5. Summarizing budget of nitrate transfer and transformation and associated nitrous oxide emissions in the Orgeval Basin. Calculations of are based on the average hydrology from 2006 to 2012. a) current situation based on measurements; b)scenario of pond reintroduction; c) scenario of organic farming.	85
Figure 5-6 a. Interannual NO ₃ -N concentrations in a drainage pond in the Orgeval watershed. Dotted line: NO ₃ -N concentration at the entrance; solid line: simulated NO ₃ -N concentrations in the pond; black dots are the measured NO ₃ concentrations. b. Simulated N fluxes at the outlet of the Orgeval watershed with a range of surface area of ponds (from the reference situation to 10 % of the total surface area of the Orgeval watershed); c. Associated N abatement is shown in comparison (recalculated from Passy et al. 2012).	87
Figure 5-7. Long-term chronicle of observed NO ₃ -N concentrations in the Melarchez River, a headwater stream in the Orgeval watershed.	88
Figure 5-8. Seasonal variations of nitrate concentrations at the outlet of the Orgeval Basin, the year 2006 taken as example. A rather good agreement is obtained between the observations and the simulation for 2006. Compared to the reference simulation, the organic agricultural scenario shows a 45% decrease in annual mean nitrate concentrations (Org. Agri., mean). The amplitude of the response is shown with the exploration of the SD range (Org. Agri., min and max).	89

Figure 6-1. Relations between (a) sub-root concentrations and SMN before the drainage period and (b) sub-root concentrations and SOM, over the profile in all the fields instrumented in ceramic cups: black circles (organic) and open circles (conventional) fields.	103
Figure 6-2. Means ($\pm$ SD) sub-root concentrations (a) from organic rotation with succeeding crops : alfalfa 1 (n=3), alfalfa 2 (n=5), wheat post-alfalfa 2 (n=3), cereals 2 (n=3), grain legumes (n=3), cereal 1 (n=5), cereal 2 (n=5); (b) from conventional rotation with maize (n=1), wheat after crops (n=2) and wheat after legume (n=3).	104
Figure 6-3. Effect of N inputs (amounts and types : organic fertilisers, legumes) and CC on sub-root concentrations in OF fields.	105
Figure 6-4. Sub-root concentrations (a) and leaching (b) from the different cropping systems in each area (S&M, Oise, Yonne) with OF in grey and CF systems in black. Standard deviations represent the range of variations within each rotation.	106
Figure 7-1. Localisation of the farms studied in 2013-14 in the Seine Basin.	113
Figure 7-2. Mean SRC in organic (a) and conventional (b and c) rotations in from the different crops studied in 2013-14 in all the farms. (*) stands for spring crops (maize, sugar beet) or legumes with or without catch-crops (CC) (n= number of field studied).	118
Figure 7-3. (a) Means sub-root concentration of spring crop fields on bare soil (n=6) and with CC (n = 13), (b) the latter being split into winter ploughing (PL) (n=6) and spring ploughing (PL) (n=5).	121
Figure 7-4. Sub-root concentration (SRC) in relation to N exogenous input for different type and amount of organic fertiliser applied in fall in organic or conventional fields and the impact of CC after organic fertiliser.	123
Figure 7-5. Means sub-root concentrations (in histograms) and N leached (black lines) in OF and CF systems studied in S&M (a), Oise (b), Yonne (c) and the West (Val d'Oise, Yvelines) (d)	124

Table des tableaux

Table 1-1. Synthèse des études réalisées sur la lixiviation dans les systèmes biologiques et conventionnels en grandes cultures dans l'Europe du Nord	11
Table 1-2. Synthèse des études ayant réalisées des mesures d'émission de N ₂ O parallèlement sur des systèmes en AB et AC	11
Table 2 1. Quantité d'azote contenu dans les engrais organiques couramment utilisés dans le réseau ABAC	32
Table 2 2. Concentrations des quatre étalons des gaz analysés	40
Table 3-1. Agricultural management of the two conventional fields studied (CF1, CF2), for the previous year (in italic) and during the measurement years.	46
Table 3-2. Agricultural management of the five organic fields for the previous year (in italic) and during the years measurements were taken. In bold, the crops measured for NO ₃ ⁻ leaching and with (*) for N ₂ O emission.	47
Table 3-3. Soil characteristics of the seven fields investigated at three depths: texture (clay, silt, sand), organic carbon (C), total nitrogen (N), soil organic carbon (SOC) and the C/N ratio.	51
Table 3-4. Cumulative N ₂ O emissions calculated with three methods using data obtained with manual or automatic chambers during the period 05 March 2013 – 27 May 2014; n correspond to the number of data sampled or registered, SD is the standard deviation on replicates (five replicates for manual chambers; three replicates for automatic chambers).	54
Table 3-5. N inputs (fertilisation and BNF), N harvest and N losses: sub-root soil concentration at 90-cm depth (SRC), cumulative water inflow (Wi), leaching and N ₂ O fluxes, over the whole OF and CF rotations during the 3 years studied.	57
Table 3-6. Compared N losses between systems over the whole rotation expressed in area-scaled, yield-scaled values or scaled to the total N input.	58
Table 4-1. Overview of dates when potential activities were measured in batch experiments at the different temperatures. Each temperature effect has been analysed for the luvisolic soil sampled at various seasonal initial conditions, and include thus the in situ variability.	64
Table 4-2. Characteristics of the soil and climate conditions: temperature (°C), rainfall (mm), water filled pore space (WFPS, %), ignition loss (%) and nitrate (NO ₃ ⁻), nitrite (NO ₂ ⁻) and ammonium (NH ₄ ⁺) stocks for the four sampling dates.	68
Table 4-3. Nitrification (NO ₃ ⁻ —nit) and denitrification (NO ₃ ⁻ —denit) potential rates at 20°C with their associated N ₂ O emissions and N ₂ O/ NO ₃ ⁻ ratios (%).	69
Table 4-4 Means (±SD) of nitrification (NO ₃ ⁻ —nit), denitrification (NO ₃ ⁻ —denit) potentials and their associated N ₂ O emissions and ratios for each temperature (T).	69
Table 4-5. Fitted parameters of the sigmoid Gaussian relationship to temperature of nitrification and denitrification: optimum activity (A _{max}), optimum temperature (T _{opt}) and σ value (dti). RMSD and R ² of the fitted relationships.	71

Table 5-1. Average potential values for agricultural soils in denitrification and nitrification in experimental conditions (batch experiments at 20°C), and associated N ₂ O production (SD for Standard Deviation, 7 experiments). Percentages of N ₂ O production are also given for comparison.	82
Table 6-1. Main characteristics of the cropping systems studied in the three regions (S&M, Oise, Yonne) in OF and CF with the number of the fields studied, the exogenous fertiliser types used and the tillage presence (X) or absence (No).	98
Table 6-2. Data of the fields instrumented in the eight systems in OF and CF including crops, previous crops with straw buried (b.) or exported (e.) and its net BNF inputs, N exogenous inputs (organic or mineral fertilisers) in 2012-13, SMN before fall tillage (Oct; Nov; Dec), soil solution concentrations (soil solution conc.) and N harvested.	99
Table 6-3. Yearly means of N inputs: N exogenous (fertilisers), total BNF and atmospheric deposition, harvest and N losses: sub-root concentrations, leaching per ha or per protein for OF and CF rotations studied in S&M, Oise and Yonne.	107
Table 7-1. Soils characteristics in the regions studied in terms of textures (clay, sand), soil organic carbon (SOC), N total (N tot), bulk density (BD) and water holding capacity at field capacity (WHC _{FC}).	117
Table 7-2. Summary of the organic (a) and conventional (b) fields instrumented with ceramic cups in 2013-14 with their N inputs and sub-root concentrations (SRC). Fertiliser types: biogas residue (BGR), cow manure (CM), horse manure (HM), mineral (Min), poultry manure (PM), slurry (S). (*) stands for spring crops with catch-crops and (**) stands for spring crops with bare soil.	119
Table 7-3. Catch-crops diversity in organic and conventional fields with sub-root concentration (SRC).	121
Table 7-4. Means of sub-root concentrations (SRC), water inflow and N leached on every farm.	123
Table 7-5. Synthesis of three studies on sub-root concentrations (a) N leaching (b) means for OF and CF systems (weighted by the regions) in the Seine basin. Standard deviation (SD) stands for the difference between farms from different region	126

Abréviations

AB : Agriculture Biologique	IRSTEA : Institut National de Recherche en Sciences et Technologiques pour l'Environnement et l'Agriculture
AC : Agriculture Conventiennelle	ITAB : Institut Technique de l'Agriculture Biologique
ADEME : Agence de l'Environnement et de la Maîtrise de l'Energie	K : Potassium
AESN : Agence de l'Eau Seine-Normandie	LAS : Laboratoire d'Analyses des Sols
BD : Bulk density	MAE : Mesures Agro-Environnementales
BNF : Biological Nitrogen Fixation	N : azote
C : Carbone	N ₂ : diazote
CAN : Calcium Ammonium Nitrate	N ₂ O : oxyde nitreux ou protoxyde d'azote
CH ₄ : Méthane	NH ₃ : Ammoniac
CC : Catch-crop	NH ₄ ⁺ : Ammonium
CF : Conventional Farming	NO ₃ ⁻ : Nitrate
CH ₄ : méthane	NO ₂ ⁻ : Nitrite
CIPAN : Culture Intermédiaire de Piège à Nitrate CO ₂ : dioxyde de carbone	NEA : Nitrification Enzyme Activity
DEA : Denitrification Enzyme Activity	OF : Organic Farming
DIM Astrea : Domaine Intérêt Majeur AgroSciences, Territoires, Ecologie, Alimentation	OGM : Organisme Génétiquement Modifié
ECD : Electron Capture Detector	P : Phosphore
ETM : Evapotranspiration Maximale	PRG : Pouvoir de Réchauffement Global
ETP : Evapotranspiration Potentielle	q : quintaux
ETR : Evapotranspiration Réelle	RU : Réserve Utile
FC : Field Capacity	SAU : Surface Agricole Utile
FAO : Food Agriculture Organisation	S&M : Seine-et-Marne
FID : Flame Ionisation Detector	SMN : Soil Mineral Nitrogen
FNAB : Fédération Nationale d'Agriculture Biologique	SOC : Soil Organic Carbon
FRAB : Fédération Régionale d'Agriculture Biologique	SOM : Soil Organic Matter
GAB : Groupement des Agriculteurs Biologiques	SRC : Sub-Root Concentration
GAP : Good Agricultural Practices	T°C : température (°C)
GES : Gaz à Effet de Serre	t : tonne
GHG : Greenhouse Gas	TCS : Technique Culturelle Simplifiée
ha : hectare	TDM : Tone Dry Matter
IC : Intercultures	TDR : Time Domain Reflector
IDF : Ile-de-France	UAL : Utilized Agricultural Land
INRA : Institut National de Recherche Agronomique	UAN : Urea Ammonium Nitrate
IPCC : Intergovernmental Panel on Climate Change	UMR : Unité Mixte de Recherche
	UR : Unité de Recherche
	Wi : lame drainante simulée
	WFPS : Water-Filled Pore Space
	WHC : Water Holding Capacity

Introduction générale

Contexte

Comme beaucoup de régions dans le monde, très urbanisées et dotées d'une agriculture intensive, le bassin de la Seine peine à restaurer la qualité de ses eaux de surface et souterraines. Après une réduction drastique des rejets ponctuels de phosphore et d'azote, grâce à une amélioration des traitements en stations d'épuration, l'azote essentiellement d'origine diffuse reste largement en excès, ce qui pose des problèmes environnementaux multiples. Dans le bassin de la Seine, la contamination des aquifères a conduit à la fermeture de nombreux captages d'eau potable. Aux zones côtières, l'eutrophisation, entraînant le développement d'algues indésirables, est aussi une manifestation des excès de nutriments issus de l'activité dans le bassin. Une volonté est désormais clairement affichée d'infléchir l'activité agricole dans les espaces ruraux qui entourent la ville vers des formes d'agriculture plus durables, utilisant moins d'intrants et donnant lieu à moins d'émissions de polluants dans l'environnement. La protection des ressources en eau, mais aussi la qualité de l'air, et la volonté de ressouder des liens plus étroits entre le milieu urbain et le milieu rural, sont des motivations principales de cette politique.

L'agriculture biologique, grâce à son cahier des charges rigoureux bannissant tout intrant industriel, constitue un modèle à explorer d'une telle agriculture durable, permettant d'améliorer la qualité de l'eau, en termes de contamination nitrique et de réduction des émissions du gaz à effet de serre qu'est l'oxyde nitreux (N_2O).

Le cadre de l'étude est vaste et complexe. Il est complexe car le cycle de l'azote fait intervenir de nombreux processus microbiens qui possèdent leurs spécificités en termes de facteurs de contrôle, mais dont il est parfois difficile d'en hiérarchiser le rôle dans l'environnement. L'étude est vaste car appréhender les fuites d'azote par l'agriculture biologique, nécessite qu'on se réfère au modèle agricole conventionnel majoritaire. En outre en prenant comme domaine d'étude des exploitations commerciales, aux pratiques variées, des réponses variées des fuites d'azote sont attendues. Deux types de fuite d'azote sont visés par l'étude, celles qui atteignent les eaux de surface ou souterraines par lixiviation du nitrate (NO_3^-), mais aussi les émissions de N_2O . Enfin, et de fait, les approches méthodologiques sont nombreuses et les plus directes ont été choisies.

Par ailleurs, si les fuites d'azote en agriculture conventionnelle ont fait l'objet de nombreux travaux, celles de l'agriculture biologique sont bien moins connues, plus particulièrement en France. C'est pour pallier ce manque de données qu'un réseau d'exploitations agricoles instrumentées a été mis en place (réseau ABAC) dans le cadre de cette thèse.

Le projet ABAC

L'objectif de durabilité des agrosystèmes conduit assez naturellement à renforcer les collaborations interdisciplinaires entre chercheurs agronomes et des autres disciplines. Mais l'implication des agriculteurs eux-mêmes dans les projets de recherche favorise aussi grandement l'innovation, l'accompagnement et le partage des connaissances. C'est dans ce contexte collaboratif que le projet ABAC (www.fire.upmc.fr/abac/) a été créé et financé par la région Ile-de-France (DIM-Astrea) et l'Agence de l'Eau Seine Normandie (AESN), depuis fin 2012, en lien avec le thème « Agriculture » du programme PIREN-Seine. L'objectif de ce réseau de mesures, est d'acquérir des données pour quantifier la lixiviation sous-racinaire de l'azote dans les agrosystèmes. Pour ce faire, dix-huit exploitations agricoles en agriculture biologique (AB) et conventionnelle (AC) sont suivies dans différentes conditions pédoclimatiques dans la région Ile-de-France et ses régions périphériques au sein du bassin de la Seine. Compte tenu de l'importance des cultures céréalières dans cet espace, les exploitations de grandes cultures sont privilégiées. Ce grand panel de situations permet (i) de répertorier les pratiques réalisées par les agriculteurs selon différents contextes pédoclimatiques (ii) de quantifier leurs impacts sur la lixiviation azotée (iii) et de comparer les performances des systèmes étudiés.

Ce projet est fédérateur car il implique les agriculteurs de divers territoires en Ile-de-France et alentour, l'AESN, les Fédérations d'agriculteurs, les Chambres d'Agriculture, etc. Il est participatif car les agriculteurs sont, autant que possible, impliqués dans les opérations d'échantillonnage et dans l'interprétation des résultats. Un comité de pilotage constitué de différents organismes (AB Picardie, Agro-transfert, AESN, Arvalis, Bergerie de Villarceaux, Chambres d'Agriculture de S&M, CNRS, Eau de Paris, Ferme expérimentale de Grignon, FNAB, GAB IDF, INRA, Irstea, UPMC), et des agriculteurs concernés, s'est réuni chaque année, afin de discuter des résultats et réorienter éventuellement le projet. D'ores et déjà, ce projet ABAC contribue donc à combler le déficit manifeste de données de référence concernant l'agriculture biologique, et à interpréter ses effets environnementaux en termes de qualité de l'eau.

Choix méthodologiques

Au cours de cette thèse, nous avons privilégié l'observation *in situ* par rapport à l'expérimentation, ce qui lui confère une dimension intégrée, comprenant de l'agronomie et de la biogéochimie. De plus, l'observation dans des exploitations agricoles appartenant à des particuliers tient compte de la logique de divers acteurs, de leurs objectifs et de leurs contraintes. La négociation et la mise en place progressive du réseau avec les acteurs locaux nous a conduit à observer les pratiques des systèmes de production et à vérifier si elles étaient compatibles avec l'objectif à ne pas dépasser, dans l'eau infiltrée la norme de potabilité de $50 \text{ mgNO}_3 \text{ l}^{-1}$ (soit $11 \text{ mg N-NO}_3 \text{ l}^{-1}$). De plus, afin d'évaluer l'impact d'un système de culture sur les fuites azotées, nous avons choisi d'étudier les rotations culturales

dans leur ensemble, pour prendre en compte les bilans d'azote successifs des différentes cultures et périodes d'intercultures. Cette approche a été rendue possible par l'utilisation d'outils classiques en agronomie (observation de terrain, enquêtes en exploitation agricole, suivis d'un assolement moyen de l'exploitation, mesures des concentrations en nitrate dans les bougies poreuses, analyse de profils en azote) et de données climatiques.

Objectifs et organisation des travaux

Si le bilan environnemental de l'AB par rapport à d'autres systèmes de cultures ne fait pas de doute en ce qui concerne la contamination par les résidus de pesticides ou la préservation de la biodiversité, l'effet des pratiques biologiques sur les fuites d'azote vers les hydrosystèmes (NO_3^-) et l'atmosphère (N_2O), reste encore très controversé. L'objectif principal de cette thèse est de **quantifier les fuites d'azote dans les exploitations biologiques et conventionnelles du bassin de la Seine afin d'examiner si les changements de pratiques et/ou de systèmes agricoles peuvent conduire à améliorer la qualité de l'eau**. Cette étude est la première en France et représente un réel défi qui pose plusieurs questions de recherche :

- (i) **Comment peut-on quantifier les fuites azotées dans des exploitations agricoles commerciales ?**
- (ii) **Pour quelles stratégies opter pour documenter des rotations de 7 à 11 en AB et de 2 à 4 ans en AC au cours d'une thèse de 3 ans ?**
- (iii) **Quelles sont les performances agro-environnementales des exploitations AB par rapport à d'autres systèmes AC ?**
- (iv) **Quel sont les facteurs environnementaux et les pratiques agricoles, qui contrôlent simultanément les fuites d'azote dans l'atmosphère et les hydrosystèmes ?**

Pour y répondre, ce manuscrit est organisé en trois parties :

Partie I. Dans un premier temps, nous présenterons un état des connaissances sur les fuites d'azote dans les agrosystèmes (**Chapitre 1**), comprenant une explication des différents concepts (cascade de l'azote, agriculture biologique), des processus de lixiviation et d'émissions de N_2O ainsi que de leurs facteurs de régulation. Les outils de mesures directes et indirectes des fuites d'azote seront présentés.

Dans un second temps, les caractéristiques majeures des bassins versants dans lesquels s'insèrent les exploitations agricoles seront exposées. Au vu de la diversité des exploitations étudiées, une synthèse de leurs caractéristiques majeures a été réalisée. De plus, l'ensemble des dispositifs expérimentaux, des méthodes utilisées et des outils analytiques mis en œuvre seront présentés (**Chapitre 2**).

Partie II. Cette deuxième partie est constituée de trois études réalisées (2 articles parus et un à soumettre prochainement) à différentes échelles spatiales et temporelles (processus, exploitation et bassin versant). Nous nous sommes intéressés dans un premier temps aux fuites azotées à l'échelle d'une exploitation agricole mixte (AB/AC) en mesurant la lixiviation du NO_3^- et les émissions de N_2O sur l'ensemble des cultures d'une rotation biologique (de 8 ans) et sur une rotation conventionnelle (de 3 ans) (**Chapitre 3**). En parallèle, une comparaison de mesures des émissions de N_2O en chambres manuelles et automatiques a été effectuée au cours d'une période culturale sur deux parcelles (AB et AC).

De plus, afin de mieux comprendre les processus de régulation des émissions de N_2O , des expériences en conditions contrôlées ont été réalisées sur les processus de nitrification et de dénitrification pour une gamme de températures (5-45°C) (**Chapitre 4**). Le but de cette expérience est de déterminer les températures optimales des processus et leur degré de sensibilité. Enfin l'ensemble de ces résultats ainsi que d'autres études antérieures ont servi à l'élaboration d'une synthèse reconstituant la cascade de l'azote à l'échelle du bassin versant de l'Orgeval (**Chapitre 5**). Cette quantification de la cascade de l'azote pourra servir à élaborer des scénarios pour réduire les fuites d'azote dans l'environnement, comme des modifications de pratiques agricoles à l'AB ou l'aménagement du paysage.

Partie III. Cette troisième partie s'articule autour de deux articles, dont un publié et le deuxième en préparation. A l'échelle de l'exploitation agricole, mais avec un échantillon plus large, nous nous sommes intéressés à différents systèmes biologiques et conventionnels du bassin de la Seine. Ainsi en 2012-13, huit systèmes agricoles ont été équipés de bougies poreuses dans trois pôles pédoclimatiques différents (**Chapitre 6**). Afin de prendre en compte différents systèmes agricoles dans un périmètre plus large, en 2013-14, ce sont dix-huit systèmes de culture qui ont été suivis dans six pôles pédoclimatiques (**Chapitre 7**). Ce réseau permet de prendre en compte un panel de pratiques agricoles réalisées en AB et AC dans des contextes pédologiques variés du bassin de la Seine. Cela nous a donc conduits à observer les pratiques réelles des agriculteurs et leurs conséquences au niveau de la lixiviation des NO_3^- ainsi que d'envisager les bonnes pratiques à mettre en place pour concilier production agricole et qualité de l'eau.

Enfin, une **conclusion générale** visera à synthétiser les points importants de ces recherches ainsi qu'à présenter les perspectives envisagées à la suite de cette thèse.

Partie I.

Cadre général de l'étude

1

Etat de l'art

1.1. La cascade de l'azote

Le concept de « cascade de l'azote » décrit quantitativement et qualitativement le phénomène de circulation et d'accumulation de l'azote dans la biosphère, l'hydrosphère et l'atmosphère, en relation avec les apports par les activités humaines, ainsi que les risques environnementaux liés à ces accumulations (Galloway et al. 2003; Billen et al. 2009a; Sutton et al. 2011).

L'azote est en grande majorité sous forme gazeuse inerte (diazote, N_2) dans l'atmosphère (78%), essentiellement sous forme organique ou minérale dans le sol et les hydrosystèmes. Dans le cycle naturel, l'azote de la biosphère est réparti entre les micro-organismes, la solution du sol et les plantes, de telle manière que les surplus d'azote sont faibles. Toutefois les activités anthropiques modifient profondément la circulation de l'azote au sein des bassins versants (**Figure 1-1**). Les rejets ponctuels des villes et les pollutions diffuses, c'est-à-dire produites sur l'ensemble des surfaces agricoles avec des intensités variables, sont responsables des principaux apports d'azote dans l'atmosphère et les hydrosystèmes (Billen et al. 2009b). Si les rejets ponctuels urbains ont pu être réduits à partir des années 1990 grâce à l'amélioration des traitements en stations d'épuration (Billen et al. 2007b), les pollutions diffuses ne peuvent pas être traitées *a posteriori*.

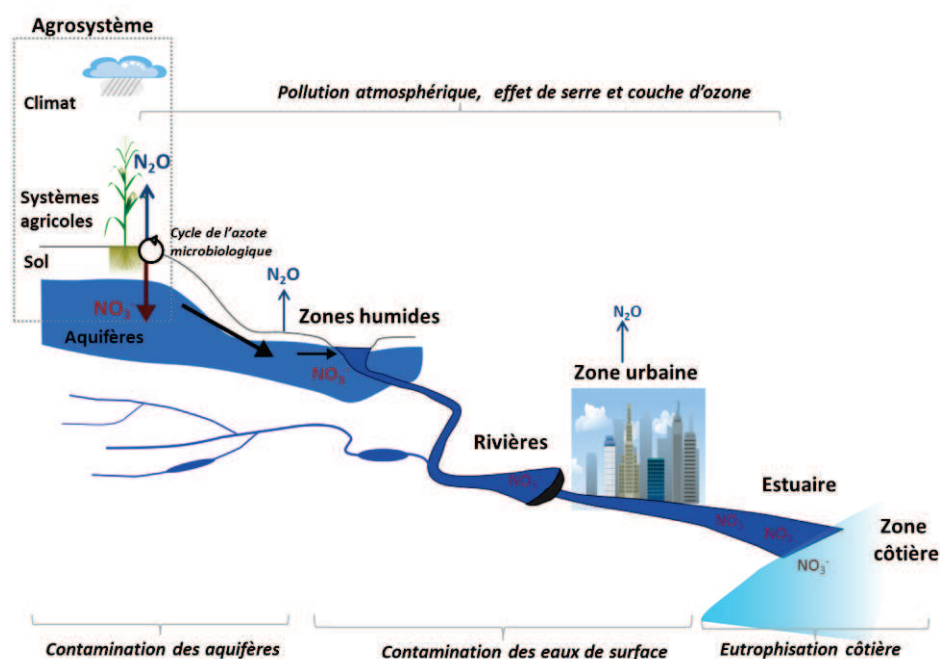


Figure 1-1. Schéma de la cascade de l'azote dans les bassins hydrographiques d'après Billen et al. (2013a)

Cette contamination nitrique génère des impacts sanitaires aux niveaux des aquifères qui dépassent fréquemment la norme de potabilité de 11 mg N l^{-1} (ou $50 \text{ mg NO}_3 \text{ l}^{-1}$), mais elle est également nocive pour la biodiversité des eaux de surface, dès lors qu'elle dépasse 2 mg N l^{-1} (ou $10 \text{ mg NO}_3 \text{ l}^{-1}$). De plus, la contamination nitrique entraînée par les réseaux

hydrographiques jusque dans les estuaires, engendre des blooms algaux et des déficits importants d'oxygène (O_2) en été (Garnier et al. 2007). Ces phénomènes d'eutrophisation et d'anoxie des milieux affectent la vie des organismes marins et l'exploitation des produits de la mer (Camargo et Alonso 2006; Diaz et Rosenberg 2008).

Les émissions de N_2O sont produites en majorité par les activités agricoles (77.9%) (Johnson et al. 2007). Ce gaz à effet de serre, dont le pouvoir de réchauffement global (PRG) est 310 fois supérieur à celui du CO_2 , est de plus actuellement la première substance destructrice de l'ozone stratosphérique (Cicerone 1987; Ravishankara et al. 2009). On distingue les émissions **directes**, lorsqu'elles sont émises par les sols des parcelles agricoles, et les émissions **indirectes**, lorsqu'elles sont émises dans certaines zones naturelles comme les zones humides, les étangs et les nappes perchées (Vilain et al. 2010; Passy et al. 2012). Toutefois les émissions directes sont prédominantes, et représentent au moins 50% des émissions de N_2O totales (Hertel et al. 2012; Jeuffroy et al. 2013). Par exemple, dans les parcelles cultivées, les émissions directes sont de l'ordre $0.2-1.3 \text{ kg N-N}_2\text{O ha}^{-1} \text{ an}^{-1}$ (Stehfest et Bouwman 2006; Vilain et al. 2010), alors que les émissions indirectes sont estimées à environ $0.02 \text{ kg N-N}_2\text{O ha}^{-1} \text{ an}^{-1}$ (Garnier et al. 2009; Garnier et al. 2013).

Par ailleurs, la volatilisation d'ammoniac (NH_3) contribue à l'acidification des sols recevant les retombées riches en ammonium (NH_4^+) et entraîne des maladies broncho-pulmonaires en association avec les particules en suspensions. En France, l'agriculture est responsable de 95% des émissions de ce gaz qui sont particulièrement liées aux activités d'élevage (bâtiments, épandage, prairies) (Misselbrook et al. 2000; Sanz-Cobena et al. 2008; Loubet et al. 2011).

Notre étude se situe à l'amont de la cascade de l'azote, au niveau des **agrosystèmes**, définis par un ou plusieurs systèmes de cultures, différents types de sols, une topographie et un climat (Meynard et al. 2001).

Afin de limiter la pollution diffuse dans l'environnement, Thieu et al. (2011) ont suggéré qu'une conversion intégrale du bassin de la Seine à l'AB permettrait de réduire considérablement la contamination nitrique. Afin de tester cette hypothèse, nous nous sommes donc intéressés à travers la littérature, aux fondements, c.-à-d. aux pratiques agricoles et aux performances agro-environnementales spécifiques des systèmes biologiques du nord de l'Europe.

1.2. Les systèmes biologiques en Europe

Les systèmes biologiques se différencient des systèmes agricoles conventionnels par des modalités techniques spécifiques décrites dans un cahier des charges européen. Actuellement, la surface agricole utile (SAU) en AB de l'Union Européenne (UE) par rapport à la SAU totale est de 6 %, dont l'Autriche, la Suède, l'Estonie possèdent les plus grandes proportions avec 19.6%, 15.4% et 15.3% respectivement. Toutefois en termes de surfaces sur l'ensemble du territoire agricole de l'UE, les surfaces dédiées à l'AB sont les plus importantes en Espagne (16%), Italie (12%), Allemagne (11%) et France (11%).

1.2.1. Cahier des charges de l'agriculture biologique

Le cahier des charges européen (Règlement CE N° 889/2008) définit l'AB comme un mode de production agricole systémique reposant sur l'activité microbienne des sols, le recyclage des déchets organiques et les cycles biologiques et biogéochimiques, en interdisant les produits de synthèse (engrais, pesticides, médicaments) et les organismes génétiquement modifiés (OGM). En contrepartie, l'AB utilise des engrais organiques, une rotation diversifiée, la sélection variétale, l'utilisation de couverts végétaux, le désherbage mécanique et thermique (Bàrberi 2002). La réglementation européenne permet de n'utiliser que quelques produits biocides d'origine naturelle, comme le sulfate de cuivre (ou bouillie bordelaise). L'utilisation de ces produits est minimale en AB, seulement 10% des exploitants en utilisent, particulièrement en maraîchage (Walz 1999).

En pratique, la définition générale de l'AB recouvre une grande diversité de pratiques, en fonction notamment de la géographie, des pays et des régions (Meynard et al. 2001; Doré et al. 2013).

Concernant la fertilisation organique, il n'existe pas de limitation pour les apports de nutriments (N, P, K), bien qu'ils soient en général inférieurs à la limitation de la Directive Nitrate de 170 kg N ha⁻¹ an⁻¹ pour l'ensemble de l'exploitation, en zone vulnérable. L'origine des produits organiques utilisés comme fertilisants n'est pas non plus spécifiquement réglementée : les engrais peuvent provenir d'exploitations d'élevages biologiques ou conventionnelles (fumier, compost, fientes, lisier), ainsi que de produits conventionnels industriels, comme de la vinasse de betterave.

1.2.2. Les performances agro-environnementales

De récentes méta-analyses ont mis en évidence les impacts particulièrement positifs de l'AB sur la biodiversité de la flore et de la faune, l'activité microbiologique des sols et la réduction des pesticides (Stolze et al. 2000), mais avec des controverses concernant la lixiviation du NO₃⁻ et des émissions de N₂O (Mondelaers et al. 2009).

Pour notre étude, nous avons restreint les études réalisées sur les fuites d'azote dans les systèmes de grandes cultures en Europe.

Concernant la lixiviation du NO₃⁻, les publications scientifiques révèlent une grande variabilité des résultats, liée à la variabilité des systèmes, aux méthodes utilisées ainsi qu'à la localisation des sites. Au final, il en résulte que dans les six études qui ont comparé la lixiviation dans les systèmes de grandes cultures, une seule étude mesure des performances négatives pour l'AB. La moyenne de l'ensemble de ces études montre des fuites par lixiviation en grande culture AB inférieures de 28% à celles observées en AC (Table 1-1).

En ce qui concerne les émissions de N₂O des grandes cultures biologiques en Europe, six publications scientifiques montrent là aussi une grande variabilité des résultats du fait des conditions expérimentales variées (Table 1-2). D'après l'ensemble de ces études, la moyenne des émissions est inférieure de 21% dans les systèmes AB par rapport aux systèmes AC.

Table 1-1. Synthèse des études réalisées sur la lixiviation dans les systèmes biologiques et conventionnels en grandes cultures dans l'Europe du Nord.

Références	Localisation	Rapport AB/AC
Hansen et al, 2000	Danemark	0.56
Korsaeth et Eltun, 2000	Norvège	0.86
		0.61
Kristensen et al, 1994	Danemark	1.22
Stopes et al, 2002	Royaume-Uni	0.88
Haas et al, 2002	Allemagne	0.46
Berg et al, 1999	Allemagne	0.46
		0.43
		0.37
		0.30
	Moyenne	0.62
	<i>ecart-type</i>	<i>0.29</i>

Table 1-2. Synthèse des études ayant réalisées des mesures d'émission de N₂O parallèlement sur des systèmes en AB et AC.

Références	Localisation	Rapport AB/AC
Syväsalo et al, 2006	Finlande	1.17
Haas et Wetterich, 2000	Allemagne	0.74
Stalenga et Kawalec, 2008	Pologne	0.34
Flessa et al, 2002	Allemagne	0.83
Olesen et al, 2006	Pays-Bas	0.48
	Allemagne	0.64
	Autriche	0.60
Petersen et al, 2006	Autriche	2.06
	Danemark	0.73
	Finlande	0.92
	Italie	0.54
	Royaume-Uni	0.42
Moyenne		0.79
<i>ecart-type</i>		<i>0.46</i>

De plus de récentes méta-analyses, au lieu d'exprimer les fuites d'azote dans l'environnement par unité de surface (area-scaled losses), les ont exprimées par unité de production (yield-scaled losses). Les rendements étant en général plus faibles en AB dans les régions d'agriculture intensive (Allemagne, Danemark, Pays-Bas, France), notamment pour les céréales (-20%) (Lotter 2003; de Ponti et al. 2012), ce mode d'expression des résultats a tendance à pénaliser les performances de l'AB, tant pour la lixiviation des NO₃⁻ (Pacini et al. 2003; Korsaeth 2008; Tuomisto et al. 2012) que les émissions de N₂O (Flessa et al. 2002; Olesen et al. 2006; Skinner et al. 2014). Toutefois ces études scientifiques ont toutes été réalisées dans **des sites expérimentaux** et non dans des exploitations agricoles commerciales et les systèmes biologiques spécifiques du bassin de la Seine ne sont pas représentés.

Afin de mieux comprendre les différences mesurées dans les systèmes en AB et AC, nous allons nous intéresser aux processus et aux facteurs de régulation des fuites d'azote (NO_3^- , N_2O) depuis les agrosystèmes jusque dans les différents compartiments de l'environnement (hydrosystème, atmosphère).

1.3. La lixiviation du nitrate dans l'hydrosystème

1.3.1. Les mécanismes de la lixiviation

La lixiviation, aussi appelée par abus de langage *lessivage*, consiste à l'extraction de molécules solubles par un solvant. Dans le sol, elle correspond au mouvement vertical de l'eau mobile et de ses solutés à travers le profil du sol hors de portée des racines, jusqu'à la nappe phréatique ou les drains. La majorité de l'azote lixivié est sous la forme de NO_3^- , très mobile et soluble dans l'eau (357 g l^{-1} à 25°C) alors que l'ammonium (NH_4^+) reste fixé par adsorption sur les fractions argileuses du sol (Scherer 1993) et que le nitrite (NO_2^-) est peu abondant.

La lixiviation à l'échelle locale est due à la conjonction de plusieurs facteurs : la présence d'un reliquat d'azote minéral à la sortie de l'automne, l'occurrence d'un drainage, l'absence d'absorption par le système racinaire et l'absence de réduction de NO_3^- dans les couches anoxiques (Sebillotte et Meynard, 1990; Addiscott et al. 1991; Hoffmann et Johnsson 1999).

Le transfert des ions NO_3^- dans le sol est un phénomène de type convectif-dispersif. La convection est due au déplacement de l'eau et la dispersion résulte à la fois de l'agitation moléculaire au sein du liquide et de l'hétérogénéité des vitesses de transfert de l'eau dans des pores de taille variable. Le déplacement du NO_3^- est non conservatif car généralement associé à des processus de transformations biochimiques (dénitrification) au cours de la cascade de l'azote (Lafolie et Neel 1997). La convection présente deux orientations variables au cours du temps : ascendant quand les plantes se développent ; descendant quand les précipitations excèdent l'évapotranspiration (Garnier et al. 2001). L'absorption de l'azote par les plantes et la minéralisation du sol est variable au cours de l'année et selon les types des cultures (Figure 1-2).

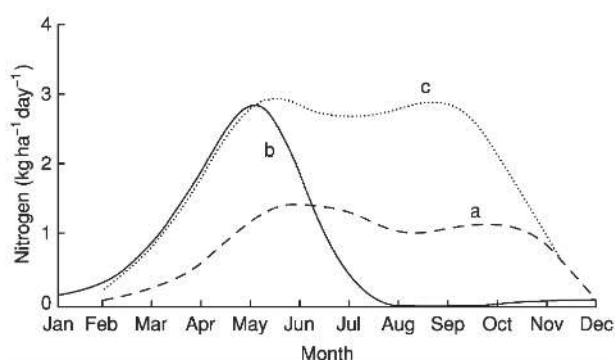


Figure 1-2. Représentation temporelle de la minéralisation de la matière organique (a), de l'assimilation de l'azote par une culture semée au printemps (b) et par une prairie (c) au Royaume-Uni (Addiscott et al. 1991).

D'une manière générale, la lixiviation se réalise à l'automne lorsque la pluviométrie est importante, la minéralisation active et que le prélèvement par les plantes est faible (Macdonald et al. 1989 ; Machet et Mary 1990 ; Powlson et Addiscott 2005). Le cas des prairies est un peu différent, du fait qu'elles sont pérennes et qu'elles puisent l'azote tout au long de l'année, toutefois en cas de retournement, de coupes ou de pâturage excessif par les animaux, des risques de lixiviation existent également (Hennart et al. 2010).

1.3.2. Les facteurs de régulation

1.3.2.1. Les facteurs environnementaux

Le climat

Les variations saisonnières des précipitations et de la température sont des facteurs de contrôles des bilans d'eau et par conséquent des fuites d'azote par lixiviation. Dans tous les cas, plus les précipitations et les écoulements de surface sont importants, plus les exportations d'azote sont fortes tant à l'échelle du groupement de parcelles (Tomer et al. 2003; Tiemeyer et al. 2006) qu'à celle des bassins versants (Thieu et al. 2009).

Ces facteurs climatiques impactent également le processus de minéralisation/organisation au cours de l'année. Par exemple, en conditions contrôlées, une augmentation de température de 10°C (de 25 à 35°C) double les quantités d'azote perdues par jour (Medina et al. 2014).

Les conditions pédologiques

Les caractéristiques d'un sol déterminent son système poreux et par conséquent ses mouvements d'eau, d'air et de nutriments. Les propriétés de rétention en eau des sols varient en fonction de la profondeur du sol, de la texture (Bruand et al. 2004), de la teneur en matière organique (Franzluebbers 2002) et de la capacité d'échange cationique (Gaines et Gaines 1994).

De nombreux auteurs s'accordent sur le fait que les quantités d'azote lixiviées dans les limons sableux sont deux à trois fois supérieures à celles en limons argileux (Bergstrom et Johansson 1991; Hansen et al. 2000; Beaudoin et al. 2005; Van Es et al. 2006). En conditions contrôlées, une étude a montré que 63% de l'ajout de NO_3^- était lixivié dans un sol sableux, contre 58% dans un sol limoneux sableux et 37% dans un limon sableux argileux (Gaines et Gaines 1994).

1.3.2.2. Les pratiques agricoles

Les apports d'engrais

Les apports d'engrais minéraux sont connus pour augmenter les risques de lixiviation, surtout lorsqu'ils dépassent le seuil des 160 kg N ha⁻¹, ce qui engendre des concentrations dans la solution du sol et dans les eaux de drainage au-dessus de la norme de potabilité de 11 mg N l⁻¹ (**Figure 1-3**) (Kristensen et al. 1994; Billen et al. 2007a; Evanylo et al. 2008). Confortant les résultats ci-dessus, plusieurs traçages à l'isotope ¹⁵N ont montré que, en bilan annuel pour un apport de 170 kg N ha⁻¹ sur du blé au printemps, 45-65% de l'azote apporté était fixé dans les parties aériennes de la plante, 20% retenu dans le sol (par les racines, les

micro-organismes du sol et la matière organique) et 5-35% perdu par lixiviation au-delà de 1m de profondeur (Recous et al. 1988; Macdonald et al. 1997; Sebilo et al. 2013), avec des variations en fonction de l'activité microbienne des sols et de l'assimilation par les plantes (Destain et al. 2010).

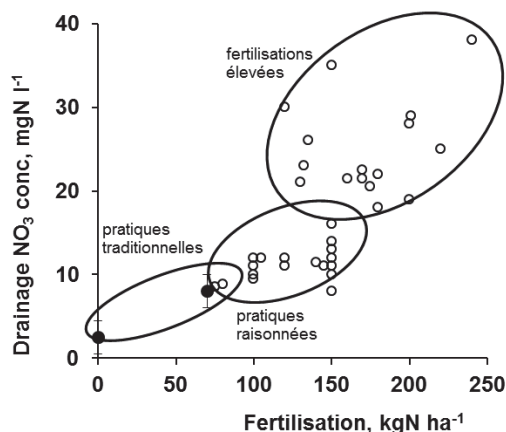


Figure 1-3. Synthèse bibliographique reliant la fertilisation et les concentrations en NO_3^- dans les eaux de drainage dans Billen et al. (2007a).

En revanche les engrais organiques réagissent au processus de lixiviation de manière très différente selon leur ratio C/N. Plus le C/N est élevé, plus les quantités d'azote immobilisée par organisation sous forme de biomasse de micro-organismes seront importantes, qu'impliquera une moindre disponibilité immédiate de l'azote pour la culture, mais aussi une meilleure résistance à la lixiviation (Sullivan et al. 1991). Ainsi lors d'une application de fumier (C/N élevé), 8% se retrouve dans les plantes dans l'année de l'application, et 84-93% dans le sol (Cheneby et al. 1994), où sa minéralisation lente le rendra disponible pour les cultures des années suivantes.

La fixation symbiotique des légumineuses.

La fixation d'azote biologique est réalisée par une symbiose dans les nodosités des légumineuses et les bactéries du genre *Rhizobium* capables de réduire l'azote atmosphérique (N_2) en azote ammoniacal. Cette réaction fortement endothermique nécessite que la plante fournisse les sucres et l'énergie issus de la photosynthèse et bénéficie en retour des acides aminés qui y sont produits. Cette relation est spécifique entre chaque plante-hôte et une espèce de rhizobium (Leclerc 2001).

En agriculture biologique, la fixation d'azote atmosphérique constitue le moyen le plus direct et économique d'apporter de l'azote dans les systèmes de cultures. L'ordre de grandeur de l'azote fixé par les légumineuses est de $400 \text{ kg N ha}^{-1} \text{ an}^{-1}$ pour la luzerne, de $250 \text{ kg N ha}^{-1} \text{ an}^{-1}$ pour le trèfle violet, $150 \text{ kg N ha}^{-1} \text{ an}^{-1}$ pour le pois et $100 \text{ kg N ha}^{-1} \text{ an}^{-1}$ pour les pâtures avec légumineuse (Muller et al. 1993). Il est cependant plus rigoureux de calculer la fixation symbiotique à partir de la production de la culture de légumineuse concernée. Anglade et al. (soumis) ont proposé des formules pour calculer la fixation symbiotique des légumineuses à

graines (**Equation 1-1**) et fourragères (**Equation 1-2**), en fonction du rendement de récolte exprimé en contenu azoté (yield-N).

Equation 1-1. Fixation symbiotique ($\text{kg N ha}^{-1} \text{ an}^{-1}$) = $\text{FS} * [0.71 * \text{yield-N/NHI}] \approx 1.23 * \text{yield-N}$

Equation 1-2. Fixation symbiotique ($\text{kg N ha}^{-1} \text{ an}^{-1}$) = $\text{FS} * [0.78 * \text{yield-N/NHI}] \approx 1.47 * \text{yield-N}$

où NHI représente le rapport de la fraction récoltée par rapport à la production aérienne totale, et FS, le facteur de correction pour tenir compte de la production souterraine.

La totalité de l'azote fixé par une culture de légumineuse est potentiellement disponible pour la culture suivante lorsqu'elle est enfouie en fin de saison. Si elle est récoltée, seule la fixation correspondant aux parties souterraines et non récoltées est disponible. Dans les deux cas, la minéralisation de l'azote organique fixé peut donner lieu à la lixiviation durant la période qui suit la destruction de la légumineuse (Justes et al, 2001).

Implantation de cultures intermédiaires pièges à nitrate.

La mise en place de cultures intermédiaires pièges à nitrate (CIPAN) est l'une des mesures principale adoptée par la Directive Nitrate en zones vulnérables, du fait de leur efficacité sans équivoque (Justes et al. 2012). Toutefois cette efficacité est variable selon les conditions climatiques et le développement de la plante. En effet, une CIPAN semée tard en automne avec un faible développement ne prélèvera pas plus de $10 \text{ kg N}^{-1} \text{ ha}^{-1}$, tandis qu'une CIPAN semée tôt après les récoltes avec un bon développement pourra prélever entre $30\text{-}50 \text{ kg N}^{-1} \text{ ha}^{-1}$ et permettre de limiter les risques de fuites. De même, l'incorporation à la fin de l'automne ou au début du printemps, n'aura pas le même impact sur la lixiviation. Un retournement de prairie (raygrass, trèfle) intervenant au début de l'automne engendre 78 kg N ha^{-1} de lixiviation, comparé à 40 kg N ha^{-1} en fin d'automne, ou 5 kg N ha^{-1} au début du printemps (Francis et al. 1992).

Enfin, la minéralisation des CIPAN est en général assez lente, se réalise sur plusieurs années et peut ainsi contribuer à la lixiviation au cours de années suivantes. Des études ont en effet montré que les CIPAN pouvaient limiter les fuites azotés sur une année mais les augmenter ensuite si leur apport d'azote n'est pas pris en compte dans le calcul de la fertilisation (Catt et al. 1998; Constantin et al. 2011).

1.4. Les émissions d'oxyde nitreux dans l'atmosphère

1.4.1. Les processus responsables des émissions

Les processus biologiques majeurs responsables des émissions de N_2O , sont la nitrification, la dénitrification et la nitrification-dénitrifiante (**Figure 1-4**).

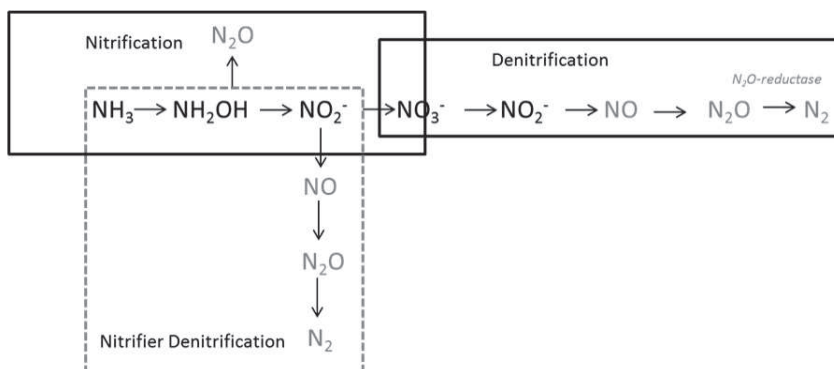


Figure 1-4. Les processus biologiques principaux de la transformation de l'azote dans le sol. En noir, les molécules en phase liquide et en gris, les molécules en phase gazeuse.

1.4.1.1. La nitrification

La nitrification est due à des bactéries oxydant NH_4^+ en NO_2^- (nitritation, **Equation 1-3**) puis en NO_3^- (nitratisation, **Equation 1-4**) en conditions aérobies. Les étapes successives de la nitrification se font par l'intermédiaire de deux familles de bactéries autotrophes présentes dans le sol : *Nitrosomas* pour la nitritation et *Nitrobacter* pour la nitratisation.

Equation 1-3. $2 \text{NH}_4^+ + 3 \text{O}_2 \rightarrow 2 \text{NO}_2^- + \text{H}^+ + 2 \text{H}_2\text{O} + \text{ATP}$

Equation 1-4. $2 \text{NO}_2^- + \text{O}_2 \rightarrow 2 \text{NO}_3^- + \text{ATP}$

1.4.1.2. La dénitrification

La dénitrification est un processus respiratoire microbien au cours duquel les oxydes d'azote solubles (NO_3^- ou NO_2^-) sont transformés en composés gazeux : oxyde nitrique (NO), N_2O et/ou N_2 . Elle fait intervenir de nombreuses espèces bactériennes, majoritairement hétérotrophes (dont celles du genre *Pseudomonas*) (dont l'activité est maximale, dans des conditions d'anoxie, de présence de NO_3^- et d'une source de carbone assimilable). Les 2/3 de la microflore réduisent les NO_3^- en NO_2^- , mais seulement un faible pourcentage semble capable de réaliser toute la chaîne de réactions ([Hénault et Germon, 1995](#)), ce qui explique que la réaction s'arrête au stade NO ou N_2O , avant d'atteindre le stade de N_2 inerte. En conditions contrôlées, les émissions lors de la dénitrification sont près de 100 fois supérieures à celles des émissions de nitrification ([Vilain et al, 2010](#)).

1.4.1.3. *La nitrification dénitrifiante*

La nitrification dénitrifiante est une autre voie par laquelle les nitrites produits lors de la nitrification sont transformés en NO, N₂O et/ou N₂. Ce processus peut contribuer jusqu'à 30% des émissions de N₂O en conditions d'anaérobiose avec peu de carbone (C) disponible (Wrage et al. 2001; Khalil et al. 2004; Kool et al. 2011).

1.4.2. Les facteurs de régulation

1.4.2.1. *Les teneurs en oxygène*

La disponibilité en oxygène dans le sol est difficilement mesurable sur le terrain et est généralement substituée par des mesures de taux de saturation en eau (appelé *water-filled pore space* en anglais, WFPS). Le degré de saturation du sol en eau est déterminant, la dénitrification disparaissant dans les sols dont le niveau d'humidité devient inférieur à 60-70% de capacité de rétention (Nommik, 1956), tandis que pour les teneurs comprises entre 30-70% la nitrification serait prédominante (Firestone et al. 1980; Bateman et Baggs 2005). Au-delà de 80% du taux de saturation en eau, le N₂O est consommé et le N₂ deviendrait le seul produit final de la réaction (Veldkamp et Keller 1997). Ainsi, il est fréquent d'observer des pics d'émission de N₂O lorsque les conditions des sols apparaissent favorables à la dénitrification, notamment après des événements d'épisode pluvieux (Dobbie et Smith 2001).

1.4.2.2. *Les apports d'azote et de carbone*

Plusieurs études suggèrent que la biodisponibilité d'azote et de carbone ont un effet direct d'accroissement des émissions de N₂O (Firestone et al. 1980; Myrold and Tiedje 1985), comme c'est le cas lors des apports de fertilisants organiques ou minéraux (Bouwman 1996; Mosier et al. 1998; Bouwman et al. 2002). Ces études soulignent que l'intensification et le développement de la fertilisation azotée sont générateurs d'émission de N₂O, tant pour les apports d'engrais azotés minéraux (Laville et al. 1999; Dobbie et Smith 2001) qu'organiques (Petersen 1999; Meng et al. 2005; Jones et al. 2007; Sanz-Cobena et al. 2012; Aguilera et al. 2013).

Hénault et al. (1998) ont montré que dans les sols fertilisés avec des engrais synthétiques, les émissions de N₂O sont linéairement liées aux apports d'azote (Equation 1-5). Cela revient à admettre qu'en absence de fertilisation l'émission de N₂O est de l'ordre de 1.5 à 1.9 kg N ha⁻¹ an⁻¹, soit de 0.4 à 0.7% de l'azote apporté transformé en N₂O. Toutefois, ces facteurs de contrôle ne deviennent importants que lorsque les conditions d'humidité du sol sont favorables à la dénitrification (Corre et al. 1996).

Equation 1-5. $N-N_2O \text{ (kg N ha}^{-1} \text{ an}^{-1}) = 1.88 + 0.0042 N \text{ (kg N ha}^{-1} \text{ an}^{-1}, \text{ fertilisants)}$.

Concernant les apports organiques, les facteurs d'émission de N₂O en fonction des apports azotés varient de 0.1-0.4% (urée), 0.1-1.4% (NH₄NO₃) et 1.3-4.3% (eaux usées), 0.2-0.5% (lisier), 0.5-2.6% (fiente de poules) (Sommer et al. 2004; Jones et al. 2007). Enfin, l'impact de l'enfouissement des cultures intermédiaires piège à nitrate (CIPAN) sur les émissions de N₂O

reste modeste, avec un supplément d'émissions de $0.11 \text{ kg N ha}^{-1} \text{ an}^{-1}$ dans l'année qui suit le retournement (Bertuzzi et al, 2012).

1.4.2.3. La température et le pH

Une corrélation positive entre la température ($T^{\circ}\text{C}$) du sol et les émissions de N_2O a été observée entre 5 et 20°C en conditions contrôlées (Maag et Vinther 1996; Schaufler et al. 2010). Cette augmentation des émissions est à mettre en relation avec l'augmentation des activités de nitrification et de dénitrification (Knowles 1982). Toutefois très peu d'études ont exploré une large gamme de températures afin de tester les conditions limites de ces processus (Godde et Conrad 1999).

Sur le terrain, l'interaction entre la température et les émissions sont encore mal connues et les conclusions divergent quant à la relation des émissions de N_2O avec la température (Barnard et al. 2005). En effet, une méta-analyse montre que l'impact d'une augmentation de T ($0.3\text{-}6.0^{\circ}\text{C}$) engendre une augmentation significative de la respiration du sol, de la minéralisation de l'azote et du développement des plantes (Rustad et al. 2001). Cette projection est en relation avec les modèles qui montrent que le réchauffement climatique augmente la vitesse des réactions microbiennes du fait d'une plus grande disponibilité des nutriments. D'autre part, une augmentation de 2°C sur une prairie a montré que les émissions de N_2O avaient tendance à décroître du fait de l'acclimatation des micro-organismes (Luo et al. 2001; Pendall et al. 2004).

Dans le cas d'une acidification ($\text{pH} < 5$), on observe un accroissement du ratio $\text{N}_2\text{O}/(\text{N}_2\text{O} + \text{N}_2)$ du fait d'une modification des communautés bactériennes de la dernière étape de dénitrification (Čuhel et al. 2010).

1.5. Les techniques de mesures des fuites d'azote

Les mesures de fuites azotées dans les parcelles agricoles ont fait l'objet de nombreuses recherches méthodologiques. Nous mentionnerons ici les méthodes de mesures directes et indirectes couramment utilisées, comme les bilans azotés, les facteurs d'émission, l'isotopie (^{15}N) et les modèles de prédiction.

1.5.1. Les méthodes directes

1.5.1.1. La lixiviation du nitrate

Quatre méthodes principales permettent de mesurer la lixiviation dans les sols agricoles. En partant des méthodes les plus locales à celles intégrant le plus de variabilité spatiale, ces méthodes se basent sur les mesures des reliquats d'azote minéral et des concentrations en bougies poreuses, le suivi de dispositifs lysimétriques ou de réseau de drainage.

Les reliquats azotés.

Les analyses de carottes de sol, réalisées en début de période de drainage, permettent de mesurer conjointement l'humidité, la teneur en azote minéral et éventuellement la masse volumique de sol. On peut alors calculer le stock d'azote minéral disponible pour la

lixiviation. Dès que la réserve en eau dans le sol est remplie, le modèle de Burns (Burns 1976) calcule, la fraction d'azote lessivé (%). Une étape de validation des résultats entre les mesures de reliquats et dans les bougies poreuses peut toutefois être nécessaire (Gaury, 1992).

Egalement, l'association des mesures de reliquats azotés réalisées à plusieurs reprises dans l'année et d'un modèle simple de calcul peut permettre de convertir les données de stock en flux (eau et azote), au moins en sol nu, par exemple via l'outil LIXIM (Mary et al. 1999). La difficulté des reliquats de sol consiste dans la représentativité des échantillons analysés, qui nécessite de répéter dans l'espace et dans le temps ces prélèvements, destructifs et coûteux en temps de travail (Hennart et al. 2009).

Les bougies poreuses.

La pose de bougies en céramique poreuse est la technique la plus classique pour des suivis de longue durée en plein champ. Elles permettent de prélever *in situ* des concentrations de façon non destructrice et répétée au cours du temps dans des parcelles agricoles. Elles peuvent être positionnées à la verticale, à l'horizontale ou avec un angle de 45° (Curley et al. 2010), à une profondeur d'environ 90 cm, en limite de zone racinaire (Lord et Shepherd 1993) ou à diverses profondeurs pour suivre l'évolution des teneurs des éléments recherchés dans un profil de sol (Wagner 1962; Hansen et Harris 1975). Le nombre de bougies installées varie de 2 à une douzaine.

Une bougie poreuse est formée d'une cupule cylindrique creuse, généralement collée à un tube de PVC fermée hermétiquement par un bouchon par lequel passent un ou deux tubes. La cupule est composée d'un matériau poreux, céramique en général, supposé inerte chimiquement. L'extraction de la solution du sol s'effectue par application d'une sous-pression à l'intérieur de la bougie, qui permet l'aspiration à travers la céramique de l'eau libre du sol. Même si la quantité d'eau dépend de la sous-pression appliquée, Poss et al. (1995) montre que les concentrations ne sont pas significativement différentes pour des sous-pressions comprises entre -0.4 et -0.6 bar.

Les bougies poreuses horizontales et verticales n'ont pas montré de différences significatives de concentrations pour des sols sans fissure (Bowman et al. 2002). Egalement, elles ont été évaluées fiables pour les mesures des concentrations en NO_3^- dans la solution du sol, avec des résultats comparables dans des lysimètres (Webster et al. 1993). Les biais de mesures peuvent être réduits en réalisant plusieurs répétitions et en utilisant des intervalles courts entre les prélèvements et des durées de sous-pression constantes (Hansen et Harris 1975). De plus, les bougies poreuses horizontales dont l'installation provoque une perturbation importante du sol, impose une année de stabilisation avant le début des mesures. L'installation des bougies nécessite un bon contact entre la bougie et le sol. Certains auteurs utilisent de la bentonite pour limiter les fuites verticales. Les premiers prélèvements ne doivent pas être pris en compte pour s'affranchir de la phase de stabilisation.

Plusieurs phénomènes peuvent intervenir au contact céramique-solution du sol. Des libérations de calcium, magnésium et sodium par les parois poreuses ont été observées (Grover et Lamborn 1970). Depuis les années 90, des travaux (lixiviation des pesticides surtout) sont réalisées sur bougies poreuses en téflon (Maître 1991). Ce matériau est inerte, mais les constructeurs doivent effectuer un traitement pour le rendre hydrophile et créer

une microporosité. Ces traitements engendrent une réactivité des bougies en téflon (Prenat™, Timco™) par rapport aux solutions du sol encore plus importante que dans les bougies en céramique (Creasey et Dreiss 1988; Maître et al. 1991). Le rinçage avant l'installation permet toutefois de limiter les problèmes de contaminations ioniques.

La profondeur à laquelle il convient de calculer la lixiviation est celle de la culture à l'enracinement le plus profond, car l'azote qui a migré en profondeur peut être ensuite absorbé par les cultures suivantes (Thorup-Kristensen et al. 2003). La profondeur des racines peut être limitée par la roche mère (socle calcaire en Beauce) ou par le développement de la plante en hiver. En période hivernale (Novembre à Avril), 80-100% des racines d'un colza sont au-dessus de 90 cm de profondeur (Barraclough 1989).

Enfin, pour estimer la lixiviation du NO_3^- , avec les concentrations dans l'eau des bougies, il faut estimer les flux d'eau (volume d'eau percolée), soit par la loi de Darcy, à partir de mesures *in situ* du gradient de charge (Vachaud et al. 1978), soit par un calcul de bilan hydrique, à partir d'une estimation de tous les autres termes du bilan (pluviométrie, évapotranspiration, variation de stock d'eau dans le sol et flux de ruissellement éventuel) (Burns 1976).

Les lysimètres.

Les lysimètres sont des dispositifs permettant d'étudier et de mesurer l'évolution de l'eau dans un sol. Le lysimètre simple se présente comme un cylindre en métal, béton ou plastique ouvert en surface et rempli par le sol à tester. Ils constituent une méthode adaptée pour les essais expérimentaux de long terme (Carter et Gregorich 2007), car leur installation nécessite d'importants travaux. Les lysimètres récupèrent l'eau d'un profil de sol lorsque l'humidité du sol dépasse la capacité au champ.

Toutefois les lysimètres ne permettent pas d'estimer le drainage, car il existe des effets de bordure et le recueil d'eau au potentiel matriciel nul conduit à le sous-estimer (Weihermuller et al. 2007). Les lysimètres ouverts avec fibre de verre sont une version plus récente et prometteuse. Ils permettent d'estimer le drainage si le nombre de répétitions est suffisant et surtout si des eaux représentatives de celles qui percolent sont recueillies (Feaga et al. 2010).

Le réseau de drainage.

Le drainage artificiel en évacuant les excès d'eau, réduit les phases d'engorgement et donc d'anoxie des sols hydromorphes. Les parcelles drainées peuvent être considérées dans des travaux scientifiques comme des lysimètres géants (Arlot, 1999; Kladvko et al. 2004). Le drainage artificiel a pour conséquence de redistribuer les termes du bilan hydrique, en diminuant la part d'infiltration profonde au profit d'un écoulement plus superficiel. Une hypothèse souvent faite, est que le réseau de drainage capte la majorité des flux de percolation (Henine et al. 2010).

Les fuites par drainage, importantes pour les ions NO_3^- et Ca^{2+} varient essentiellement avec la pluviométrie (ampleur et répartition des précipitations), l'espèce cultivée, la nature du sol et la profondeur des drains (Billy, 2008). Une précipitation atmosphérique lente et régulière survenant à la surface des parcelles produit tout d'abord un déplacement de la solution du

sol, suivi progressivement du mélange d'eau et de phase liquide du sol. Il en résulte que les eaux de drainage sont des solutions plus diluées que les solutions du sol. Selon les cas, le rapport drainage/pluviométrie s'établit entre 20 et 50% (Tournebize et al. 2008).

1.5.1.2. Les émissions de protoxyde d'azote

En chambres d'accumulation des gaz.

L'accumulation passive du gaz dégagé sous des enceintes placées en surface du sol est la méthode la plus fréquemment utilisée pour déterminer les émissions de gaz à effet de serre, dont notamment le N_2O . Cette méthode est décrite en détail dans Hutchinson et Livingston (1993). Une chambre consiste en un cylindre ou un cube ouvert sur deux faces avec un couvercle détachable au-dessus. La surface couverte est variable, de 0.01 à 1 m². Cette chambre est enfoncée dans le sol, et fermée hermétiquement pendant la durée de la mesure. Le gaz émis (ou consommé) par le sol s'accumule à l'intérieur de la chambre et est analysé en fonction du temps.

On distingue deux types de chambres: les systèmes fermés pour lesquels il n'y a pas de remplacement de l'air et les systèmes ouverts pour lesquels on impose un flux d'air constant à l'intérieur de la chambre (enceintes dynamiques). Dans les systèmes fermés, les flux sont calculés à partir de la cinétique d'augmentation de la concentration au cours du temps. Dans les systèmes ouverts, le flux est communément estimé à partir de l'écart de concentration entre l'air entrant et sortant de la chambre (Reichman et Rolston 2002).

La chromatographie en phase gazeuse (CPG) couplée à un détecteur d'électrons (ECD) est la méthode la plus couramment utilisée pour l'analyse de gaz et permet une mesure fine du N_2O . La principale caractéristique du détecteur ECD est sa grande sélectivité basée sur les coefficients d'absorption électronique des différents composés qui traversent ce détecteur. Les chambres au sol sont encore souvent utilisées dans une configuration manuelle, ce qui a comme désavantage principal de ne permettre que des mesures ponctuelles dans le temps, pas toujours suffisamment représentatives des fluctuations rapides des émissions.

De manière à réduire le temps requis aux mesures sur le terrain et d'en affiner la résolution temporelle, des chambres automatiques ont été développées d'abord en CPG (Conrad et al. 1983) puis avec le développement d'analyseurs par spectrométrie infrarouge (Bessou et al. 2010) ce qui permet la réalisation de mesures en semi-continu.

Les méthodes micro-météorologiques.

Les approches micro-météorologiques sont basées sur les mesures des mouvements de masses d'air au-dessus des écosystèmes et ont l'avantage par rapport aux chambres de réaliser une intégration spatiale et temporelle des flux de N_2O (Fowler et al. 2001). Les capteurs montés sur des tours détectent les mouvements et le contenu en gaz de l'air au-dessus et à l'intérieur de la canopée; l'intensité et la direction du flux peuvent être déterminés par les concentrations de gaz, les gradients et la turbulence.

Le développement des détecteurs à diode laser et à laser à cascade quantique a ainsi permis le développement de ces méthodes et de leurs applications sur divers écosystèmes terrestres (Wang et al. 2013). Trois types de méthodes existent selon les échelles considérées. La méthode « eddy covariance » relève de la mesure directe dans l'atmosphère de la vitesse du vent vertical et de la concentration en N_2O . Cette méthode permet l'étude à l'échelle de la

parcelle et a souvent été utilisée en parallèle des méthodes de chambres automatiques (Laville et al. 1999), conduisant à des estimations comparables. La méthode par gradient de flux permet de couvrir des parcelles agricoles. Cette méthode implique de réaliser des mesures de la vitesse du vent, de la température et de la concentration en N_2O . Cette méthode est généralement associée à l'équipement d'une tour de mesure, dont la hauteur définit la zone de couverture. Enfin la méthode « relaxed eddy accumulation » permet de couvrir efficacement des zones agricoles de plusieurs km^2 , notamment à l'aide d'appareils de mesures aéroportées (Pattey et al. 2007). Cette méthode bien qu'intégrant une échelle spatiale importante sont encore souvent peu précises

1.5.2. Les méthodes indirectes

Plusieurs méthodes indirectes existent également pour la quantification des fuites d'azote, comme par exemple les bilans azotés particulièrement utiles pour la quantification de la lixiviation, les équations de facteurs d'émissions de N_2O lors des apports d'engrais, les méthodes isotopiques (^{15}N) et les modèles de prédictions.

1.5.2.1. Les bilans azotés

A l'échelle de la rotation, le bilan d'azote permet d'estimer les fuites d'azote par le solde des apports d'azote au sol (engrais, produits résiduaux organiques, fixation symbiotique et le dépôt atmosphérique) et des exportations d'azote par la récolte et le pâturage. Cette méthode s'appuie sur l'hypothèse d'un régime permanent, où ce solde encore appelé surplus ou balance azotée du sol est un bon indicateur du potentiel des fuites environnementales des terres arables (Scoones et Toulmin 1998; Oenema et al. 2003; de Vries et al. 2011). Billen et al. (2013a) ont montré que 75-100% du surplus calculé était perdu sous formes de NO_3^- dans les zones sous-racinaires des parcelles conventionnelles (Figure 1-5). Toutefois les cultures avec CIPAN et les prairies permanentes peuvent limiter la lixiviation en dessous d'un surplus plafond de 60 à 120 $kg\ N\ ha^{-1}\ an^{-1}$ respectivement. Le surplus qui n'est pas lixivié peut être stocké dans le sol sous forme de matière organique, ou émis dans l'atmosphère lors des processus de dénitrification (N_2 , N_2O) et de volatilisation de l'ammoniac (Beek et al. 2003).

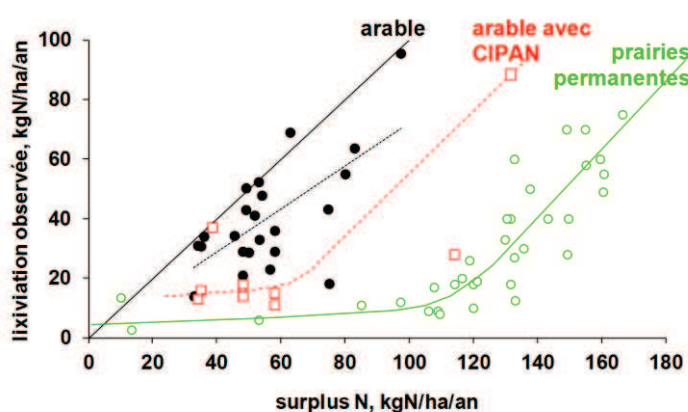


Figure 1-5. Relation empirique observée entre le surplus et la lixiviation en dessous de la zone racinaire dans les terres arables, avec ou sans CIPAN, et les prairies permanentes pâturées (Billen et al. 2013a).

1.5.2.2. Les facteurs d'émission

Les lignes directrices de l'IPCC (2006) décrivent une méthodologie complète afin d'estimer les émissions de N₂O liées à l'agriculture à partir de facteurs d'émission. Concernant les émissions directes provenant des sols, plusieurs études ont permis d'estimer à 0.9 % le facteur d'émission lié à la fertilisation apportée (Bouwman 1996; Stehfest et Bouwman 2006). Le facteur d'émission (EF en %) fait référence à la différence de N₂O émis entre une parcelle fertilisée et une parcelle non fertilisée, par rapport à la quantité totale d'azote apportée (Equation 1-6).

$$\text{Equation 1-6. EF} = \frac{\text{N}_2\text{O-N (F)} - \text{N}_2\text{O-N (C)}}{\text{N apporté}}$$

où N₂O-N (F) et N₂O-N (C) sont les émissions de N₂O cumulées sur l'année (kg N ha⁻¹ an⁻¹) sur la période fertilisée (F) et le contrôle (C) non fertilisé, respectivement ; N apporté est la quantité d'azote sous forme d'engrais appliquée pendant l'étude (kg N ha⁻¹ an⁻¹). Par défaut, le pourcentage recommandé par l'IPCC est de 1%, comprenant une certaine variabilité locale (Lesschen et al. 2011). Toutefois pour les engrais organiques, n'ayant pas tous la même vitesse de minéralisation, il est préférable d'utiliser la quantité d'azote « disponible » plutôt qu'apporté, ce qui augmenterait l'estimation de EF par l'IPCC (2006), sans sous-estimer leur effet (Aguilera et al. 2013).

Les émissions indirectes associées à la volatilisation, au lessivage et à la consommation finale des aliments doivent aussi être estimées pour établir un bilan complet des fuites d'azote. Les émissions indirectes dues au lessivage de l'azote, sont reconnues comme les plus importantes. Ce facteur d'émission (EF₅) correspond à 0.75% de l'azote apporté sous forme d'engrais minéral et organique. Ces valeurs comprennent une grande incertitude tout au long des étapes de cheminement de l'azote dans les nappes, les rivières jusqu'à l'estuaire et sont parfois remises en cause (Vilain et al. 2012c).

1.5.2.3. Les méthodes isotopiques (¹⁵N)

Pour mesurer les fuites d'azote dans l'environnement, la méthode isotopique consiste dans le suivi du ¹⁵N, l'isotope lourd et stable de l'azote, qui peut être utilisé au champ comme au laboratoire.

Sur le terrain, il est possible, par la méthode du marquage isotopique de suivre l'apport d'un produit artificiellement enrichi en un isotope donné alors qu'il naturellement peu représenté (Recous et al. 1988) ou par la méthode du traçage isotopique naturel de déterminer l'abondance naturelle d'un isotope dans un compartiment donné comme une signature des processus qui l'affectent, sachant que certains processus comme la volatilisation ou la dénitrification sont fractionnants, c'est-à-dire qu'ils affectent proportionnellement plus l'isotope léger ¹⁴N que l'isotope lourd ¹⁵N, enrichissant ainsi légèrement le produit de départ en ce dernier (Sebilio et al. 2003).

Toutefois ces approches présentent plusieurs limites et en premier lieu elles nécessitent de disposer d'un spectromètre de masse à haute sensibilité. Le marquage oblige en outre à travailler avec une source d'azote exogène dont il faut évaluer la dilution dans le pool de NO_3^- du sol. Le coût du marqué ^{15}N et la nécessité de travailler avec un produit fortement marqué conduisent à opérer sur des surfaces restreintes. Enfin, la nécessité de couvrir longuement le sol pour quantifier les faibles flux limite fortement l'usage de la méthode en présence de végétation (Hénault et al. 2012).

1.5.2.4. *Les modèles de prédictions des fuites azotées*

Plusieurs modèles ont été développés afin de permettre une estimation des fuites d'azote dans l'environnement. Les modèles ont pour objectif de calculer les flux ou les quantités d'azote mis en jeu à partir de lois ou de relations préétablies par des approches mécanistes, empiriques ou mixtes. Modéliser les processus implique nécessairement de simplifier et de conceptualiser la multitude de processus physiques, chimiques et biologiques qui existent dans la nature. Pour chaque modèle, on s'attache donc en général à caractériser un fonctionnement de façon conceptuelle, en formalisant ou en utilisant des équations que l'on peut paramétrer. Une fois développés et validés par des observations *in situ*, ces outils permettent en explorant une multitude de contraintes, de limiter les campagnes de terrain.

Concernant la modélisation à l'échelle de la parcelle, les modèles STICS (Brisson et al. 2003) ou CERES-EGC (Jones et al. 1986) sont des modèles agronomiques qui permettent de définir l'impact des pratiques culturales sur les fuites d'azote (N_2O , NO_3^-). Ils prennent en compte la croissance des plantes, le bilan d'eau dans les sols, et les cycles de l'azote et du carbone. D'autres modèles dérivés des précédents, plus simples, calculent uniquement le transport vertical de l'eau et des nutriments dans le profil de sols, comme LEACHN (Jemison et al. 1994) ou LIXIM (Mary et al. 1999). Des modules ont été développés pour quantifier les émissions de N_2O (Frolking et al. 1998; Chen et al. 2008), comme NOE (Hénault et al. 2005), NLOSS (Riley et Matson 2000), intégrés ou pas dans les modèles agronomiques.

La majeure partie des modèles s'appliquent à l'échelle de la parcelle, mais certains modèles permettent également d'aller au-delà de l'exploitation agricole (Olesen et al. 2002), jusqu'à l'échelle régionale, comme SWAT (Bioteau et al. 2002) ou de l'Europe, comme CAPRI-DNDC (Leip et al. 2008).

Toutefois le recours à la modélisation pour décrire des situations agricoles réelles (et non expérimentales), soulève la question du nombre de paramètres à prendre en compte par le modèle (Hansen et al. 2001; Meynard et al. 2003). En particulier, certaines cultures spécifiques de l'AB (par exemple les légumineuses) ne sont pas encore paramétrées dans les modèles classiques, comme dans le modèle STICS qui ne peut être appliqué de façon satisfaisante que dans les systèmes conventionnels (Beaudoin et al. 2008).

2

Localisation des sites, matériels et méthodes

2.1. Localisation et caractéristiques des sites d'études

L'ensemble de nos sites d'études sont situés au centre du bassin de la Seine, au niveau de la région IDF et ses environs. Le bassin de la Seine fait l'objet de recherches approfondies depuis 1989 avec la création du programme interdisciplinaire en environnement (PIREN-Seine). Le bassin versant est une zone géographique privilégiée pour réaliser des études sur les flux d'eau en relation avec les éléments biogènes issus des pratiques agricoles. Un bassin versant est une aire délimitée par des lignes de partage des eaux (très souvent ligne de crête), à l'intérieur de laquelle toutes les eaux tombées alimentent un même exutoire. Les caractéristiques du bassin de la Seine et d'un de ses sous bassin, le bassin de l'Orgeval vont être décrites ainsi que les exploitations agricoles qui ont été suivies.

2.1.1. Le bassin de la Seine

Le bassin de la Seine (78 600 km²), soit 12% du territoire national, concentre 25% de la population française et est parcouru par 23 000 km de cours d'eau au travers desquels s'écoulent chaque année 14 milliard de m³ d'eau. Les densités de population les plus importantes sont rencontrées en aval, avec l'agglomération parisienne, qui comporte 60% de la population du bassin (10.5 millions d'habitants en 2010), tandis que les activités agricoles sont localisées majoritairement en amont, recouvrant 46% du bassin. Le réseau hydrographique de la Seine comprend les affluents principaux de la Marne, l'Oise, l'Eure outre la Seine amont. Le bassin de la Seine est sédimentaire, constitué de deux formations calcaires jurassiques.

2.1.1.1. Le contexte climatique

La pluviométrie du bassin est due à un apport assez constant d'humidité par les vents d'ouest issus de l'Océan Atlantique (climat océanique pluvial). Le bassin de la Seine possède une pluviométrie moyenne de 745 mm (1970-2004) comprenant un gradient de 620 mm dans la Beauce jusqu'à 1100 dans le Morvan et sur les reliefs sud-est du bassin (**Figure 2-1, a**). Les plateaux du centre du bassin sont moins arrosés car les vents d'ouest humides n'y rencontrent pas d'obstacle ; les altitudes maximales ne dépassant pas les 200 m en région IDF.

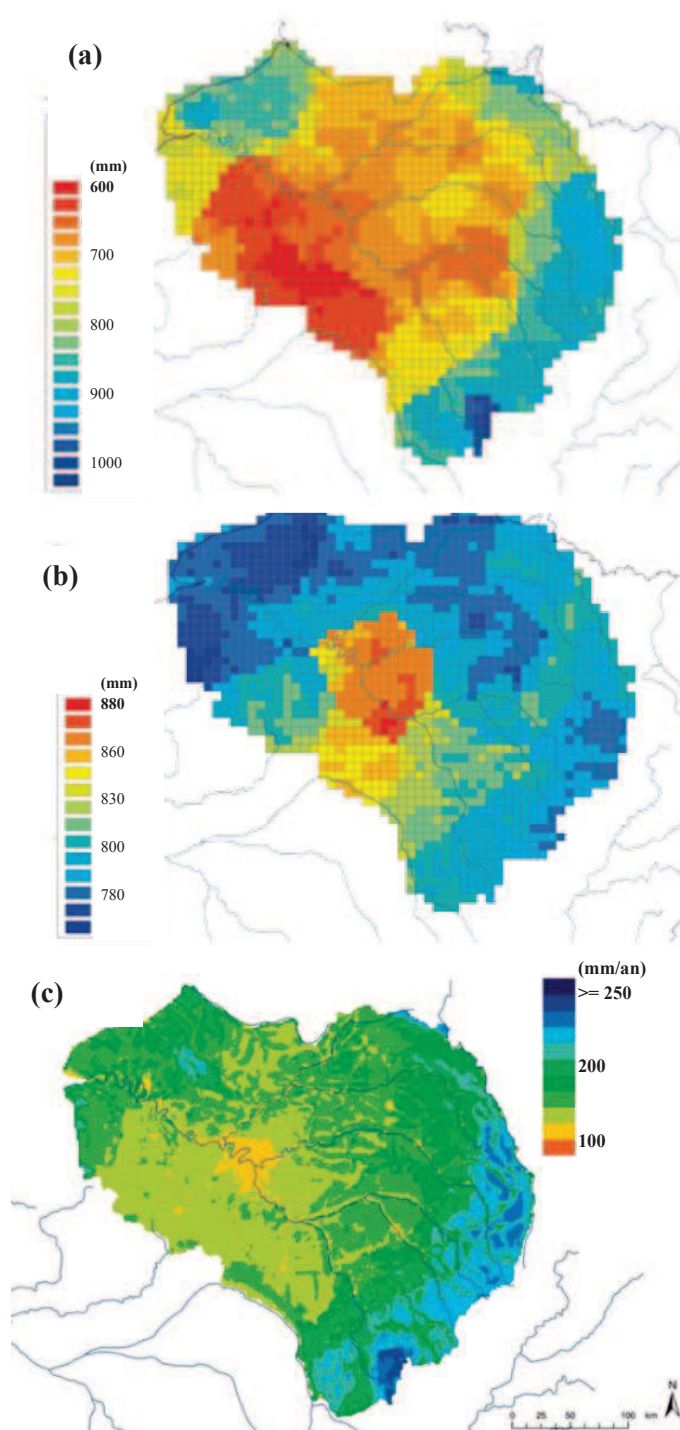


Figure 2-1. Pluviométrie moyenne annuelle (a), ETP (b) et lame d'eau potentiellement infiltrée (c) dans le bassin de la Seine (Viennot et al, 2011, Fascicule PIREN-Seine, http://www.sisyphe.upmc.fr/piren/?q=webfm_send/817).

L'évapotranspiration potentielle (ETP) moyenne annuelle est de 759 mm (1970-2004). Elle varie entre 670 mm en région parisienne jusqu'à 750 mm en Seine Maritime (**Figure 2-1, b**). L'ETP correspond à la quantité d'eau susceptible d'être évaporée par une surface d'eau libre. Dans la réalité, la quantité d'eau réellement évaporée (ETR) est donc inférieure à l'ETP. Sur le bassin, elle est voisine de 600 mm.

Il en découle, que la lame d'eau potentiellement infiltrée dans les sols, est en moyenne de 154 mm par an sur le bassin, avec des valeurs inférieures à 100 mm en Beauce et supérieure à 200 mm en Meuse et Côte d'Or (**Figure 2-1, c**) en relation avec la nature des sols (cf. **Figure 2-4**). On constate que la lame d'eau est plus importante dans les régions à forte pluviométrie, au socle calcaire et à forte activité agricole (sud est du bassin).

2.1.1.2. Un bassin majoritairement céréalier

Le bassin de la Seine se compose en 2000 de 52% de terres arables, 12% de prairies permanentes, 24% de surfaces forestières, ainsi que 6% de surface urbanisée et 6% d'autres occupations du sol. Il compte alors 15% des exploitations françaises et 23% de la SAU du territoire national.

Le bassin de la Seine est dominé par des exploitations de grandes cultures, avec une taille moyenne de 68 ha, contre 42 ha en moyenne nationale.

Les cultures majoritaires sont le blé, l'orge, le maïs et le colza qui recouvrent plus de 60% SAU pour quasiment l'ensemble des départements du bassin de la Seine ([Agreste 2010](#)). Le blé tendre d'hiver représente parfois plus du tiers de l'assolement des exploitations. La succession colza-blé-orge est très répandue sur tout l'est du bassin de la Seine (classe 14 et 15, **Figure 2-2**). Au Nord de Paris, les betteraves et les pommes de terre alternent avec du blé (classe 1 à 3, **Figure 2-2**).

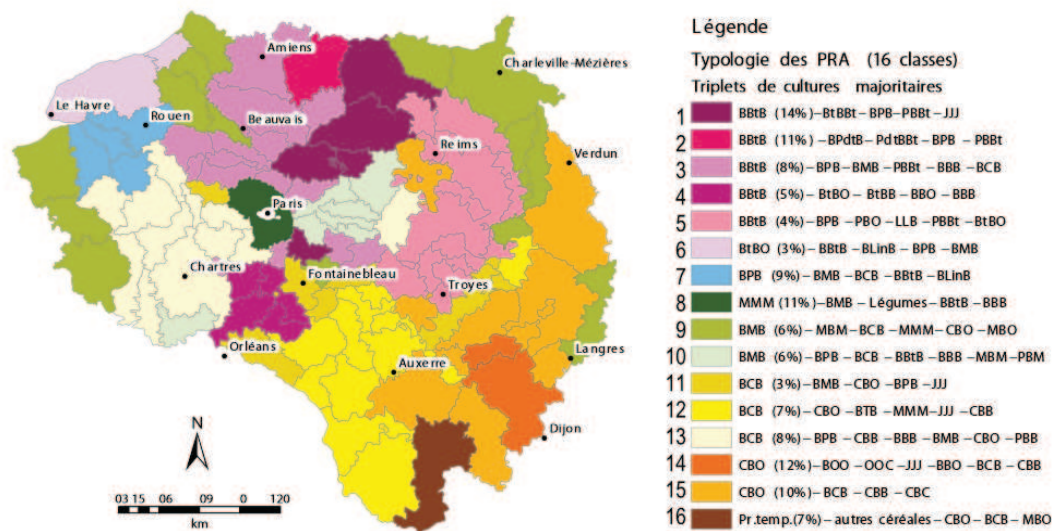


Figure 2-2. Typologie des petites régions agricoles (PRA) du bassin de la Seine en fonction des principaux triplets de cultures observés entre 1992 et 2003. Abréviations : M = Maïs, B = Blé, O = Orge, T = Tournesol, L = Luzerne, J = Jachère, Bt = betterave, C = Colza, Pdt = Pomme de terre, P = Pois, L = lin textile dans le fascicule du PIREN-Seine Agriculture ([Schott et al, 2011](#)).

La moyenne des doses d'engrais apportées est de 145 kg N ha⁻¹ d'azote minéral et ces doses peuvent atteindre 170 kg N ha⁻¹ pour le blé, le colza et le maïs grain. Certaines cultures, comme le maïs fourrage et la betterave, reçoivent des apports supplémentaires en engrais organique. En ce qui concerne l'utilisation des pesticides, l'itinéraire technique des céréales comporte en moyenne deux désherbants, trois fongicides, un insecticide et un régulateur de croissance (Rolland et al. 2006).

Le rendement moyen national est de 78 q ha⁻¹ l'un des plus élevés du monde, avec une fertilisation minérale sur plus de 85% SAU de grandes cultures (Agreste, 2012). Le bassin parisien produit 25 fois plus de céréales que Paris n'en consomme et les exporte, tant vers des régions, qui se sont spécialisées dans l'élevage, que hors du territoire national (Billen 2009a). En revanche, les rendements moyens en AB sont de 40 q ha⁻¹ en blé, 37 q ha⁻¹ en triticales, 30 q ha⁻¹ en orge de printemps, 32 q ha⁻¹ en avoine, 63 q ha⁻¹ en maïs, 26-29 q ha⁻¹ en féverole, 21 q ha⁻¹ en pois, 105 q ha⁻¹ en luzerne, 16 q ha⁻¹ en colza et 22 q ha⁻¹ en tournesol (moyenne pluriannuelle 2005-2007, Agence Bio 2010). Le maintien d'un différentiel de prix élevé en faveur de l'AB (près de 3 fois le prix conventionnel) ainsi que les plus faibles dépenses en intrants compensent régulièrement la baisse de rendement observée en AB, soit en général un prix autour de 300 € t⁻¹ en céréales et 200 € t⁻¹ en colza.

2.1.1.3. Les systèmes biologiques en région Ile-de-France

Dès la fin des années 1990, face à l'émergence de la demande en céréales biologiques, des exploitations sans élevage en France et en région IDF, se sont converties en AB (Allard et al. 2000). Ces exploitations bio sans élevage gardent beaucoup de points communs avec les exploitations de grande culture conventionnelles caractérisées par de nombreuses interventions culturales et la prédominance du blé (30% des assolements) (Lamine et Bellon 2009).

La région IDF est une région largement agricole avec 570 870 ha de SAU soit 50% de sa surface), dont 94% en grandes cultures (Agreste, 2010). En 2013, les surfaces en AB représentent 8 926 ha, soit 1.56% de la SAU francilienne et 3.7% des exploitations (GAB IDF, 2013). Les producteurs biologiques franciliens sont majoritairement des maraîchers (45%) suivis par les stricts céréaliers (30%). Toutefois en raison de la petite taille des exploitations maraîchères (quelques ha), les grandes cultures biologiques occupent la majorité des surfaces agricoles franciliennes, avec 84% de la SAU en AB. Les exploitations franciliennes spécialisées en grandes cultures biologiques sont converties majoritairement (70%) en totalité. Sinon, dans les exploitations mixtes (AB et AC), la part des surfaces consacrées à l'AB est en moyenne de 61% (Petit, 2013). Toutefois depuis novembre 2013, la mixité au sein d'une exploitation est possible seulement si les variétés biologiques et non biologiques sont facilement distinguables (Article 11 modifié, CE n°834/2007). Quatre dérogations peuvent être appliquées pour les pâtures, les cultures pérennes, la production de semences ou pour l'enseignement et la recherche. La mixité est perçue comme un critère de risque, listé dans le plan de contrôle cadre de l'AB, ce qui implique une plus grande fréquence des contrôles inopinés ainsi que des analyses plus importante dans ce type d'exploitation.

On distingue deux grands types de rotation dans la région, avec pour chacune, des variantes liées à des contextes pédoclimatiques particuliers : des rotations longues (8 ans en moyenne) reposant sur la luzerne, et des rotations courtes (4-6 ans) basées sur des légumineuses à graines. Environ 50% des céréaliers bio franciliens pratiquent des rotations avec luzerne, celle-ci étant surtout commercialisée ([Programme RotAB, 2011](#)). La rotation type avec luzerne peut être plus ou moins longue suivant les cas, avec éventuellement (cf culture entre parenthèses) l'introduction de céréales secondaires : 2 à 3 ans de luzerne, blé, céréale secondaire, (céréale secondaire ou oléagineux), légumineuse, blé, (céréale secondaire), légumineuse, blé, (orge).

Les rotations sans luzerne sont pratiquées par l'autre moitié des céréaliers AB, qui pour des raisons de commercialisation, d'équipement et/ou de type de sol, ne peuvent pas cultiver de luzerne. La rotation type, alors basée sur l'introduction de légumineuses à graines (trèfle, féverolle, pois), est généralement conduite sur 6 ans, mais elle peut être plus ou moins longue suivant les cas (cf. culture entre parenthèses) : légumineuse, blé, céréales secondaires ou oléagineux (tournesol, colza), légumineuse, blé, (céréale secondaire) ([Programme RotAB, 2011](#)).

En guise de précision, les cultures peuvent être semées en tant que culture de printemps associées à des cultures pièges à nitrate (CIPAN) ou en tant que cultures d'hiver (Figure 2-3).

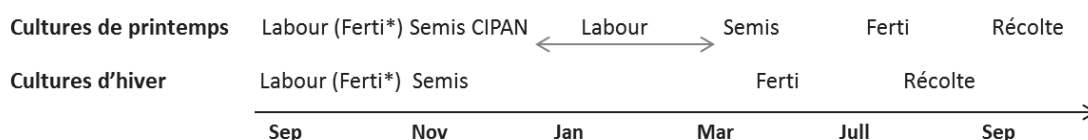


Figure 2-3. Temporalité des principaux travaux dans les champs pour les cultures de printemps ou les cultures d'hiver dans les systèmes de grandes cultures du bassin de la Seine (France). Les abréviations Ferti et Ferti* correspondent à la fertilisation minérale et organique respectivement.

2.1.2. Le réseau ABAC

Les exploitations agricoles que nous avons sélectionnées suivent quatre conditions :

- Elles sont localisées dans le bassin de la Seine (région IDF et périphérie)
- Les exploitants ont donné leur accord pour réaliser le suivi en azote de leurs parcelles, et participent éventuellement aux prélèvements
- La profondeur des sols est supérieure ou égale à 90 cm
- Les rotations sont constituées principalement de grandes cultures céréalières

2.1.2.1. Localisation des exploitations

Le réseau ABAC a débuté avec l'année culturale 2012-13 dans huit systèmes dans trois pôles pédologiques : (1) Seine & Marne (S&M), (2) Oise et (3) Yonne (**Figure 2-4 a**). Au total, huit systèmes de cultures ont été étudiés dont cinq en AB et trois en AC, ce qui correspond à 30 parcelles en AB et huit parcelles en AC du fait des rotations plus courtes en AC (cf. **Chapitre 6**).

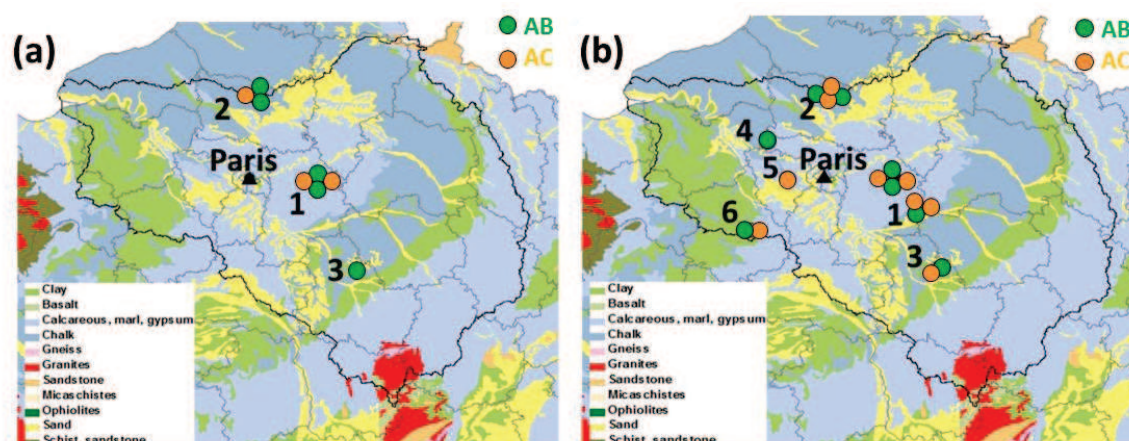


Figure 2-4. Carte lithologique du bassin de la Seine avec les exploitations agricoles équipées en 2012-13 (gauche) et 2013-14 (droite).

En 2013-14, le réseau ABAC s'est agrandi avec un total dix-huit systèmes de cultures qui ont été suivies dans six pôles pédoclimatiques différents : (1) En Brie (S&M) où huit exploitations ont été équipées, dont quatre systèmes en AB et cinq en AC ; (2) Sur le plateau Picard (Oise) avec deux systèmes en AB et deux en AC ; (3) Dans le pays d'Othe (Yonne), entre la Champagne crayeuse et la Champagne humide, avec un système en AB et un système en AC ; (4) Dans le Vexin (Val d'Oise), dans un essai biologique ; (5) Dans le Hurepoix (Yvelines) dans une exploitation expérimentale en AC ; (6) En Beauce (Centre), un essai d'un lycée agricole, équipé sur 3 parcelles en AB et une en AC (**Figure 2-4 b**). Soit un total de 49 parcelles AB et 30 en AC (cf. **Chapitre 7**). La présence conjointe de systèmes AB et AC dans chaque pôle pédoclimatique permet de mieux appréhender l'impact des systèmes AB dans des conditions pédologiques et climatiques spécifiques.

2.1.2.2. Les systèmes de cultures du réseau ABAC

De manière représentative de la région, les rotations étudiées en AB possèdent des rotations de 6-9 ans avec en tête de rotation, deux à trois années de luzerne, généralement exportée.

Pour rappel, la Directive Nitrate a caractérisé les engrais selon trois catégories : les fertilisants de type I, sont ceux qui contiennent de l'azote organique à vitesse de minéralisation lente et qui ont un rapport carbone sur azote supérieur à 8 (exemple : fumier de bovins), aussi appelés amendements ; les fertilisants de type II qui contiennent de l'azote organique à vitesse de minéralisation rapide et ont un rapport carbone sur azote inférieur ou égal à 8 (exemple : lisier de bovins) ; et les fertilisants de type III qui sont les fertilisants minéraux de synthèse (ammonitrate, urée ou des mélanges).

Les rotations en agriculture biologiques

Contrairement à l'azote minéral, les quantités d'azote dans les engrais organiques varient selon leurs types. Afin de pouvoir estimer les quantités d'azote apporté, des analyses ont été

effectuées par la chambre d'Agriculture de Seine-et-Marne ou par les agriculteurs. Au total six différents types d'engrais organiques ont été utilisés dans les exploitations étudiées, comportant des teneurs en azote et des C/N spécifiques (**Table 2-1**). Les doses appliquées en engrais organiques sont en moyenne de $33 \pm 6 \text{ kg N ha}^{-1} \text{ an}^{-1}$. En AB, le non-labour est très rare, car le labour permet de « nettoyer » les parcelles des adventices en l'absence de traitement herbicide.

Certaines exploitations pour être totalement autonomes n'utilisent que des légumineuses comme apport d'azote.

Table 2-1. Quantité d'azote contenu dans les engrais organiques couramment utilisés dans le réseau ABAC

Type d'engrais organique	Teneur en azote dans matière brute (g/kg)	C/N total	Année d'analyse
vinasse de betterave	17.0	8.1	2013
fiente de volaille	43.1	7.1	2012
fumier cheval	7.1	8.9	2004
fumier de bovin	7.3	16.7	2005
digestat de méthanisation	13.1	11.8	2014
Lisier de bovins	4.5	10.2	2013

Les rotations en agriculture conventionnelle

Les rotations AC sont plus courtes 2-5 ans avec une plus faible présence de légumineuse. La proportion des cultures suivies dans le réseau ABAC (2013-14) est proche de celle que l'on retrouve dans les différents départements du bassin de la Seine étudiés, soit 50% de l'assolement est dédié à la culture du blé, 14% à l'orge/escourgeon, 18% au maïs, 14% aux oléagineux et 4% aux protéagineux (**Figure 2-5**).

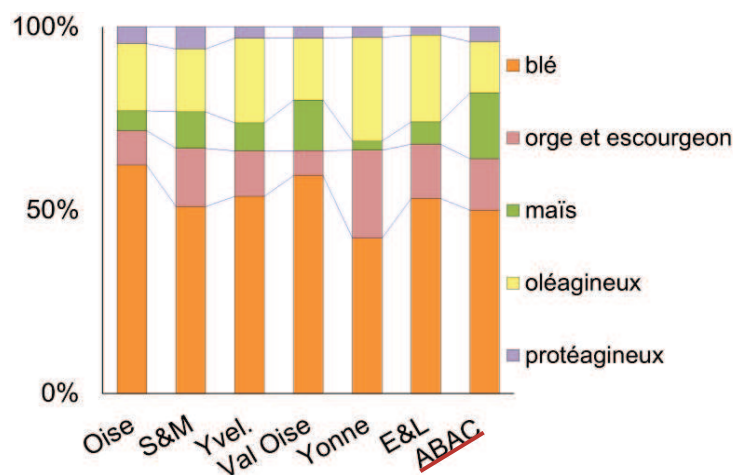


Figure 2-5. Proportion des cultures dans les six départements (Recensement Agreste 2012) et au sein du réseau ABAC.

La gestion des intercultures (variétés et durée d'implantation) varient selon les exploitations. Les techniques culturales simplifiées (TCS) et le non-labour sont pratiqués dans quelques exploitations. Les apports minéraux d'azote, effectués au printemps, dans les exploitations sont en moyenne de $150 \pm 46 \text{ kg N ha}^{-1} \text{ an}^{-1}$.

Toutes les caractéristiques physico-chimiques et hydrologiques des sols des exploitations AB et AC sont synthétisées dans l'Annexe 3.

2.1.3. Le bassin de l'Orgeval

2.1.3.1. Contextes pédologique et climatique

Le bassin d'Orgeval est situé dans le bassin de la Seine, à 70km à l'Est de Paris, en S&M (**Figure 2-6**). C'est un bassin de référence où travaillent conjointement des équipes de recherche (IRSTEA, UPMC, CNRS, INRA, etc.). Ce bassin versant d'une superficie de 104 km^2 est occupé à 81% par des cultures, 7 % par des prairies permanentes, 11% par des forêts et 1% par des zones urbaines ou des routes.

Les conditions climatiques moyennes sur le bassin versant d'Orgeval correspondent à une pluviométrie de 677 mm, une ETP de 782 mm et une température de 11°C (mesures sur 30 ans à Boissy-le-Châtel, $48^\circ49'15''\text{N}$ $3^\circ08'19''\text{E}$). Les aquifères de Brie et de Champigny sont les aquifères les plus proches de la surface. Le sol présente une texture limoneuse-argileuse, avec des couches d'argile en profondeur qui lui confère un comportement hydromorphe. Pour pallier à l'engorgement, des sols les parcelles cultivées sont drainées (90% de la SAU) drainées à 90 cm de profondeur (Tournebize et al, 2011). Ce sol est un luvisol glossique hydromorphe, selon la référence pédologique régionale de l'INRA-Orléans.

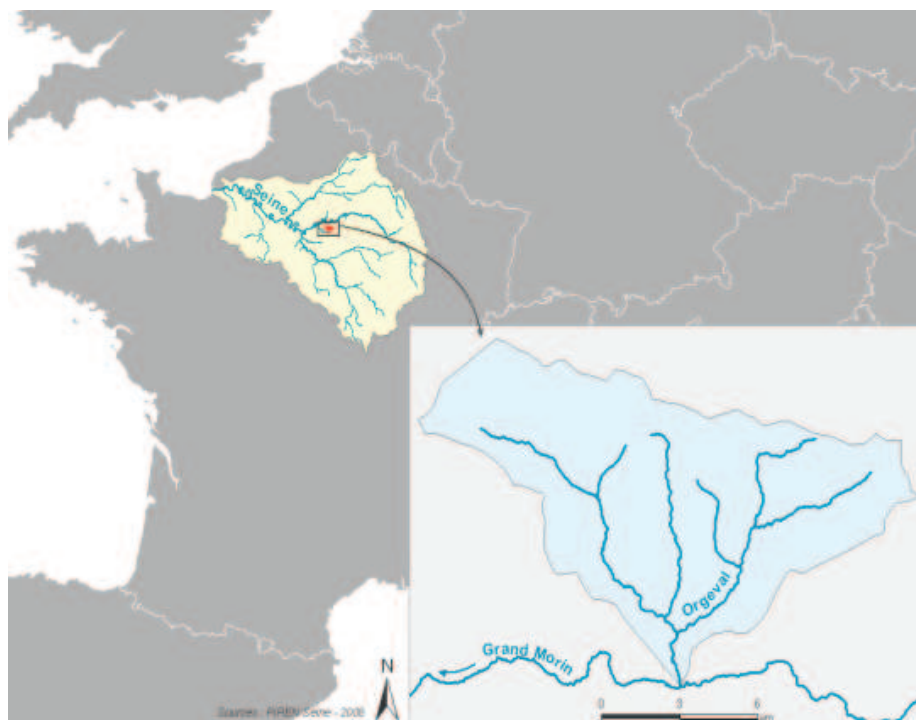


Figure 2-6. Localisation du bassin versant de l'Orgeval dans le bassin de la Seine.

2.1.3.2. Les pratiques agricoles

Dans le bassin de l'Orgeval, la polyculture-élevage a été remplacée par une production intensive de céréales, avec comme cultures principales : blé, maïs, betterave, féverole/pois sur des exploitations de 133 ha en moyenne (Nicola et al, 2011).

En Seine-et-Marne, l'AB ne recouvre que 1.3% SAU et elle est largement tournée vers les grandes cultures (86%), mais elle correspond à 50% de la SAU des conversions AB franciliennes (INSEE – Agence Bio 2012).

Le site particulièrement étudié est une exploitation agricole mixte (AB et AC) de grandes cultures céréalière de 280 ha, située proche du ru de Bourgogne, à Aulnoy (N 48° 50' 49.888" E 3°5'21.134"). Pour notre étude, nous avons choisi une exploitation agricole mixte (AB/AC) à responsabilité limitée (EARL). Sur les 280 ha de cette exploitation, un quart de la surface (70 ha) a été convertie en AB en 2009. La rotation en AB se réalise sur huit ans, et celle en AC sur trois ans.

Nous avons équipé cinq parcelles en AB et une en AC en bougies poreuses pour la mesure de la lixiviation, et deux parcelles (AC et AB) en chambres d'accumulation manuelles et automatiques pour quantifier les émissions de N₂O, ainsi que deux autres parcelles en chambres manuelles seulement. Nous avons installé sur l'exploitation deux stations météo (Campbell) permettant d'enregistrer les données météorologiques, comprenant un pluviomètre, des capteurs de température et d'humidité (air et sol), ainsi que les radiations solaires.

En équipant plusieurs parcelles de l'assolement simultanément, on peut suivre la rotation de l'agriculteur en un temps plus court que la durée de la rotation. Ainsi au cours de ces trois années de mesures, treize cultures en AB et cinq en AC ont été suivies concernant la lixiviation et cinq cultures en AB et trois en AC pour les chambres d'accumulation de GES (cf. *Chapitre 3*).

2.2. Matériels et méthode

2.2.1. Les propriétés physico-chimiques des sols

Quelques soient les investigations sur le terrain, (lors de la pose des bougies, des mesures en chambres d'accumulation de gaz et des expériences en laboratoire) des triplicats de sol (30 g) sont utilisés pour déterminer l'humidité et la teneur en matière organique du sol. Les échantillons sont placés à 105°C (48h) et 450°C (4h) respectivement et pesés à chaque étape. Les concentrations en azote minéral du sol (NO₃⁻, NO₂⁻, NH₄⁺) sont également déterminées après extraction au KCl, en plaçant 5 g de sol dans 20 ml de KCl (2 M) pendant 2 h sur une table agitante. Les suspensions sont centrifugées pendant 10 min à 3000 tours par minute et le surnageant est congelé à -18°C jusqu'à l'analyse avec un auto-analyseur (Quatro, Bran & Luebbe, ou Gallery, Thermofisher). Des mesures simultanées avec les deux appareils montrent une pente et un coefficient de corrélation proche de 1.

Les concentrations d'azote dans les reliquats ont été mesurées en mg N l⁻¹ puis converties en kg N ha⁻¹ via la relation de Sterckeman and Ciesielski (1991) (Equation 2-1).

$$\text{Equation 2-1. } N \text{ kg/ha sec} = \frac{1000 * V + M * h}{M * (1000 - h)} * \frac{BD * t}{10} * N \text{ mg/l}$$

où V est le volume d'extraction (20 ml) ; M, la masse de sol frais (5 g) ; h, l'humidité massique ; BD, la densité apparente du sol ; t, l'épaisseur (cm)

Le taux de remplissage en eau des pores du sol (ou water filled pore-space en anglais, WFPS) est une mesure utile pour appréhender l'influence de l'humidité sur les processus biologique car il intègre l'aération du sol et la disponibilité de l'eau (**Equation 2.2**).

$$\text{Equation 2-2. \%WFPS} = \frac{(h \times BD \times 100)}{1 - \frac{BD}{Dp}}$$

avec h : l'humidité massique

BD : la densité apparente du sol

Dp : la densité des particules de 2.65g.cm⁻³.

Enfin l'ensemble des sols (sur les trois horizons) ont été lyophilisés pour déterminer la granulométrie, le carbone organique, l'azote total et le rapport C/N. Ces analyses ont été réalisées par le laboratoire analyses des sols d'Arras (LAS, INRA-Arras) (méthodologie en **Annexe 2**). Un total de 120 échantillons a été analysé en 2012-13 et 148 échantillons supplémentaires en 2013-14.

2.2.2. La lixiviation du nitrate

2.2.2.1. Protocole des bougies poreuses et analyse des concentrations

Nous avons retenu la méthode directe des bougies poreuses verticales pour estimer les fuites d'azote dans l'eau infiltrée, car elles se mettent en place rapidement, à faible coût d'investissement (main d'œuvre non comprise) et sans destruction des parcelles. En effet, une pose à la verticale nous a permis de ne pas engager d'importants travaux de terrassement dans les parcelles des agriculteurs, mais aussi d'effectuer des mesures peu après leur installation sans destruction de la structure du sol. Toutefois, cette disposition verticale impose le retrait des bougies avant le labour, ce qui rend difficile le suivi à long terme.

Dans chaque parcelle, six bougies poreuses (3 mm de diamètre, 85 cm de long) ont été installées à la tarière manuelle sur chaque terme de la rotation, soit un total de 234 bougies installées en 2012-13 et 570 bougies en 2013-14. Le diamètre de la tarière est égal à celui des bougies, ce qui évite les infiltrations. Les bougies sont disposées en ligne, parallèlement au travail du sol, à un minimum de 14 m de la bordure du champ et en dessous de la surface du sol (5 cm) pour ne pas gêner le travail des agriculteurs. Les bougies sont préalablement lavées avant d'être installées dans les parcelles.

Le dispositif de bougies poreuses permet de récupérer l'eau sous la zone racinaire (soit une profondeur de prélèvements d'eau d'environ 90cm), grâce à une dépression préalable (-800 mbar, pompe à vide électrique PAV 2000, SDEC) pendant une durée régulière de 48h avant les prélèvements. L'eau sous-racinaire est ensuite récoltée à l'aide d'un tube de prélèvement relié à un flacon de recueillement, lui-même relié à la pompe à vide (**Figure 2-7**). Les

prélèvements sont effectués une fois par semaine dès le début de la saison hydrologique puis tous les quinze jours, selon les épisodes pluvieux et la saturation en eau du sol. Lorsque les parcelles sont drainées, les sorties des drains des parcelles (90 cm) sont également prélevés pendant la période de drainage (plan du système de drainage en **Annexe 1**).

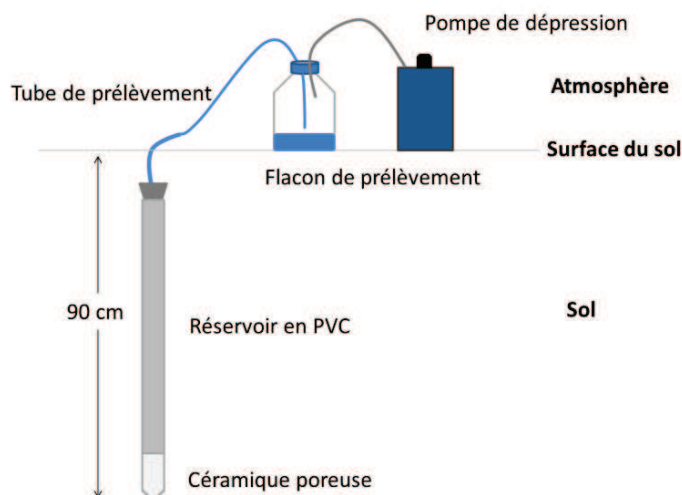


Figure 2-7. Schéma des instruments nécessaires pour le prélèvement d'eau sous-racinaire dans une bougie poreuse.

L'eau prélevée dans chaque bougie est congelée jusqu'à l'analyse des concentrations en azote inorganique (NO_3^- , NO_2^- , NH_4^+) à l'auto analyseur. La méthode utilisée pour mesurer l'ammonium est basée sur le réactif de l'indophénol bleu (Slawyk and MacIsaac 1972). Les nitrites et les nitrates sont mesurés avec la méthode au sulphanilamide (Jones 1984). Les nitrates sont déterminés après réduction en nitrite. Le calcul des concentrations en nitrate de l'eau de percolation a été pondéré en fonction des dates de prélèvements sur la période de mesures. Le premier prélèvement dans les bougies en début de saison de drainage n'a pas montré d'anomalie par rapport aux suivants.

2.2.2.2. Calcul de la lixiviation par un modèle hydrique

En raison de la lourdeur des déterminations permettant de connaître les propriétés de rétention en eau, les réserves utiles ont été déterminées par des classes de pédotransfert (Bruand et al. 1996 ; Wosten et al. 2001). En France, la fonction de pédotransfert établie dans l'Aisne par Jamagne et al. (1977) est toujours utilisée. Toutefois, sur les sols du bassin parisien, il a été montré que l'utilisation de celle-ci conduit à des biais d'estimation d'autant plus grands que les textures sont argileuses (Bruand et al. 2002). La classe de pédotransfert utilisée a donc été déterminée grâce à la base de données SOLHYDRO 1.0, qui rassemble 320 horizons prélevés principalement dans le bassin de Paris (Bruand et al. 2004).

Les transferts hydriques des solutés débutent quand l'humidité est égale à la capacité au champ, c'est-à-dire quand la réserve utile (RU) en eau du sol est remplie. La simulation de la lame drainante (W_i) a été réalisée via un modèle réservoir une couche (90 cm). La pluviométrie (P_i) et l'évapotranspiration potentielle (ETP_i) journalières ont été mesurées par

les stations météorologiques les plus proches. L'évapotranspiration maximale (ETM) correspond à l'ETP multiplié par un coefficient cultural compris entre 0.5 et 1. Pour la période hivernale, nous avons estimé un coefficient de 0.5 pour l'ensemble des parcelles étudiées, afin de ne pas sous-estimer les flux d'eau percolée (Perrier et al. 1980; Allen 2000). L'eau stockée journalièrement dans le sol correspond à la différence entre les entrées d'eau (Stock d'eau_{i-1}, P_i) et les flux sortants (ETM_i, W_{i-1}) (Equation 2-3). La lame drainante (W_i) correspond à l'excès d'eau qui ne peut pas être retenu par la réserve utile du sol, avec un coefficient de vitesse d'infiltration (k₂) compris entre 0.5 et 1 (Equation 2-4). Lorsque la capacité au champ est atteinte (RU_{cc}), la période de drainage commence avec un flux qui dépend des épisodes pluvieux.

La lixiviation est calculée en multipliant les concentrations pondérées par les lames drainantes entre deux prélèvements (Equation 2-5).

Equation 2-3. Stock d'eau (i) = max (Stock eau_{i-1} + P_i - ETM_i - W_{i-1}; 0)

Equation 2-4. Lame drainante W_i = max (k₂ * (Stock d'eau_i - RU_{cc}); 0)

Equation 2-5. Q = W_i * SRC * 0.01

avec Q : flux d'azote (kg N ha⁻¹)

W_i : flux d'eau écoulee pendant la période considérée (mm)

SRC : concentrations pondérées dans les bougies poreuses (mg N l⁻¹)

10⁻² : pour la conversion en kg N ha⁻¹

2.2.3. Les émissions de protoxyde d'azote

2.2.3.1. Protocole des chambres d'accumulation de gaz

Les mesures de GES sont effectuées en utilisant la technique des chambres d'accumulation statiques. Sur l'exploitation mixte, quatre séries de cinq chambres de base carrée (50 cm x 50 cm x 30 cm de hauteur) ont été implantées dans le sol à environ 15 cm, afin d'assurer l'étanchéité tout au long du développement de la culture. La stratégie avait été de placer une série de chambres manuelles sur les termes des rotations AB et AC, deux parcelles contigües. En mars 2013, des chambres automatiques ont également été installées sur ces mêmes parcelles, dans le cadre du réseau EFEMAIR-N₂O (financé par l'ADEME, coordonné par laboratoire Agro-Impact de l'INRA-Laon).

Le principe des chambres d'accumulation est de fermer hermétiquement un volume au-dessus d'une surface émettant ou consommant du gaz, de sorte que le gaz ne puisse pas s'échapper. Les chambres sont donc fermées à l'aide d'un couvercle hermétique équipé d'un septum permettant le prélèvement du gaz après accumulation. Lors des prélèvements manuels, quatre échantillons de gaz sont prélevés avec une seringue de 30ml dans chaque chambre (0, 10, 20, 30 minutes après fermeture, ou 0, 20, 40, 60 minutes lorsque le développement de la culture nécessite la mise en place de rehausses); les échantillons sont transférés dans un flacon préalablement mis sous vide de 12.5 ml (LabcoExetainer®) (Figure 2-8, a). Ces prélèvements sont effectués tous les quinze jours afin d'assurer un suivi régulier et d'estimer au mieux les émissions gazeuses (Laville et al. 1999; Hénault et al. 2005; Vilain et al. 2010). Les concentrations en N₂O gazeux (mais aussi en CO₂ et CH₄) sont

mesurées à l'INRA de Grignon (collaboration avec Pierre Cellier et Céline Decuq, cf. 2.2.2.3). Les émissions de N_2O sont déterminées à partir de cinq réplicats par parcelle. Depuis mars 2013, deux séries de trois chambres automatiques (70 cm x 70 cm x 22 cm de hauteur) ont été installées sur les deux parcelles (AB et AC) afin de mesurer les flux de N_2O et CO_2 de manière quasi-continue, soit 4 mesures de 20 min dans chaque chambre toutes les 6h (0-2h ; 6h-8h ; 12-14h ; 18h-20h). L'air passe à travers l'analyseur de CO_2 (LiCor 820: LiCor Biosciences, Lincoln, NE USA) et de N_2O (Thermo 46C: Thermo Electron, Saint Aubon, France) avec des capteurs de sensibilité d'environ 0.5 ppm et 2 ppb respectivement (Oorts et al. 2007) (Figure 2-8, b). A partir de juin 2013, les parties aériennes des cultures dépassant la hauteur des chambres, sont régulièrement coupées afin de permettre une fermeture étanche des chambres. Les flux de CO_2 et de N_2O sont calculés via un modèle de régression linéaire ou exponentielle.

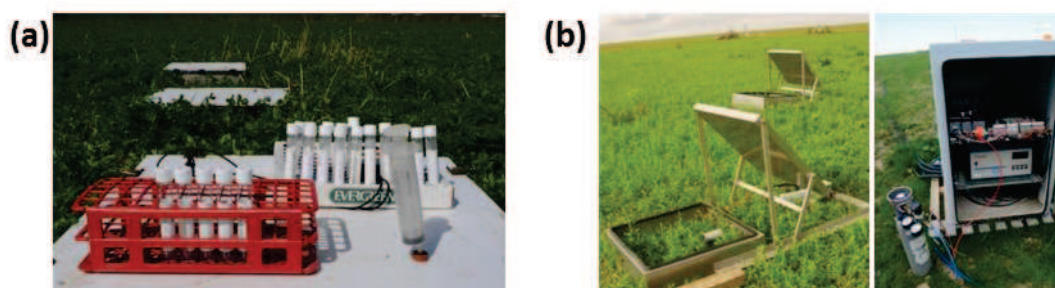


Figure 2-8. Photos des prélèvements de GES dans les chambres manuelles (a) et en chambres automatique avec l'analyseur par spectrométrie infra-rouge.

2.2.3.2. Expériences en conditions contrôlées

Afin de mesurer les émissions potentielles de N_2O lors des transformations de nitrification et dénitrification, des expériences en batch (en conditions contrôlées et répétables) sont effectuées. Le principe est de mesurer les activités de nitrification et de dénitrification du sol dans des conditions non limitantes, sur une période courte (quatre heures) (Figure 2-9, cf. Chapitre 4).

Des volumes de sols de 400 ml sont prélevés sur les premiers 20 cm de profondeur. Ils sont ensuite conservés au frais (4°C) jusqu'à l'incubation. Les sols sont homogénéisés avant l'expérience en laboratoire. L'expérience des batchs est réalisée avec 5g de sédiments frais, insérés dans un flacon de 100mL, et le flacon fermé à l'aide d'un bouchon étanche muni d'un septum en caoutchouc afin d'y prélever un échantillon de gaz.

Pour l'étude de la **dénitrification**, on utilise une solution saturante de KNO_3 à 60 mg N- KNO_3 l⁻¹, flushée au diazote gazeux (N_2) afin de chasser l'oxygène dissous. Les flacons sont également flushés au N_2 pendant 5 minutes avant l'introduction du sol. Les 5 ml de la solution KNO_3 sont alors injectés dans le flacon à travers le septum. On obtient ainsi un milieu en anaérobiose.

Pour l'étude de la **nitrification**, on ajoutera 5 ml d'une solution de NH_4Cl à 40mg N- NH_4Cl oxygénée à l'aide d'un bulleur (~20% d' O_2).

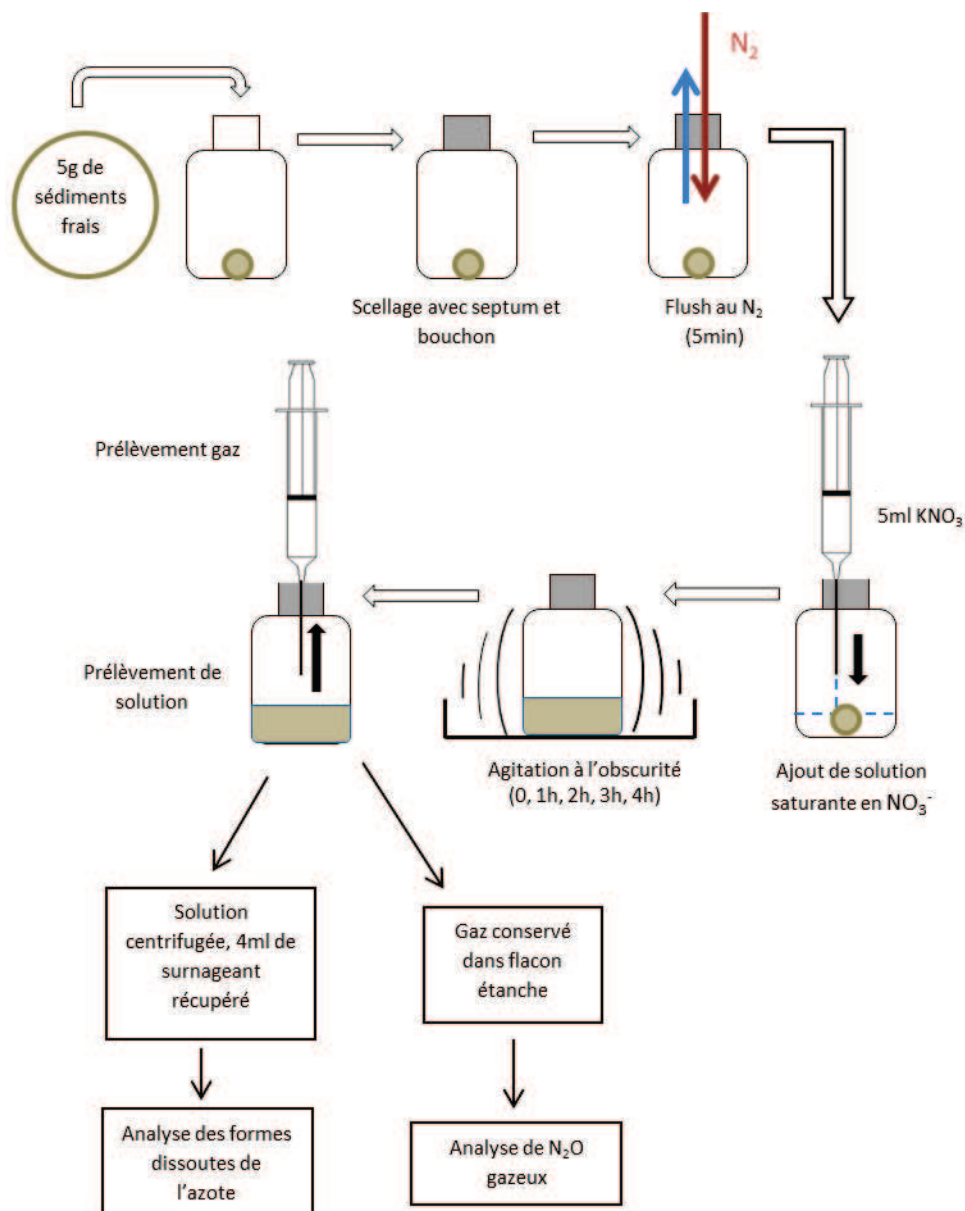


Figure 2-9. Schéma descriptif de mesure du potentiel de dénitrification. Pour la nitrification, le principe est le même, les seuls différences sont : l'étape de flush à l'air, et la solution de KNO₃ qui est remplacée par une solution (5 ml) de NH₄Cl.

Pour chacun des processus d'un sol, 15 flacons sont ainsi préparés (5 temps : 0, 1h, 2h, 3h, 4h x 3 réplicats par temps) ; les flacons sont alors disposés sur une table agitante à l'obscurité (enroulés dans une feuille d'aluminium) pour éviter la photosynthèse. A chaque temps d'incubation, 3 flacons sont retirés de la table agitante pour les prélèvements. Après un prélèvement de 30 ml du gaz à l'aide d'une seringue, injectés dans des tubes étanches jusqu'à l'analyse du N₂O (cf. ci-dessus 2.2.2), les suspensions de sols sont centrifugées pendant 10

minutes à 3000 rpm et le surnageant récupéré. Les surnageants sont conservés à -20°C jusqu'à l'analyse des formes dissoutes de l'azote (NH_4^+ , NO_3^- , NO_2^-) à l'auto-analyseur.

2.2.3.3. Analyse de la phase gazeuse (N_2O)

Le chromatographe en phase gazeuse (INRA-AgroParitech à Grignon) permettant d'analyser les gaz (N_2O , CO_2 et CH_4) est un Varian (3800) couplé à un détecteur à capture d'électrons (ECD) et d'un détecteur d'ionisation à flamme (FID). Les concentrations CO_2 , CH_4 sont ensuite calculées en comparant les aires des pics obtenus avec leurs étalons respectifs (**Table 2-2**). Les flux de GES sont déterminés à partir des valeurs des pentes des régressions linéaires réalisées sur les mesures des cinq chambres manuelles ou celles des trois chambres automatiques.

Table 2-2. Concentrations des quatre étalons des gaz analysés

gaz	CH_4 (ppm)	CO_2 (ppm)	N_2O (ppb)
A	0,997	202	289
B	2,013	997	501
C	2,999	2000	651
D	3,977	2962	805

Toutes ces méthodes maintenant décrites, vont permettre de quantifier l'impact des agrosystèmes sur les fuites d'azote dans l'environnement. Afin de conserver les articles dans leurs intégralités, les méthodes spécifiques de chaque chapitre ont été gardées en l'état.

Partie II.

*Les fuites d'azote à différentes échelles
spatiales et temporelles*

Plusieurs facteurs régissent les fuites d'azote tant à l'échelle microscopique (activités des microorganismes), macroscopique (itinéraires techniques) que régionale (climats, transferts). Dans cette seconde partie, on s'intéressera aux fuites d'azote à l'échelle de la parcelle et de l'exploitation sur trois années, puis à l'échelle du processus sur quelques heures et enfin à l'échelle du bassin versant pour plusieurs années hydrologiques.

Tout d'abord à l'échelle d'une exploitation mixte (AB/AC), nous étudions les fuites d'azote au cours d'un cycle de rotation culturale reconstitué, ce qui permet de prendre en compte l'impact des itinéraires techniques ainsi que celui des précédents culturaux. Les fuites d'azote dans les systèmes agricoles biologiques sont particulièrement complexes à étudier car leurs pratiques impliquent des reports d'azote d'une année sur l'autre, en raison des rotations longues, de 7 à 9 ans. La rotation (temporelle) dans cette exploitation ne correspond pas à l'assolement annuel (spatial). En conséquence, un suivi pluriannuel a été nécessaire pour reconstituer l'ensemble de la rotation. Le suivi de cette exploitation mixte nous a également permis de tester différentes approches méthodologiques pour la quantification des émissions de N_2O (chambres automatiques et manuelles), la lixiviation du NO_3^- (drains, bougies) et les flux d'eau (observés, calculés) (**Chapitre 3**).

Les mesures *in situ* se prêtent parfois mal à l'étude analytique de certains facteurs de régulation. Cela nous a conduit à la mise en œuvre d'une méthodologie en laboratoire permettant d'effectuer un traitement sur un lot d'échantillon (*batch* en anglais) en conditions contrôlées. En particulier, nous avons testé l'effet de la température (un facteur de régulation difficile à isoler lors des observations *in situ*) sur les processus de dénitrification, nitrification et leurs émissions de N_2O associées. Les potentiels de nitrification et de dénitrification sont également des outils utiles pour comparer la prédominance des processus (nitrification, dénitrification) selon les contextes. Les échantillons de sols ont été prélevés à quatre reprises sur l'une des parcelles de l'exploitation mixte précédemment étudiée (**Chapitre 4**).

Enfin, l'ensemble de ces mesures réalisées sur le terrain (bougies, piézomètre, drains, chambres d'accumulation de gaz) ont permis de reconstituer le bilan des transferts d'azote contribuant à la cascade de l'azote, à l'échelle du petit bassin versant de l'Orgeval où se situe l'exploitation agricole mixte. De plus, les chroniques de concentrations nitriques à l'exutoire du bassin ont été simulées grâce à la mise en œuvre du modèle Sénèque-RiverStrahler. Des scénarios de réduction des concentrations nitriques des eaux de surface sont aussi testés (**Chapitre 5**).

3

Nitrous oxide emissions and nitrate leaching in an organic and a conventional cropping system

Benoit M¹, Garnier J^{1,2}, Billen G^{1,2}, Tournebize J³, Gréhan E⁴

Ce chapitre a été soumis dans la revue *Agriculture Ecosystems & Environment*.

¹ UPMC, UMR 7619 Metis, Box 105, 4 place Jussieu, 75005 Paris, France

² CNRS, UMR 7619 Metis, Box 105, 4 place Jussieu, 75005 Paris, France

³ IRSTEA, UR HBAN, 1 rue Pierre Gilles de Gennes CS 10030 9276 Antony, France

⁴ INRA UR Agro-Impact, Pôle du Griffon, 180 rue Pierre-Gilles de Gennes, 02000 Laon, France

3.1. Abstract

Agricultural activities can lead to nitrogen losses in the environment, particularly nitrate (NO_3^-) leaching and nitrous oxide (N_2O) emissions. This study aims to measure N losses from organic farming (OF) and conventional farming (CF) arable cropping systems, both adopted in a single farm, located on the same drained loamy soil in the Seine basin, in the North of France. Leaching was measured with ceramic cups and N_2O emissions with automatic and manual chambers over the 2011–2014 period.

Manual chambers showed the same trend as automatic chambers but underestimated small variations and overestimated peak emissions. On average, N_2O emissions were lower in OF ($0.65 \text{ kg N ha}^{-1} \text{ yr}^{-1}$) than in CF fields ($0.91 \text{ kg N ha}^{-1} \text{ yr}^{-1}$). The mean amount of N leached was $14.6 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ in the OF system during the 8-year rotation (alfalfa 1, alfalfa 2, wheat, green bean, wheat, faba bean, wheat, flax) and $19.5 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ in the CF system for a 3-year CF rotation (legume, wheat, wheat), corresponding to 11% and 14% of total N inputs, respectively. Leached N and N_2O expressed per unit of protein-N harvested were slightly higher in CF ($0.12 \text{ g NO}_3\text{-N g}^{-1} \text{ N}$ and $5.6 \text{ mg N}_2\text{O-N g}^{-1} \text{ N}$, respectively) than in OF systems ($0.11 \text{ g NO}_3\text{-N g}^{-1} \text{ N}$ and $4.8 \text{ mg N}_2\text{O-N g}^{-1} \text{ N}$, respectively). These results show the lower environmental impact of organic farming practices compared to CF practices but show further margins of progress in both systems (types, date and amount of fertilisers, legume management and catch-crops).

3.2. Introduction

Nitrogen (N) is necessary to plant growth, but when in excess its impacts negatively affect the environment, namely through nitrate (NO_3^-) leaching and nitrous oxide (N_2O) emissions. Agricultural activity is recognised as significantly contributing to diffuse-source contamination in groundwater (Ledoux et al. 2007) and N_2O emissions from agriculture are estimated to account for more than 75% of total global anthropogenic emissions (Isermann 1994; Stehfest and Bouwman 2006).

Organic farming (OF) is often considered as a credible alternative system for preventing environment contamination from pesticides and enhancing biodiversity (Pelosi et al. 2009; Henneron et al. 2014). However, concerning its impact on NO_3^- leaching and N_2O emissions, the situation is not that straightforward, and meta-analyses on this subject do not lead to an unequivocal conclusion (Mondelaers et al. 2009).

With regard to studies on NO_3^- leached in arable crops in northern Europe, using field measurements by means of porous ceramic cups (Berg et al. 1999; Stopes et al. 2002), drainage perimeters (Korsaeth and Eltun 2000) or soil mineral nitrogen (SMN) (Haas et al. 2002), 45% less leaching was observed in OF than in CF. However, (Kristensen et al. 1994) showed 103% more SMN in OF fields than in CF fields, mainly due to regular application of manure. Concerning N_2O emissions, a meta-analysis based on 16 publications showed that arable crops in OF led to higher N_2O emissions ($3.22 \text{ kg N}_2\text{O-N ha}^{-1} \text{ y}^{-1}$) than in CF ($2.97 \text{ kg N}_2\text{O-N ha}^{-1} \text{ y}^{-1}$), in contrast to grassland where OF shows a lower impact (Skinner et al. 2014). However, a long-term study (12 years) in an experimental station and a 1-year survey in 20 commercial OF systems both led to the conclusion that N_2O emissions in OF were 60% lower than in CF systems (Stalenga and Kawalec 2008). This high level of heterogeneity between studies originates from the differences in the pedo-climatic context, farming systems, experimental research methods, the time period and the scope of the measurements time and units of the results. For example, considering area-scaled or yield-scaled expression does not lead to the same conclusion: while area-scaled leaching and N_2O emissions were generally lower in OFs than in CFs, the reverse is often true for yield-scaled values (Flessa et al. 2002; Olesen et al. 2006; Tuomisto and Helenius 2008; Korsaeth 2008). Although all these results were based on experimental data from the North of Europe, no study from the North of France has yet been published on these issues.

Therefore, to complete the data concerning OF impacts on N losses, we instrumented a commercial farm, chosen because it cultivates both OF and CF fields, in order to determine the main factors of N losses in the two systems. The measurements lasted 2–3 years, in order to take into account most of the crops from the OF and CF rotations. Leaching was calculated based on ceramic cup data. For N_2O emissions, we used both manual and automatic chambers. Nitrogen losses (leaching and emissions) were expressed per area (area-scaled) as well as per amount of protein-N in the harvest (yield-scale).

3.3. Materials and methods

3.3.1. Experimental site and design

The arable crop farm studied is located in Aulnoy (N 48° 50' 49.888" E 3°5'21.134") and one-third of its surface has been converted to OF (**Figure 3-1**).

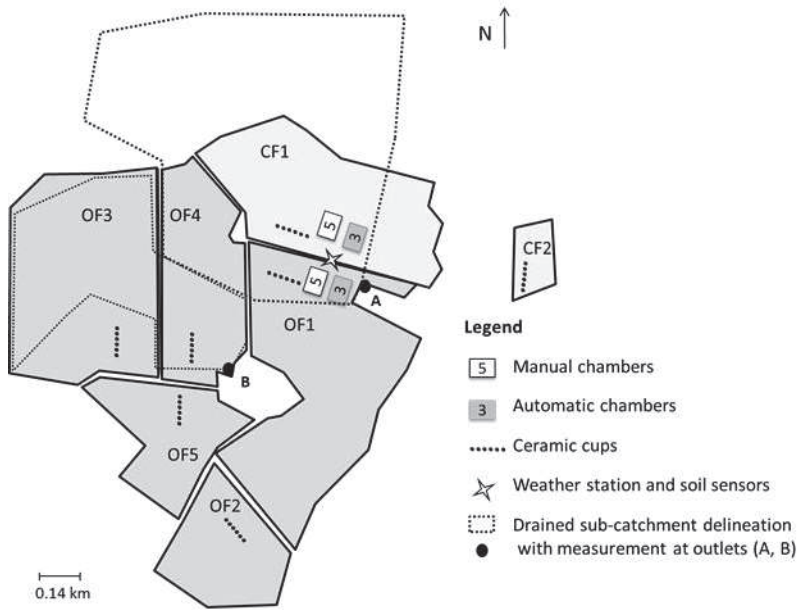


Figure 3-1. Location of experimental equipment in the five organic (OF) and two conventional farming (CF) fields. The sub-catchments of two equipped drain collectors (A + B) are also indicated.

The climate is semi-oceanic, with a local mean annual precipitation of about 680 mm and a mean ambient temperature of 4°C in winter and 19°C in summer. Most of the farm is covered by a quaternary loess deposit (up to 10 m thick). This upper layer is enriched in-depth in limestone and the lower layer is enriched in clay (around 1 m deep, with a low permeability), causing a shallow water table and waterlogged soil in winter, due to the low soil permeability. As a consequence, all the fields studied are artificially drained 90 cm deep by PCV perforated pipes with 12-m spacing.

In conventional farming (CF), two fields (CF1 and CF2) were monitored; they were cropped with sugar beet, wheat, grain legumes (faba beans or peas) with CC. Mineral fertilisers were fractioned into three applications in spring for a total amount between 130 and 200 kg N ha⁻¹ calcium ammonium nitrate (CAN 27%) and urea ammonium nitrate (UAN) solution (composition: 25% NH₄⁺, 25% NO₃⁻ and 50% CO(NH₂)₂) (**Table 3-1**).

Five OF fields were studied from 2 to 3 years, in order to cover the whole rotation. The OF rotation generally includes eight successive crops: alfalfa 1, alfalfa 2, wheat/rye, green bean, wheat/spelt, faba beans, wheat, flax with catch crops (CC). The organic fertilisers, applied in fall (September), are sugar beet vinasse (60 kg N ha⁻¹, C/N = 8) and/or horse manure

(around 25 kg N ha⁻¹, C/N ≈ 25). All the farm's OF fields were studied (OF1, OF2, OF3, OF4, OF5) in order to represent all the crops of the OF rotation during the 3 years studied. Cereal straw was generally buried and alfalfa was cut three times a year, and either exported (OF4) or buried (OF5) (**Table 3-2**).

We assumed that monitoring several plots during the same period (instead of only one field for a longer period) allowed us to calculate the agricultural balance representing a full crop rotation for both OF and CF.

Meteorological data were recorded on site by weather stations equipped with a rain gauge. Soil data were recorded by soil temperature and volumetric moisture sensors (CS215, all equipment from Campbell Scientific Ltd, Shepshed, UK). Both time domain reflectometry (TDR) and temperature sensors were set 20 cm deep. Moisture (from TDR) and temperature were automatically measured at hourly intervals. A relationship between isolated volumetric water contents and corresponding TDR values was used to obtain continuous volumetric content.

Table 3-1. Agricultural management of the two conventional fields studied (CF1, CF2), for the previous year (*in italic*) and during the measurement years. In bold, the crops measured for NO₃⁻ leaching and with (*) for N₂O emission.

Field Management	2010-2011	2011-2012	2012-2013	2013-2014
CF1 crops	<i>sugar-beet</i>	wheat*	faba bean*	wheat*
seeding - harvest dates	25/03-15/10	25/10 - 20/07	15/03 - 22/08	dbt oct
catch crops	<i>Mustard</i>		mix-seeds**	
seeding - harvest dates	20/08-03/12		01/08- 05/01	
Fertilisation type (rates, kgN.ha ⁻¹ ; dates, dd/mm)	UAN, CAN: 150; 30/04	CAN: 58; 22/02 UAN: 57 N; 27/03 UAN: 48; 10/04 UAN: 15; 04/05	vinasse: 60; 01/09	CAN: 58; 25/02 UAN: 110; 25/03 UAN: 35; 30/04
Tillage (dates)			10/01_	
crop residues	exported	buried	buried	buried
Yield (t.ha ⁻¹)	9.7	9.11	5.4	8
CF2 crops	<i>wheat</i>	<i>peas</i>	wheat	wheat
seeding - harvest dates	25/10 - 20/07	20/06-25/09	20/10 - 20/07	25/10 - 25/07
Fertilisation type (rates, kgN.ha ⁻¹ ; dates, dd/mm)	CAN: 58; 25/02 UAN: 110; 25/03 UAN: 35; 30/04	UAN, CAN: 130; 01/05	CAN: 58; 25/02; UAN: 110; 25/03	CAN: 58; 25/02 UAN: 110; 25/03 UAN: 35; 30/04
Tillage (dates)		15/12_		
crop residues	<i>buried</i>	exported	buried	buried
Yield (t.ha ⁻¹)	9.7	2.5	9.2	8.1

Mixed-seeds**: 60% Vetch, 15% Clover, 15% Phacelia, 10% Mustard.

Table 3-2. Agricultural management of the five organic fields for the previous year (*in italic*) and during the years measurements were taken. In bold, the crops measured for NO₃⁻ leaching and with (*) for N₂O emission.

Field Management	2010-2011	2011-2012	2012-2013	2013-2014
OF1 crops	<i>triticale</i>	faba bean*	wheat*	alfalfa*
seeding - harvest dates	01/10-20/07	01/03 -22/08	20/10 - 30/08	21/08 - 04/09
catch crops		Mustard		3 cuts exported
seeding - harvest dates		25/07 - 22/11		(5/05; 16/06; 02/08)
Fertilisation type: (rates, kgN.ha ⁻¹ ; dates, dd/mm)	<i>vinasse: 50; 05/09</i>		manure: 25; 01/08 vinasse: 56; 01/09	
Tillage, <i>harrows and hoe</i> (dates)	15/02-15/03	15/12	15/02-15/03	
crop residues	<i>buried</i>	buried	buried	exported
Yield (t.ha ⁻¹)	7.1	4.1	3.8	9
OF2 crops	<i>alfalfa 2</i>	wheat/rye	green bean*	wheat/spelt
seeding - harvest dates	- 04/09	20/10-20/07	20/06-25/09	15/10 - 25/07
catch crops			mix-seeds**	
seeding - harvest dates			01/08- 10/12	
Fertilisation type (rates, kgN.ha ⁻¹ ; dates, dd/mm)	<i>2 cuts buried (01/05 - 01/07) + exported (01/08)</i>		vinasse: 50; 30/09	
Tillage, <i>harrows and hoe</i> (dates)		15/02-15/03	01/07-01/08	15/03_
crop residues	<i>exported</i>	buried	buried	buried
Yield (t.ha ⁻¹)	7.2	3.4	1.5	4.5
OF3 crops	<i>wheat</i>	alfalfa	alfalfa*	wheat
seeding - harvest dates	15/10 - 25/07	3 cuts : 15/05, 15/08, 30/10	3 cuts : 15/05, 15/08, 30/10	25/10 - 15/07
Fertilisation type (rates, kgN.ha ⁻¹ ; dates, dd/mm)				
Tillage, <i>harrows and hoe</i> (dates)				20/03_
crop residues	<i>buried</i>	exported	exported	buried
Yield (t.ha ⁻¹)	5.5	9.8	12	6.5
OF4 crops	<i>faba beans</i>	<i>wheat</i>	flax	green bean
seeding - harvest dates	25/02 - 20/08	15/10-30/07	28/03-20/08	22/06 -
catch crops			mix-seeds**	flax, aot, clover
seeding - harvest dates			01/08- 21/11	01/09-15/03
Fertilisation type (rates, kgN.ha ⁻¹ ; dates, dd/mm)		<i>vinasse: 50; 01/09</i>	vinasse: 60; 01/09	bones powder : 90; 20/05
Tillage, <i>harrows and hoe</i> (dates)	10/01_13/04	15/02-15/03		
crop residues	<i>buried</i>	<i>buried</i>	exported	buried
Yield (t.ha ⁻¹)	5.5	5.5	5.6	n.d
OF5 crops	<i>wheat</i>	<i>alfalfa</i>	alfalfa	wheat
seeding - harves dates	15/10 - 25/07	3 cuts: 15/05, 15/08, 30/10	3 cuts: 15/05, 15/08, 30/10	25/10 - 15/07
Fertilisation type (rates, kgN.ha ⁻¹ ; dates, dd/mm)				
Tillage, <i>harrows and hoe</i> (dates)				20/03_
crop residues	<i>buried</i>	<i>buried</i>	buried	buried
Yield (t.ha ⁻¹)	5.7	9.8	12	6.5

Mixed-seeds**: 60% Vetch, 15% Clover, 15% Phacelia, 10% Mustard.

3.3.2. Soil measurements

Soil was sampled at a 0- to 20-cm depth using a hand-auger, for each gas emission measurement. Triplicate samples of soil were weighed (30 g) to determine moisture and soil organic carbon (SOC) in the laboratory. Soil samples were heated at 105°C (48 h) for the former, and calcined at 450°C (4 h) for the latter, and re-weighed after each step. Soil mineral N stocks were determined after KCl extraction, with 5 g of fresh soil in 20 ml of KCl (2 M) for 2 h on a shaking table. The suspensions were centrifuged for 10 min at 3000 rpm and the supernatant was frozen at -18°C until analysis in the autoanalyzer (Quattro, Bran & Luebbe).

The water-filled pore space (WFPS) is currently used to compare the level of anoxia in different soils, which is one of the main factors controlling N₂O emissions (Groffman and Tiedje 1991). The WFPS is determined by the following equation (Equation 3-1).

Equation 3-1.

$$WFPS = \frac{Hv}{1 - (BD/PD)} \times 100$$

where Hv, the volumetric soil moisture, BD is soil bulk density and PD is soil particle density (2.65 g cm⁻³).

3.3.3. Soil solution measurement

For the 3 years, N losses below 90 cm deep were measured during the leaching period (December to March) with vertical ceramic cups. During this period, we assumed that the roots were within the 0- to 90-cm-deep soil horizon, so that the soil solution concentration 90 cm deep represents the sub-root concentration, a term that will be used throughout this paper. A total of seven fields were equipped with vertical ceramic cup devices (SDEC, France, SPS Ø 31 mm, 85 cm long), implemented with a manual auger of the same diameter, with the head placed 5 cm below the ground surface. The ceramic cups (six per plot) were arranged on a line parallel to the tillage direction at a minimum of 14 m from the edge of the field, in order to avoid any side effect. Water was sampled over 4 months, with an average of 10 sampling dates. After 48 h of vacuum setting (-0.8 bar), samples were taken weekly at the beginning of the rainy hydrological season and fortnightly for the rest of the period. We assumed that most of the NO₃⁻ loss occurred during this period, in relation to the rainfall distribution during the year and the plant uptake in spring.

All soils studied are subsurface-drained, with one collector outlet (A) equipped for discharge measurement and one collector outlet (B) where water samples were taken in 2013–2014 in order to compare nitrate concentrations in the drain (7.9 ha) and in ceramic cups from the two organic fields (OF3, OF4) (cf. Figure 3-1).

The flow in the drain pipe (A) collects 33 ha of an area including OF1, OF4, CF1 and fields not studied here; its discharge was measured during the 3-year period. The water level measurement devices (SE-200 OTT and Druck PDCR1830 sensors) were installed in 90° V-notch sections situated on the main drain pipe A. Discharge gauges were used to relate water levels to flow rates by means of rating curves with a 15-min time step. We assume that the drained flow covering 33 ha is representative (in terms of specific flow expressed in mm) of all the monitored plots (OF and CF).

The soil solution taken from each ceramic cup was frozen until analysis of N concentrations using an autoanalyser Quattro (Bran & Luebbe). Nitrite (NO_2^-) and ammonium (NH_4^+) were measured using the sulphanilamide method (Jones 1984) and NO_3^- concentrations were determined after NO_2^- reduction. It was verified that deep freezing the samples does not alter the concentration.

3.3.4. Nitrous oxide emissions

N_2O flux was measured fortnightly from October 2011 to June 2014 in OF1 and CF1 and from October 2012 to June 2013 in OF2 and OF3, using the closed manual chamber technique (Hutchinson and Livingston 1993). For the gas flux measurement, a series of five aluminum chambers (open base, 50 cm $\times$ 50 cm) were inserted into the soil to a depth of 15 cm and left throughout the cropping cycle. Sampling was done in the morning, around 11 a.m. Chambers were removed only during field operations such as harvest or tillage. When vegetation was taller than the height of the chambers, 50-cm extensions were added. Each chamber was therefore hermetically closed with a lid, equipped with a small septum to allow gas sampling after accumulation, and maintained with two clips per side. A strip of adhesive Neoprene rubber foam, 25 mm wide $\times$ 5 mm thick, was stuck on its back to ensure airtightness.

Four gas samples were taken from each chamber headspace (0, 10, 20 and 30 min after closure or 0, 30, 60 and 90 min when the extensions were used) with a 30-ml Terumo® syringe and transferred to a 12.5-ml pre-evacuated glass vial (Labco Exetainer®) for transport back to the laboratory. According to the four measurement times, accumulation kinetics were considered to be significant when the correlation coefficient was higher than 0.95. Nitrous oxide fluxes were calculated as the slope of the linear regression between the N_2O concentrations and the measurement period and converted into $\text{g N}_2\text{O-N ha}^{-1} \text{ d}^{-1}$. Cumulative fluxes were calculated assuming a constant flux rate beginning with the date of each gas sampling until the next gas sampling.

Also, N_2O emissions were measured continuously using the automated chamber method described in (Bessou et al. 2010). Since March 2013 onwards, two series of three automatic chambers (70 cm $\times$ 70 cm $\times$ 20 cm) were installed in the two fields (OF1, CF1), in order to measure N_2O emissions as a quasi-continuous flow. Every 6 h, N_2O emissions were measured in each chamber for 20 min (at 0, 6, 12 and 18 h GMT). Nitrous oxide was determined with a correlation wheel infrared analyser, with sensitivity around 2 ppb (Thermo 46i: Thermo Electron, Megatec France). Recorded data were then stored on a data logger CR 1000 (Campbell Scientific). Nitrous oxide fluxes were calculated as the slope of the linear regression between the N_2O concentrations and the 20-min measurement period and converted into $\text{g N}_2\text{O-N ha}^{-1} \text{ d}^{-1}$; cumulative fluxes were also calculated.

3.3.5. Calculations

3.3.5.1. Statistical analysis

All the statistical analyses were performed using R 3.0.3. Anova tests were determined for soil texture and nutrient. The Student test (*t*-test) was used to compare means for N losses between the two systems. Significance was accepted at a 5% level of probability.

3.3.5.2. Percolating water flux

We used a simple water balance modeling a single-layer reservoir in order to calculate the daily water percolation (W_i , mm d⁻¹) during the sampling period. The soil water content at field capacity (WHC_{FC} , mm) over 0–90 cm was defined using a pedotransfer function (Bruand et al. 2004). The daily water storage (WS_i) was calculated using the daily precipitation (R_i , mm) and the reference evapotranspiration (ETP_i , mm) multiplied by a crop coefficient (k_1) (Morizet et al. 1984), set at 0.5 during the winter period (Perrier et al. 1980; Allen 2000). The model operates at a daily time step to calculate the water storage (WS_i , mm d⁻¹) (Equation 3-2). Once WS_i reaches the field capacity (WHC_{FC}), the excess water is percolating at a first-order rate (Equation 3-3) with a velocity coefficient (k_2), calibrated on the basis of the comparison with observed drain flows.

Equation 3-2. $WS_i = \max (WS_{i-1} + R_i - k_1 * ETP_i - W_{i-1}; 0)$

Equation 3-3. $W_i = \max (k_2 * (WS_i - WHC_{FC}); 0)$

The N leached flow was calculated between each collection date by multiplying the average concentration by the amount of percolated water. Leaching was measured for a 6-month sampling period, from December to May, and is assumed to represent the total leaching of the year, given that vegetation uptake and evaporation prevent leaching during the rest of the year.

3.3.5.3. Biological nitrogen fixation

The calculation of total N inputs takes into account exogenous inputs, via the application of organic and mineral fertilisers and biological nitrogen fixation (BNF) by legumes. The amount and N content of fertilisers (organic and mineral) and the straw management (buried or exported) were documented for each field. Biological nitrogen fixation (BNF) was calculated as a linear function of N accumulated in shoot (N_y) and the amount of fixed N_2 derived from the atmosphere, using regression coefficients (α_{cult} ; β_{cult}) depending on species (Anglade et al. submitted), i.e. for six legume species commonly grown in Northern Europe, namely faba bean, field pea, lentil, alfalfa, and white and red clover. In order to take into account below-ground contributions (BGN), comprising N associated with roots, nodules and rhizodeposition via exudates and decaying root cells and hyphae, we applied a multiplicative factor (BG) derived from a literature survey, amounting to 1.3 and 1.7, respectively, for grain and forage legumes (Equation 3-4).

Equation 3-4. $BNF = BG * [\alpha_{cult} * N_y + \beta_{cult}]$

Then net N input by BNF (net BNF) available for the next crop was obtained by subtracting N harvested in grain or in herbage from the estimated total N input (including BGN) derived from N_2 fixation.

3.4. Results

3.4.1. Climate, soil and hydrologic conditions

During the three cultivation periods studied (2011–2014), cumulative rainfall was 661,698 and 764 mm. The mean daily air temperature ranged from 6°C (min) to 15°C (max), with the extreme temperature down to –10 °C in February 2012 and January 2013 and up to 37°C in August 2012. Daily soil temperatures (–20 cm) never decreased below 0°C or exceeded 20°C. The mean WHC_{FC} was 180 mm. The Wi calculated in 2011–2012, 2012–2013 and 2013–2014 were 72.2, 220.6 and 218.6 mm, respectively, in coherence with the flow measured in the drain pipe of 51.8, 174.1 and 130.2 mm, which accounts for only a fraction (60–80% depending on the year) of the total infiltrated water flux.

Different velocity coefficients (k_2) were tested for the calculation of Wi and the value of 0.5 day⁻¹ provides the best correlation ($R^2 = 0.65$) between Wi and drain flow between each sampling date.

All soils have a silty loam texture. The main characteristics of these soils are presented in **Table 3-3**. The physical and chemical components of each soil layer (clay, SOC, total N contents and C/N) were not significantly different (P -values > 5%) between fields. In each field, the silt content decreased as depth increased, whereas clay and sand contents increased. Soil organic carbon and total N decreased significantly with depth (P -values = 0.0004*** and 0.00019***, resp.).

Table 3-3. Soil characteristics of the seven fields investigated at three depths: texture (clay, silt, sand), organic carbon (C), total nitrogen (N), soil organic carbon (SOC) and the C/N ratio.

	Depth	Clay	Silt		Sand		SOC	N total	C/N
		< 2 µm g kg ⁻¹	2/20 µm g kg ⁻¹	20/50 µm g kg ⁻¹	50/200 µm g kg ⁻¹	200/2000 µm g kg ⁻¹	g kg ⁻¹	g kg ⁻¹	
OF1	0-30	258	303	388	42	9	9.84	1.03	9.53
pH = 7.02	30-60	280	306	373	36	5	5.90	0.66	8.95
	60-90	288	329	335	41	7	2.06	0.29	7.05
OF2	0-30	190	262	343	96	57	15.70	1.58	9.97
pH = 6.41	30-60	188	273	318	93	45	6.53	0.68	9.62
	60-90	210	278	311	85	44	5.09	0.55	9.20
OF3	0-30	166	291	395	58	19	11.00	1.11	9.92
pH = 7.04	30-60	214	295	401	44	16	5.53	0.64	8.59
	60-90	271	298	383	32	12	2.91	0.40	7.36
OF4	0-30	232	274	466	115	36	9.03	0.93	9.76
pH = 7.10	30-60	270	219	431	148	45	3.75	0.39	9.52
	60-90	251	309	387	98	31	6.29	0.70	9.05
OF5	0-30	223	268	353	111	45	15.70	1.49	10.50
pH = 6.76	30-60	256	250	360	97	37	8.75	0.87	10.10
	60-90	336	207	282	120	55	4.90	0.50	9.85
CF1	0-30	247	264	362	84	43	10.20	1.05	9.70
pH = 7.28	30-60	251	209	250	150	140	1.09	0.16	6.78
	60-90	240	218	321	98	123	4.45	0.47	9.59
CF2	0-30	168	274	448	101	9	11.10	1.09	10.20
pH = 6.57	30-60	163	287	432	105	13	6.28	0.67	9.40
	60-90	195	313	368	113	11	6.74	0.74	9.10

3.4.2. N₂O emissions with manual and continuous sampling

3.4.2.1. *Comparing manual and automatic chambers*

Measurements with manual chambers captured the highest peak of N₂O emissions; however, automatic chambers revealed a “fine structure” of emissions (short-lived peaks) between high emission peaks (**Figure 3-2**). As a result, manual chambers did not take into account all small background variations in cumulative emissions. The manual and automatic chambers were compared at the same time for CF1 and OF1 by selecting the 19 dates corresponding to the manual chambers among the continuous measurements. The results show that instantaneous N₂O fluxes were lower in manual chambers than in automatic chambers (–16% for CF1 and –75% for OF1).

The most interesting comparison concerns the integrated fluxes (using trapezoidal integration) in manual chambers (with 19 measurements) and automatic chambers (four measurements per day over 349 days). Manual chambers overestimated N₂O cumulative fluxes in CF1 (+62%) due to the integration over time of high transient N₂O peaks, as revealed with automatic chambers (**Table 3-4**). In contrast, manual chambers underestimated N₂O cumulative flux in OF1 (–77%), because all small N₂O emissions were not sampled, revealing that small background emissions cannot be neglected. It is also notable that the standard deviations were approximately three times lower in automatic than in manual chamber measurements. This is mainly due to the high frequency of measurements in automatic chamber that advantageously compensate the smaller number of replicates in automatic chambers (3 vs 5).

As a whole, N₂O emissions rapidly following fertilisation application accounted for 30% of the annual crop emissions with automatic chambers, lower than the 50% found with manual chambers. Consequently, for the last measurement period with both methods (2013–2014), cumulative fluxes from continuous measurements were chosen as a better integration of N₂O fluxes.

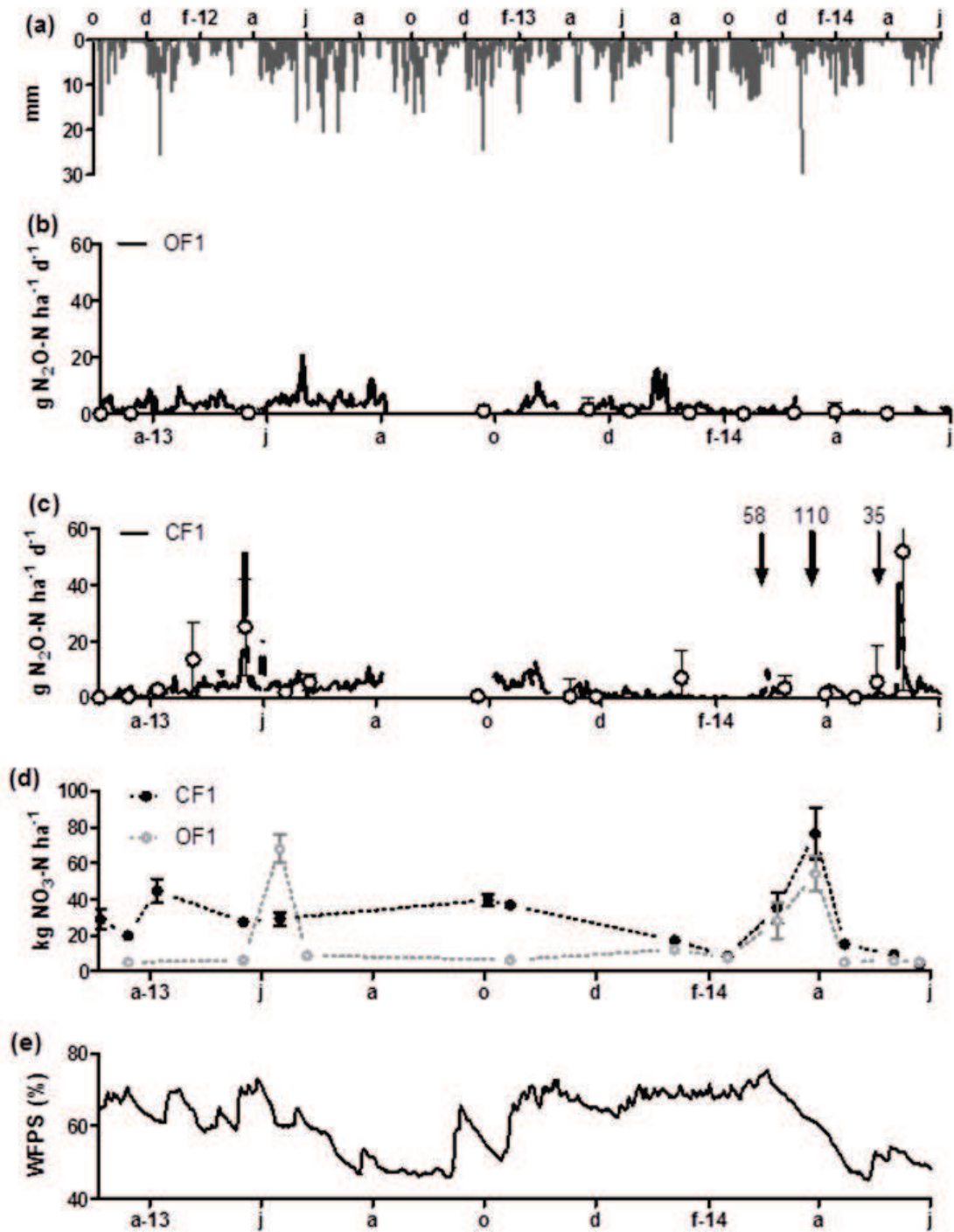


Figure 3-2. Rainfall (a), N₂O fluxes measured in automatic and manual chambers for OF1 (b) and CF1 (c), soil mineral N in the 0- to 10-cm layer (d) and WFPS (e) measured between OF1 and CF1 from March 2013 to June 2014. Arrows indicate the dates and rates (kg N ha⁻¹) of fertilisation.

Table 3-4. Cumulative N₂O emissions calculated with three methods using data obtained with manual or automatic chambers during the 05 March 2013 to 27 May 2014 period; n corresponds to the number of data sampled or recorded, SD is the standard deviation on replicates (five replicates for manual chambers; three replicates for automatic chambers).

Chambers	n	Cumulative N ₂ O emissions g N ₂ O-N.ha ⁻¹			
		CF1		OF1	
		mean	SD	mean	SD
Manual (disconnected sampling)	19	2240	603	207	179
Automatic (corresponding to disconnected sampling)	19	2664	520	844	303
Automatic (continuous)	1396	1390	204	939	57

3.4.2.2. Factors influencing N₂O emissions

Continuous monitoring made it possible to detect the moderate N₂O emissions mainly due to rainfall events, as in OF1, in July 2013 (20.9 g N₂O-N ha⁻¹ d⁻¹) or December 2014 (15.6 g N₂O-N ha⁻¹ d⁻¹) (**Figure 3-2**). This pattern showed the great sensitivity of N₂O emissions to short-term fluctuations in soil physical conditions, particularly WFPS. The highest N₂O emissions (> 10 g N₂O-N ha⁻¹ d⁻¹) observed without fertilisation occurred when WFPS was close to or higher than 60% in OF1 and CF1 (**Figure 3-3**). These emissions are likely due to denitrification (Oorts et al. 2007; Vilain et al. 2010), whereas the small N₂O emissions observed when WFPS lies between 35% and 60% can be attributed to nitrification (Garrido et al. 2002; Bateman and Baggs 2005).

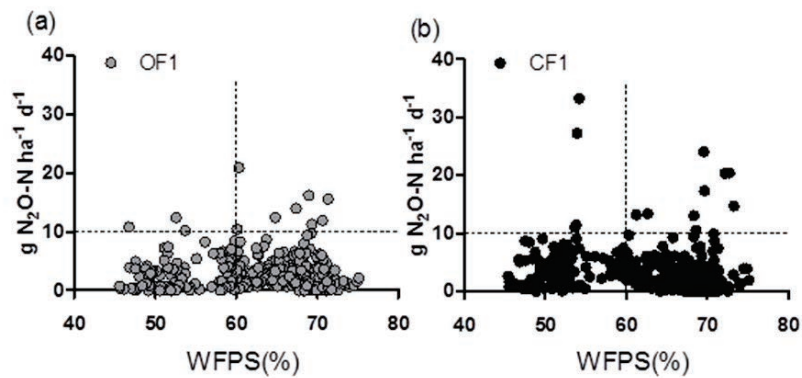


Figure 3-3. Relationship of soil N₂O emissions measured with continuous sampling in OF1 (a) and CF1 (b) and soil WFPS from March 2013 to June 2014.

Soil mineral N stocks also appeared to be an important controlling factor of N₂O emissions of a given field. Indeed, the mean N₂O flux of each field was related to the mean NO₃⁻ contents, which varied from 5 to 54 kg NO₃-N ha⁻¹ in OF1 and from 4 to 76 kg NO₃-N ha⁻¹ in CF1 (**Figure 3-4**).

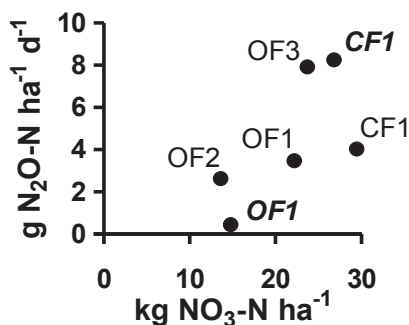


Figure 3-4. N₂O emission mean fluxes versus soil nitrate contents (0–10 cm) measured (between 5 and 10 cm) during the periods studied in four fields (OF1, OF2, OF3 and CF1) in 2012–2013 and two fields (OF1, CF1) in 2013–2014 (in italics and bold).

Finally, fertilisation events are also strong determinants of N₂O emissions. The highest N₂O emission values were observed in spring, after mineral fertilisation in CF1 and decreased within the following month to the level found before fertilisation. In February 2014, the CAN application (58 kg N ha⁻¹) led to a peak of 14.7 g N₂O-N ha⁻¹ d⁻¹ 4 days later (**Figure 3-2**). The N₂O emissions from UAN application (110 kg N ha⁻¹) were not sampled due to a failure of the automatic chambers, but the third application (UAN, 35 kg N ha⁻¹) at the end of April led to 46 g N₂O-N ha⁻¹ d⁻¹ 10 days later.

In OF fields, vinasse application (50 kg N ha⁻¹) in fall 2012 on OF1 and OF2 led to N₂O emissions up to 54 ± 40 g N₂O-N ha⁻¹ d⁻¹, whereas with no fertilisation (OF3), N₂O emissions remained low (< 1 g N₂O-N ha⁻¹ d⁻¹). From these measurements, N₂O emissions were twice as high for UAN fertilisation as for vinasse or CAN, but the difference in the monitoring method for organic and conventional post-fertilisation measurements must be kept in mind. After CC (mixed-seed including legumes) incorporation in the soil in winter, no N₂O peak emissions were measured in OF2 or CF1. However, in CF1, substantial N₂O emissions (25 ± 17 g N₂O-N ha⁻¹ d⁻¹) were measured in spring following soil disturbance (sowing and CC burying).

3.4.2.3. Cumulative N₂O fluxes for crops in OF and CF rotations

Examining the different crops, alfalfa fields (OF3 in 2012–2013, OF1 in 2013–2014) showed the lowest cumulative N₂O emissions (mean, 0.33 ± 0.01 kg N₂O-N ha⁻¹); wheat OF and CF fields (OF1 in 2012–2013 and CF1 2011–12, 2013–2014) produced almost the same cumulative emissions (0.65–0.75 ± 0.06 kg N₂O-N ha⁻¹). Regarding the cumulative N₂O fluxes of grain legumes, they were lower in OF1 in 2011–2012 (0.65 ± 0.48 kg N₂O-N ha⁻¹) and OF2 (1.10 ± 1.08 kg N₂O-N ha⁻¹) than in CF1 in 2012–2013 (1.42 kg N₂O-N ha⁻¹) (**Table 3-5**).

Considering the rotation, mean N₂O emissions in the OF rotation (five crops out of eight) were 0.65 ± 0.33 kg N₂O-N ha⁻¹ and 0.91 ± 0.44 kg N₂O-N ha⁻¹ in CF; the two values are not statistically different (*P*-value = 0.39).

3.4.3. N leached in organic and conventional systems

3.4.3.1. The impact of agricultural practices

In the OF rotation, the lowest sub-root concentrations ($< 5 \text{ mg NO}_3\text{-N l}^{-1}$) were found during alfalfa cropping and the highest concentrations were measured for crops following 2 years of alfalfa (mean, $18.9 \pm 1.4 \text{ mg NO}_3\text{-N l}^{-1}$) (**Figure 3-5 a**). No increases in sub-root concentrations (SRC) were measured when alfalfa was buried and not exported as in OF5 (mean, $17.4 \text{ mg NO}_3\text{-N l}^{-1}$). Grain legumes showed low SRC ($< 5 \text{ mg NO}_3\text{-N l}^{-1}$) due to their association with CC during the fall period (from September to December). Wheat after grain legumes without fertilisation logically led to lower SRC ($5.9 \text{ mg NO}_3\text{-N l}^{-1}$) than after fertilisation ($12.1 \text{ mg NO}_3\text{-N l}^{-1}$). At the end of the rotation, exogenous fertilisation (vinasse) was applied before CC, as in OF4 (2012–13), which led to SRC of $11.3 \text{ mg NO}_3\text{-N l}^{-1}$ (cf. **Table 3-2**).

In the CF rotation, the lowest concentration was measured for grain legumes with CC ($2.5 \text{ mg NO}_3\text{-N l}^{-1}$) and the highest concentration was found for wheat after legumes ($18.5 \pm 6.3 \text{ mg NO}_3\text{-N l}^{-1}$) (**Figure 3-5 b**). Wheat after cereals or sugar beet had intermediate SRC ($11.5 \pm 1.5 \text{ mg NO}_3\text{-N l}^{-1}$). Considering all crops in a full rotation, the SRC averaged $10.8 \pm 8.0 \text{ mg NO}_3\text{-N l}^{-1}$ for CF (3 years) and $7.6 \pm 6.1 \text{ mg NO}_3\text{-N l}^{-1}$ for OF (8 years). In 2013–2014, the mean concentration in drain B was 16.1 mg N l^{-1} , i.e. the average of OF3 and OF4 concentrations, 24 and 4.85 mg N l^{-1} , respectively, the two B-drained fields.

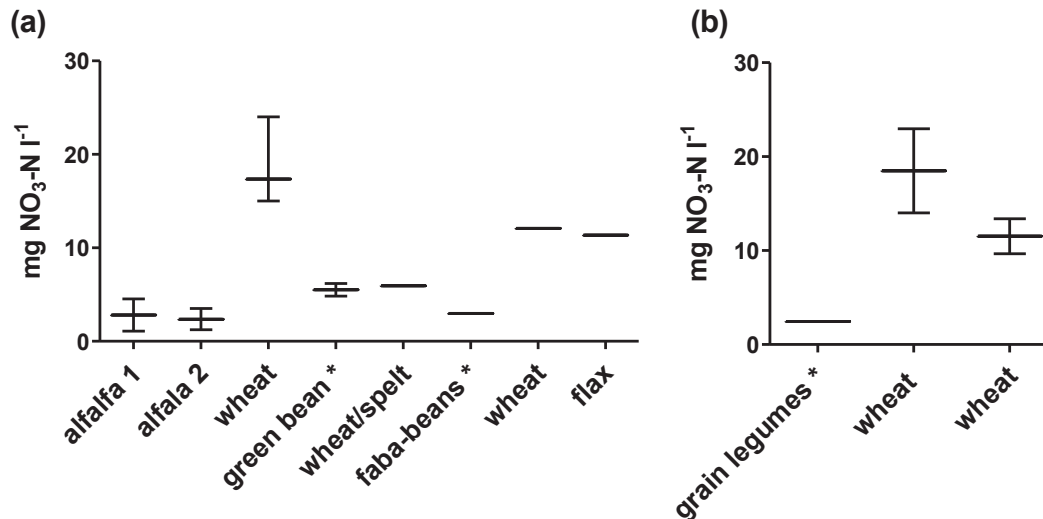


Figure 3-5. Sub-root concentrations at 90 cm deep for OF rotation (a) and CF rotation (b), with median and SD. Spring crops following catch crops or bare soil are indicated with (*).

3.4.3.2. Leaching and water inflow

When calculating leaching for the 3 years, the lowest leaching ($< 10 \text{ kg NO}_3\text{-N ha}^{-1}$) was observed during the first leaching period (2011–2012) due to the lowest water inflow (72.2

mm). In 2012–2013 and 2013–2014, water inflow around 220 mm led to leaching from 3 to 56 kg NO₃-N ha⁻¹, depending on the crops. Finally, mean Wi during the CF and the OF rotations differed by 6%, taking into account all the fields measured (**Table 3-5**), except OF5 due to its testing-farming practices, i.e. alfalfa was not harvested. As a whole, mean leaching over the whole rotation and three different hydrological years equaled 14.59 ± 11.24 kg NO₃-N ha⁻¹ for the eight terms of OF rotation, whereas mean N leaching was 19.54 ± 16.35 kg NO₃-N ha⁻¹ over the 3 years of CF rotation.

Table 3-5. N inputs (fertilisation and BNF), N harvest and N losses: sub-root soil concentration 90 cm deep (SRC), cumulative water inflow (Wi), leaching and N₂O fluxes, over the whole OF and CF rotations during the 3 years studied.

Crops	Field studied	Exogenous fertilisation	Total BNF	N Harvest	Means SRC	Cumulative Wi	Cumulative leaching	Cumulative N ₂ O fluxes
units	years	kg N ha ⁻¹	kg N ha ⁻¹	kg N ha ⁻¹	mg N l ⁻¹	mm	kg N ha ⁻¹	kg N ha ⁻¹
legumes	CF1 (2012-13)	60	165	200	2.46	221	5.39	1.42
wheat	CF2 (2012-13) CF1 (2013-14)	150	0	133	18.48	220	37.48	0.65
wheat	CF1 (2011-12) CF2 (2013-14)	140	0	154	11.54	145	15.75	0.67
CF	mean	117	55	162	10.83	195	19.54	0.91
	<i>SD</i>	<i>49</i>	<i>96</i>	<i>34</i>	<i>8.03</i>	<i>43</i>	<i>16.38</i>	<i>0.44</i>
alfalfa	OF3 (2011-12) OF1 (2013-14)	0	349	270	2.81	145	7.10	0.32
alfalfa	OF3 (2012-13)	0	464	372	1.24	221	3.01	0.34
wheat	OF2 (2011-12) OF3 (2013-14)	0	0	86	20.69	145	33.22	<i>n.d</i>
green bean	OF2 (2012-13) OF4 (2013-14)	50	43	49	5.52	220	10.42	1.10
wheat/spelt	OF2 (2013-14)	0	0	76	5.93	219	14.53	<i>n.d</i>
faba beans	OF1 (2011-12)	0	141	140	2.98	72	2.04	0.75
wheat	OF1 (2012-13)	75	0	67	12.08	221	26.56	0.73
flax	OF4 (2012-13)	65	0	28	11.35	221	19.89	<i>n.d</i>
OF	mean	24	125	136	7.83	183	14.59	0.65
	<i>SD</i>	<i>33</i>	<i>183</i>	<i>122</i>	<i>6.51</i>	<i>56</i>	<i>11.24</i>	<i>0.33</i>

3.4.4. N losses in OF and CF systems

All the N inputs, N outputs and N losses are summarised in **Table 3-5**. The yields obtained for the farm studied are close to the average yields of the region, i.e. for wheat (74 q ha⁻¹ in CF and 40 t ha⁻¹ in OF), faba beans (3.2 q ha⁻¹) or peas (4.0 q ha⁻¹). Atmospheric deposition was estimated as 6 kg N ha⁻¹ yr⁻¹ according to Loubet et al. (2011). For the rotations studied, N total inputs (N exogenous, BNF, atmospheric deposition) was 14% higher in CF rotation (182 kg N ha⁻¹) than in OF (159 kg N ha⁻¹yr⁻¹), to be compared to the 16% higher N harvested in the CF (162 kg N ha⁻¹ yr⁻¹) than in the OF (136 kg N ha⁻¹ yr⁻¹) rotations. However, in the CF

system, fertilisation comprised two-thirds of exogenous mineral fertiliser, whereas in the OF system, exogenous fertiliser accounted for only 20% (including manure).

Relative to N inputs, the parts leached were slightly lower for the OF than for the CF rotations (10% vs 12%) (**Table 3-6**). Taking into account the overall yield of both cropping systems allows calculating losses per unit protein-N produced (Table 6). These yield-scaled losses (NO_3^- leaching and N_2O emissions) are also slightly lower, by about 15%, in the OF system than in CF. Although N losses in gaseous N_2O were 20 times lower than dissolved NO_3^- leaching, the same trends were found for N_2O emissions, slightly lower in OF than CF, both as area- or yield-scaled values, and relative to N inputs.

Table 3-6. Compared N losses between systems over the whole rotation expressed in area-scaled, yield-scaled values or scaled to the total N input.

Systems	Leaching			Emissions		
	per hectare	per N harvested	per total N input	per hectare	per N harvested	per total N input
	$\text{kg NO}_3\text{-N ha}^{-1} \text{ yr}^{-1}$	$\text{g NO}_3\text{-N g}^{-1} \text{ N}$	$\text{g NO}_3\text{-N g}^{-1} \text{ N}$	$\text{kg N}_2\text{O-N ha}^{-1} \text{ yr}^{-1}$	$\text{g N}_2\text{O-N g}^{-1} \text{ N}$	$\text{g N}_2\text{O-N g}^{-1} \text{ N}$
CF	19.54	0.120	0.11	0.91	0.0056	0.0051
OF	14.59	0.107	0.09	0.65	0.0048	0.0042
$-\Delta(\text{CF-OF})/\text{CF}$	-25%	-11%	-14%	-29%	-15%	-18%

3.5. Discussion

3.5.1. Comparing manual and automatic chambers

Continuous measurements by automatic chambers revealed a “fine structure” of emissions (short-lived peaks) within high emission peaks. These patterns cannot be captured by the less frequent manual sampling. The most plausible explanation for these patterns is great sensitivity of N_2O emissions to short-term fluctuations in the soil’s physical conditions, particularly WFPS. However, when integrated N_2O emissions with disconnected measurements, manual chambers under-estimated flux from 15% to 43%, especially when seasonal variations were dominated by such background variations, as was the case in OF without large peaks due to mineral fertilisation. Conversely, using manual chambers within a strategy designed to catch major events (e.g. fertilisation, thaw, rain, etc.) might over-estimate cumulative N_2O fluxes, by over-representing the corresponding peaks.

3.5.2. Mineral and organic fertilisation

Fertiliser application has been clearly shown to enhance N_2O fluxes, but some forms of N fertilisers can lead to higher N_2O emissions than others. Regarding N mineral fertilisers, higher cumulative N_2O emissions are especially related to anhydrous ammonia (NH_3), NH_4NO_3 and $\text{CO}(\text{CH}_2)_2$ (Bouwman, 1996; Henault et al. 1998). Concerning the impacts of organic fertilisers, pig slurry applied, in the same conditions as mineral fertilisers, led to the same amount of N_2O emissions (Petersen, 1999; Chirinda et al. 2010). Vinasse applied to the OF fields during our study never led to peak ranges as high as those observed after mineral fertiliser application in CF. The emissions factor from organic fertilisers (sludge

pellets, poultry manure) were shown to be higher than NH_4NO_3 and urea, but those after cattle slurry application were the lowest (Jones et al. 2007). A similar result was reported for compost added to soil, which did not result in greater N_2O emissions than mineral fertilisers (Meng et al. 2005; Aguilera et al. 2013).

The impact of the organic fertilisers on leaching depends on their total N availability, C/N ratio and the stability of the organic substances. Organic amendments (manure or compost) generally have a C/N ratio greater than 15–20, which has slow-release characteristics, whereas organic fertiliser such as vinasse or poultry manure, with C/N lower than 15, can be directly available. In agreement with the results reported in the literature, vinasse application in 2013–2014 led to the highest SRC, e.g. for OF1 and OF4 (Benoit et al. 2014). Indeed, vinasse, with a C/N of 8.1, leads to a net mineralisation of 75% N and 94% C within 6 days in control conditions (Parnaudeau et al. 2008), or 30–50 days in field conditions (Delin and Engstrom 2009). This indicates that this last type of fertiliser should be applied at least 1 month before the end of crop N uptake and that autumn application is associated with a risk of N leached, unless a crop with a high N uptake is present during winter. Regarding animal manure, after a long-term application (7 years), the result would be greater soil N accumulation (Gutser et al. 2005) and higher N leached (Basso and Ritchie 2005).

3.5.3. Catch crops and legumes

In this study, mustard and a seed-mixture (60% vetch, 15% clover, 15% phacelia, 10% mustard) were used as CC and showed the lowest N leaching, in agreement with most studies (Askegaard et al. 2011; Justes et al. 2012), especially for spring ploughing (Stenberg et al. 1999). However, over the long-term, CC can increase the N pool in the soil (Constantin et al. 2011) and consequently result in possible delayed leaching.

Regarding N_2O emissions from CC, based on 14 publications, the impact would be an increase of $0.11 \pm 1.12 \text{ kg N}_2\text{O-N ha}^{-1} \text{ yr}^{-1}$ (Bertuzzi et al. 2012). However, N_2O emission peaks could be related to the incorporation of CC (including 75% legume) or green manure, as shown for clover which, after incorporation, increased the cumulative N_2O emissions by five (Baggs et al. 2000; Pappa et al. 2011). This question, which is only treated in a few papers, would need further investigation, because winter cover of soils has recently become a current regulation in the area studied.

Concerning legumes, we found emissions of $0.33 \text{ kg N}_2\text{O-N ha}^{-1} \text{ yr}^{-1}$, less than $1 \text{ kg N}_2\text{O-N ha}^{-1} \text{ yr}^{-1}$ currently accepted in an unfertilised pasture in the northern hemisphere (Bouwman 1994), which is called the “background” emission in Bouwman et al. (2002). Legumes would therefore be among the crops with the lowest N_2O emissions. Although losses through leaching are minimal during the vegetation period of alfalfa, the destruction of this crop and the mineralisation of its underground material, with a C/N between 16 and 20, can have an impact on N leaching lasting for several years. Indeed, the high amount of net BNF brought by alfalfa that we estimated here, from 141 to $464 \text{ kg N ha}^{-1} \text{ yr}^{-1}$, was well within the range of data measured in other studies, from 132 to $435 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ (Justes et al. 1999).

3.5.4. Nitrogen balance over the rotations

The percentage lost as N_2O emissions relative to exogenous input is 0.8% in the CF rotation. This estimate is on the order of the 1% standard emission factor for N fertiliser for conventional farming, also used in the IPCC Guidelines for National GHG Inventories

(Bouwman et al. 2002), a value also found for the whole Seine Basin (Garnier et al. 2009). However, taking into account the BNF, the ratio of N_2O emitted over total N inputs is lower (0.5%).

Based on the report of the IPCC (2006), 0.75% of the NO_3^- leached may be transformed into N_2O in groundwater, rivers and estuary (indirect emissions). Here, the amount of indirect N_2O emissions is estimated to be around $0.1\text{--}0.2 \text{ kg N}_2\text{O-N ha}^{-1}$, corresponding to 15% of the total N_2O emissions, within the same range of values estimated (10–14%) with another approach in Garnier et al. (2009).

3.6. Conclusion

In this study, two agricultural systems were studied (OF and CF) within a single farm, characterised by the same conditions (soil, climate, farmer practices) for different successive years. The results showed slightly lower N losses for OF rotations than for CF rotations, either expressed per area or per unit yield.

However, N losses evidenced high variability dependent on climate factors (rainfall event, drought, temperature, etc.) and agricultural practices (fertilisation, crops, CC).

The highest N_2O emissions were observed after mineral or organic fertilisation, which accounts for 30% of the annual emissions. Even if N_2O fluxes did not quantitatively contribute much to the N losses ($1 \text{ kg N}_2\text{O-N ha}^{-1}$) in comparison to the other terms of the N budget (NO_3^- leaching around one order of magnitude higher), N_2O is a powerful greenhouse gas and its emissions are known to have a high impact, equal to 16% of the total greenhouse gas balance.

As for leaching, the lowest losses were attributed to legumes and the highest to crops after legumes. The lower N losses in OF were due to a longer and more diverse rotation. Moreover, annual leaching variations were impacted by the amount of water inflow infiltrated.

Acknowledgement

This study was funded by a PhD scholarship and an equipment sponsorship from the DIM Astrea of the Ile-de-France Region and the Water Agency of the Seine-Normandy Basin. The Research Federation FIRE (FR3020), the ZA-Seine (CNRS), EFEMAIR- N_2O project (Ademe) and the Escapade project (ANR) are also acknowledged for providing funding for the analysis and equipment. Météo-France and Irstea stations are gratefully acknowledged for providing weather data. Many thanks are extended to the farmer, who allowed us to work on his farm and helped us when possible. We also sincerely thank P. Ansart, A. Azougui, C. Chaumont and B. Mercier for their kind laboratory and field technical assistance.

4

Temperature dependence of nitrous oxide production of a luvisolic soil in batch experiments

Benoit M¹, Garnier J^{1,2}, Billen G^{1,2}

Ce chapitre est un article révisé et accepté dans la revue *Process Biochemistry*.

¹UPMC, UMR 7619 Metis, Box 105, 4 place Jussieu, 75005 Paris, France

²CNRS, UMR 7619 Metis, Box 105, 4 place Jussieu, 75005 Paris, France

4.1. Abstract

This study aims to determine the effect of temperature on N₂O produced from a surface soil both in nitrification and denitrification experimental conditions. In a context of changes of both temperature and rainfall, formalizing temperature relationships of N₂O production is useful for model parameterization of process-based modeling approaches.

Samples of luvisolic soils were incubated in destructive slurry batches in controlled specific conditions designed to promote nitrification and denitrification. The experiments were run at seven temperatures, from 5 to 45°C. When fitted with a sigmoid function overall potential activities show that the potential production rate of nitrate (NO₃^{-nit}) is optimal at a temperature of 32.5°C and the potential reduction rate of nitrate (NO₃^{-denit}) being optimal at around 45°C. Regarding the production rate of N₂O, optimal values occur at a temperature about 10°C higher than the optimum temperature for the respective processes, namely 42°C for nitrification and 54.5°C for denitrification. N₂O emissions and nitrite production due to denitrification are around ten times higher than nitrification. These findings show that a sigmoid function of N₂O emissions from both denitrification and nitrification is more relevant than an exponential one.

4.2. Introduction

Nitrous oxide (N_2O) concentration is 1000 times lower than the CO_2 concentration, but its global warming potential is 298 times higher and, in addition, contributes to the destruction of the stratospheric ozone (Cicerone 1987). Besides, 80% of N_2O emissions are due to agriculture activities (Mosier et al. 1998). The main microbial processes responsible for nitrous oxide (N_2O) emissions from soils are nitrification and denitrification (Firestone and Davidson 1989), although pathways such as dissimilatory nitrate reduction to ammonia (DNRA) (Smith and Zimmerman 1981) and nitrifier denitrification (nitrate reduction to N_2O and N_2 by ammonia oxidizers under sub-oxic conditions) (Kool et al. 2011) can lead to nitrous oxide production. Nitrification is the microbial autotrophic oxidation of ammonia (NH_4^+) to nitrate (NO_3^-), with N_2O production as a byproduct, while denitrification is the heterotrophic reduction of NO_3^- to gaseous nitrogen, mostly as dinitrogen (N_2), but also as N_2O . It is well known that nitrification and denitrification depend on temperature, which differ between sites and microbial populations (Focht and Verstraete 1977). However nitrification optimal temperature are usually explored between a narrow range, from 15 to 25°C (Grundmann et al. 1995; Dalias et al. 2002). For denitrification, in the range of 15 to 35°C, an increase of 10°C raised the activity by a factor of about 2 (Q_{10}) (Stanford et al. 1975; Fischer and Whalen 2005).

The amount of N_2O production to the overall rate of nitrification or denitrification is a matter of great significance since it determines the emissions of greenhouse gas from soils. In field conditions, controversies still remain regarding the N_2O dynamics in function of temperatures (Barnard et al. 2005). In laboratory, the role of temperature on the rate of soil N_2O emissions, is generally assumed as an exponential relationship for temperatures below 30°C (either using the Q_{10} formalism or the more rigorous Arrhenius equation) (Addiscott et al. 1991; Smith et al. 1998; Schindlbacher et al. 2004). Regarding this last aspect, in contrast to this generally used approach, we preferred a sigmoid relationship in order to explicitly take into account the existence of an optimum temperature, above which the rate is decreasing and better express the temperature dependence of the rate of microbial processes. A Gaussian function, as proposed in two studies for phytoplankton growth and for nitrification (Lehman et al. 1975; Brion and Billen 1998), generally correctly fits the experimental results. Besides the revision of the formalism of the N_2O potential production, denitrification and nitrification in function of temperature, improved knowledge on N_2O emissions in a wide range of temperature, not yet explored, is still needed in order to better simulate N_2O flux in a climate change context (Rodrigo et al. 1997; Garnier et al. 2009).

In the present study, we investigated the effect of temperature on N_2O production from nitrification and denitrification, including the associated nitrite production, within a range of temperature varying from 5 to 45°C in controlled conditions. A luvisolic soil, well representative of the middle of the Paris basin and Seine watershed, was taken from a parcel of an arable crops farm at several occasions. The aims of this study are (i) to explore the immediate effect of temperature on N_2O production from an agricultural soil (ii) to identify when N_2O production prevails from nitrification or denitrification in such a range of temperature and (iii) to mathematically formalize the relationships for further possible modeling (e.g. Seneque/Riverstrahler) (Ruelland et al. 2007; Passy et al. 2013).

4.3. Materials and methods

4.3.1. Study site

The soil samplings took place in the Orgeval Basin (48°50'23"N, 03°08'22"E), a small sub-basin (104 km²) of the Seine Basin (75000 km²), on the Brie plateau (Seine-et-Marne, East of Paris, France). This small basin is representative of the whole Seine Basin, in terms of lithology, land use and climate. The geology of the area is sedimentary (Oligocene, Eocene) with calcareous subsoil (Brie and Champigny). A brown-leached soil (luvisolic soil) covers a green clay layer, which has led to field drainage. The basin area is mainly occupied by a cultivated area (81%) of intensive cereal crops, the rest is covered by forest (18%) and urban zones (1%). In this region, agricultural land presents higher N₂O emission than forest and grassland, mainly due to denitrification activity, with an emission: nitrification ratio (N₂O emitted/NO₃⁻-produced) around 0.1-0.2 % and an emission: denitrification ratio (N₂O emitted/ NO₃⁻-produced) of 10%, at 20°C (Vilain et al. 2014), twice higher than in sediments (Garnier et al. 2010).

The climate is moderately oceanic, with 763 mm mean annual pluviometry in 2012–13 and 11°C mean temperature. Meteorological data were recorded from two weather stations (Campbell Scientific Ltd, Shepshed, UK) each equipped with a rain gauge (ARG100, Campbell Scientific Ltd, Shepshed, UK) and air temperature probe (Model CS215, Campbell Scientific Ltd, Shepshed, UK).

4.3.2. Batch experiments: soil denitrifying and nitrifying potential

4.3.2.1. *Range of temperatures chosen and soil sampling*

A total of seven temperatures (5±1°C, 10±1°C, 16±1°C, 20±1°C, 30±1°C, 37±1°C, 45±1°C) were tested, in the laboratory for the fixed temperature of 20°C or in a thermostatic chamber (MLR-352 Sanyo) for the rest of the tested temperatures. Samplings have been carried out at four periods: in spring (May 2012), summer (June 2012), fall (September 2012) and winter (January 2013). For each temperature (except for 16°C), the experiments were run two or three times (**Table 4-1**), in order to take into account the in situ variability of the soil and climate. On each date, three 180 ml subsamples were taken from the first 20 cm layer of the soil, pooled and homogenized. Samples were stored at 4°C in hermetic boxes during one night, before the beginning of the experiment.

Table 4-1. Overview of dates when potential activities were measured in batch experiments at the different temperatures. Each temperature effect has been analysed for the luvisolic soil sampled at various seasonal initial conditions, and include thus the *in situ* variability.

T (°C)	5 ± 1	10 ± 1	16 ± 1	20 ± 1	30 ± 1	37 ± 1	45 ± 1
14-May-12	x	x		x	x		
26-Jun-12			x x			x	x
19-Sep-12	x	x		x	x		
14-Jan-13				x	x	x	x

4.3.2.2. Incubation conditions

In order to calculate the rates of the process under study, batches were incubated during four time periods ($t_0 = 0$ min, $t_1 = 60$ min, $t_2 = 120$ min, $t_3 = 180$ min and $t_4 = 240$ min). For each time period, destructive triplicates of the fresh soils were prepared (i.e., five times three samples -15 batches- for each treatment, nitrification or denitrification). These incubation times are relatively short so as to avoid the establishment of limiting conditions (low substrate concentrations), oxygen level modification and bacterial changes both in composition and biomass.

4.3.2.3. Denitrification

For each temperature, the potential of nitrite (NO_2^-) and NO_3^- reduction rates were obtained from the slope of the linear regression between sample concentrations and time, and considered valid when the correlation coefficient significantly differed from zero, taking into account the number of data pairs. The denitrification rate under non limiting NO_3^- and anoxic conditions is called NO_3^- -potential reduction rate ($\text{NO}_3^-_{\text{denit}}$). They have been evaluated from samples of fresh soil (5 g) placed in 100 ml glass flasks wrapped with aluminum foil to prevent photosynthesis in light. All triplicates of (t_0, t_1, t_2, t_3, t_4) were sealed with aluminum caps and flushed with pure N_2 for 5 min to remove oxygen and establish anoxic conditions. Then, 5 ml addition of potassium nitrate (KNO_3 , 60 mg N l^{-1}) flushed with pure N_2 were added with a syringe (5 ml) to each triplicates, initiating the incubation experiment.

4.3.2.4. Nitrification

For each temperature, the potential ammonium (NH_4^+) oxidation, NO_2^- and NO_3^- production rates were obtained from the linear regression between concentrations and time. Similarly, nitrification under non limiting NH_4^+ is called NO_3^- -potential production rate ($\text{NO}_3^-_{\text{nit}}$). In parallel to denitrification experiments, 5 ml of ammonium chloride (NH_4Cl , 30 mgN. l^{-1}) were added with a syringe to other triplicates of fresh soils (5 g). The solution was flushed with ambient air (20.95% O_2 , 356 ppb N_2O and 388 ppm CO_2) during 10 min before each incubation, which ensure aerobic condition for the 4h incubation time. The flasks were also hermetically sealed with aluminum caps.

During incubation, all the samples were agitated on shaking tables at 110 rpm for all temperatures.

At the end of each incubation period, 30 ml of gas was extracted from each triplicates, using gas syringes and transferred into a 12.5 ml pre-evacuated glass vial (Exetainers, Labco Inc., Ceredigion, UK) to determine N₂O concentrations. In order to measure the inorganic nitrogen concentrations in the batch, slurries were centrifuged at 3000 *g* for 10 min and 4 ml of the supernatant was stored at -18°C until analysis, for determining the concentrations in NO₃⁻, NO₂⁻ and NH₄⁺.

4.3.3. Chemical analysis

4.3.3.1. Soil properties

Triplicates of fresh mixed soil (30 g) were used to determine water filled pore space (WFPS) and soil organic matter (SOM) by measuring the loss of weight after heating at 105°C (48 h) and 450°C (4 h), respectively.

The WFPS is currently used and allows determining the level of anoxia. The volumetric soil moisture was calculated by equation (**Equation 4-1**) and the WFPS was then determined by equation (**Equation 4-2**).

Equation 4-1

$$\begin{aligned} \text{volumetric soil moisture} \\ = \text{gravimetric soil moisture} \times \frac{\text{bulk density}}{\text{soil water density}} \end{aligned}$$

$$\text{Equation 4-2 WFPS} = \frac{\text{volumetric soil moisture}}{1 - (\text{bulk density} / \text{soil particle density})} \times 100$$

with the soil bulk density set at 1.4 g.cm⁻³ according to previous work and soil particle density at 2.65 g cm⁻³.

The soil mineral nitrogen (SMN), NH₄⁺, NO₃⁻ and NO₂⁻, was determined by extracting 5 g of fresh soil mixed with 20 ml of KCl (2 M) for 2 h on a shaking table. Suspensions were centrifuged for 10 min at 3000 rpm and supernatants frozen at -18°C until analysis. SMN analysis was conducted on the colorimetric spectrophotometer automate (Quattro, Bran and Luebbe, Norderstedt, DE). Last analysis were done on dried and sieved (2 mm) soil samples, for particle size, organic carbon and total organic nitrogen at the French National Institute of Agronomics Research laboratory in Arras (LAS-Arras, INRA).

4.3.3.2. Dissolved inorganic nitrogen

The method used to analyze NH₄⁺ was based on the indophenol blue method according to [Slawyk and MacIsaac 1972](#). NO₃⁻ and NO₂⁻ were measured with the sulfanilamide method according to [Jones \(1984\)](#) and NO₃⁻ was determined after cadmium reduction into NO₂⁻.

4.3.3.3. N₂O emissions

The concentration of N₂O in the samples were also measured and defined as N₂O potential production rate for nitrification (N₂O_{nit}) or denitrification (N₂O_{denit}) ([Vilain et al. 2012b](#)). N₂O

gas emissions were analyzed using a gas chromatograph (Varian 3800) equipped with an electron capture detector (ECD). Concentrations were calculated by comparing peak areas integrated with those obtained with standard N₂O concentrations (288, 500, 651, 805 ppb).

N₂O production by dry weight (dw) soil unit was calculated taking into account N₂O in the headspace of the flask (100 ml) and dissolved in the slurry (5 ml) (**Equation 4-3**).

$$\text{Equation 4-3. } N_2O (\mu g N g^{-1} dw \text{ soil}) = N_2O (ppb) \times \frac{1}{M_s \times (1 - H_s)} \times \left(\frac{kRT \times V_s \times M_N}{22.4} + \frac{V_f \times M_N}{22.4} \right)$$

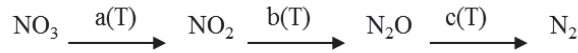
where k is the solubility coefficient of N₂O in water at 20°C ($2.7 \times 10^{-2} \text{ mol l}^{-1} \text{ atm}^{-1}$), R is the perfect gas constant ($0.08 \text{ atm l K}^{-1} \text{ mol}^{-1}$), T is the absolute temperature (K), M_s is the soil mass (g), H_s is the soil gravimetric moisture, V_s is the nitrate solution volume (l), V_f is the flask volume (l), and M_N is the molecular weight of nitrogen in N₂O (28 g mol^{-1}) with 22.4 l mol⁻¹ as the molar volume.

Similar calculations as for NO₃⁻_{denit} or NO₃⁻_{nit} potential rates, allowed determining N₂O_{nit} and N₂O_{denit} potential rates. All rates were expressed per unit of dry weight of soil and per time unit; values were again kept only when the correlation coefficients were statistically different from zero, taking into account the number of data pairs. Lastly, their ratios N₂O/NO₃⁻ produced from nitrification and N₂O/NO₃⁻ reduced from denitrification were calculated in percentage, as an indicator of completion of the reactions.

4.3.4. Calculations

4.3.4.1. Potentials related to temperature variations

The potential rates of each nitrogen form have been described as a function of temperature (T). For denitrification, nitrate reduction into nitrite, a(T), nitrite reduction into nitrous oxide, b(T) and dinitrogen production, c(T), were calculated as:



$$\text{Equation 4-4. } -\frac{dNO_3}{dt} = a(T)$$

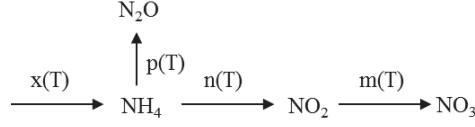
$$\text{Equation 4-5. } \frac{dNO_2}{dt} = a(T) - b(T)$$

$$\text{Equation 4-6. } \frac{dN_2O}{dt} = b(T) - c(T)$$

$$\text{Equation 4-7. } c(T) = a(T) - \frac{dNO_2}{dt} - \frac{dN_2O}{dt} = \frac{dNO_3}{dt} - \frac{dNO_2}{dt} - \frac{dN_2O}{dt}$$

$$\text{Equation 4-8. } b(T) = \frac{dN_2O}{dt} + c(T) = a(T) - \frac{dNO_2}{dt} = -\frac{dNO_3}{dt} - \frac{dNO_2}{dt}$$

For nitrification, ammonium oxidation into nitrite, $n(T)$, nitrite oxidised into nitrate, $m(T)$, and nitrous oxide production, $p(T)$ and the ammonification reaction, $x(T)$, which has been determined thanks to the other reaction rates.



Equation 4-9. $m(T) = -\frac{d\text{NO}_3}{dt}$

Equation 4-10. $p(T) = \frac{d\text{N}_2\text{O}}{dt}$

Equation 4-11. $\frac{d\text{NO}_2}{dt} = n(T) - m(T)$

Equation 4-12. $n(T) = m(T) + \frac{d\text{NO}_2}{dt} = \frac{d\text{NO}_3}{dt} + \frac{d\text{NO}_2}{dt}$

Equation 4-13. $-\frac{d\text{NH}_4}{dt} = n(T) + p(T) - x(T)$

Equation 4-14. $x(T) = n(T) + \frac{d\text{NH}_4}{dt} + p(T) = \frac{d\text{NO}_3}{dt} + \frac{d\text{NO}_2}{dt} + \frac{d\text{NH}_4}{dt} + \frac{d\text{N}_2\text{O}}{dt}$

4.3.4.2. Determination of optimum temperatures

As stated above, the measured rates observed in function of temperature were adjusted to a Gaussian relationship of the form:

Equation 4-15. $A = A_{max} \cdot e^{-\frac{(T-T_{opt})^2}{dti^2}}$

where A_{max} is the maximum rate of the process by dw soil unit ($\mu\text{g N h}^{-1} \text{g}^{-1} \text{dw}$), T_{opt} , the corresponding optimal temperature ($^{\circ}\text{C}$); and dti ($^{\circ}\text{C}$), the scaling temperature factor at the values minimizing the root-mean-square deviation (RMSD), calculated from the differences between the values predicted by the estimation and those actually observed. The optimization of the parameters (A_{max} , T_{opt} , dti) is based on all the potential rates measured at the different temperatures. The coefficient of determination R^2 was also calculated.

4.4. Results

4.4.1. Initial soil chemical properties and potentials determinations

The annual mean rainfall is 700 mm and temperature 9.7°C (40 years of measurements at Boissy-le-Châtel). The studied luvisolic soils are typically composed of silt (66%), clay (20%) and sand (14%). Air temperature at the dates of sampling varied between -2°C (January

2014) and 16.5°C (September 2012), with no or little rain (**Table 4-2**). Soil C/N ratio amounted to approximately 10 and pH between 6 and 7, common values for agricultural soils. The WFPS of the soil samples varied between 54-69%. Some changes of the characteristics of the studied luvisolic soil have been observed during the sampled period, by a factor of about two. Soil organic matter regularly increased during the sampling period from 2.2% in May 2012 to 4.8% in January 2013. Soil mineral N contents means were 33.52 kg NH₄⁺-N ha⁻¹ and 18.82 kg NO₃⁻-N ha⁻¹ and NO₂⁻ remained invariably close to zero.

Table 4-2. Characteristics of the soil and climate conditions: temperature (°C), rainfall (mm), water filled pore space (WFPS, %), ignition loss (%) and nitrate (NO₃⁻), nitrite (NO₂⁻) and ammonium (NH₄⁺) stocks for the four sampling dates.

date	T°C	Rainfall	WFPS	SOM	NO ₃ ⁻	NO ₂ ⁻	NH ₄ ⁺
dd-mm-yy	°C	mm	%	%	kg N ha ⁻¹	kg N ha ⁻¹	kg N ha ⁻¹
14-May-12	14	0	54	2.2	17.35 ± n.d	0.18 ± n.d	39.36 ± n.d
26-Jun-12	16.5	0.3	61	3.5	16.92 ± 1.51	0.18 ± 0.01	19.08 ± 4.10
19-Sep-12	7.5	0	63	4.2	16.81 ± 7.33	0.17 ± 0.01	38.13 ± 0.20
14-Jan-13	-2	0	69	4.8	24.20 ± 8.53	0.48 ± 0.52	37.52 ± 0.07

4.4.2. Seasonal variation of N₂O production from nitrification and denitrification potentials at 20°C

N₂O production from nitrification and denitrification potentials were tested at 20°C in spring, fall and winter, all situations characterized by different *in situ* nutrient compositions and temperatures. Nitrification and denitrification potential rates (NO₃⁻_{nit} and NO₃⁻_{denit}, respectively) averaged 0.73 ± 0.31 µg N g⁻¹ dw h⁻¹ and 0.96 ± 0.57 µg N g⁻¹ dw h⁻¹, respectively (**Table 4-3**). Both NO₃⁻_{nit} and NO₃⁻_{denit} increased by a factor two from spring to winter, in relation with *in situ* initial conditions throughout the season, beneficial to the two processes, i.e., mineralization and an increase in SOM. Corresponding N₂O potential production rates at 20°C changed comparatively to nitrification and denitrification potential rates, and thus remained about two orders of magnitude higher for denitrification than for nitrification (mean, N₂O_{denit} = 0.11 ± 0.02 µg N g⁻¹ dw h⁻¹ and N₂O_{nit} = 0.003 ± 0.002 µg N g⁻¹ dw h⁻¹). Consequently, the N₂O/NO₃⁻ ratio of denitrification was also about two orders of magnitude higher than that related to nitrification whatever the seasonal conditions of this luvisolic soil (**Table 4-3**).

Table 4-3. Nitrification ($\text{NO}_3^-_{\text{nit}}$) and denitrification ($\text{NO}_3^-_{\text{denit}}$) potential rates at 20°C with their associated N_2O emissions and $\text{N}_2\text{O}/\text{NO}_3^-$ ratios (%).

date	$\text{NO}_3^-_{\text{nit}}$	$\text{N}_2\text{O}_{\text{nit}}$	$\text{N}_2\text{O}/\text{NO}_3^-$	$\text{NO}_3^-_{\text{denit}}$	$\text{N}_2\text{O}_{\text{denit}}$	$\text{N}_2\text{O}/\text{NO}_3^-$
dd-mm-yy	$\mu\text{g N g}^{-1} \text{ dw h}^{-1}$	$\mu\text{g N g}^{-1} \text{ dw h}^{-1}$	%	$\mu\text{g N g}^{-1} \text{ dw h}^{-1}$	$\mu\text{g N g}^{-1} \text{ dw h}^{-1}$	%
14-May-12	0.38	0.001	0.13	0.31	0.13	41.98
19-Sep-12	0.88	0.004	0.48	1.35	0.11	8.23
14-Jan-13	0.94	0.004	0.47	1.22	0.09	7.16

4.4.3. Effects of temperatures on N_2O production from nitrification and denitrification

Potential nitrification and denitrification activities, and their associated N_2O emissions, determined at the different temperatures have been summarized in **Table 4-4**. For all temperatures, concentrations of the analyzed nitrogen forms respectively increased or decreased linearly as a function of time for nitrification, denitrification, as well as for their associated N_2O emissions. This linearity supports the non-limiting substrates conditions set out.

Table 4-4 Means ($\pm$ SD) of nitrification ($\text{NO}_3^-_{\text{nit}}$), denitrification ($\text{NO}_3^-_{\text{denit}}$) potentials and their associated N_2O emissions and ratios for each temperature (T).

	$\text{NO}_3^-_{\text{nit}}$	$\text{N}_2\text{O}_{\text{nit}}$	$\text{N}_2\text{O}/\text{NO}_3^-$	$\text{NO}_3^-_{\text{denit}}$	$\text{N}_2\text{O}_{\text{denit}}$	$\text{N}_2\text{O}/\text{NO}_3^-$
T (°C)	$\mu\text{gN g}^{-1} \text{ dw h}^{-1}$	$\mu\text{gN g}^{-1} \text{ dw h}^{-1}$	%	$\mu\text{gN g}^{-1} \text{ dw h}^{-1}$	$\mu\text{gN g}^{-1} \text{ dw h}^{-1}$	%
5	0.20 $\pm$ 0.17	0.0004 $\pm$ 0.0003	0.22 $\pm$ 0.04	0.18 $\pm$ n.d	0.010 $\pm$ 0.005	7.55 $\pm$ n.d
10	0.23 $\pm$ 0.22	0.0006 $\pm$ 0.0004	0.29 $\pm$ 0.12	0.52 $\pm$ 0.57	0.029 $\pm$ 0.014	10.22 $\pm$ 8.54
16	0.43 $\pm$ 0.02	0.0024 $\pm$ 0.0016	0.58 $\pm$ 0.40	0.63 $\pm$ 0.16	0.082 $\pm$ 0.041	12.66 $\pm$ 3.32
20	0.73 $\pm$ 0.31	0.0030 $\pm$ 0.0022	0.36 $\pm$ 0.20	0.96 $\pm$ 0.57	0.109 $\pm$ 0.022	19.12 $\pm$ 19.80
30	1.47 $\pm$ 1.26	0.0201 $\pm$ 0.0184	1.92 $\pm$ 1.64	1.71 $\pm$ 1.27	0.302 $\pm$ 0.119	24.40 $\pm$ 18.27
37	1.16 $\pm$ 1.05	0.0550 $\pm$ 0.0092	7.40 $\pm$ 5.90	2.16 $\pm$ 0.10	0.369 $\pm$ 0.011	17.04 $\pm$ 0.24
45	0.80 $\pm$ 0.44	0.0638 $\pm$ 0.0566	7.11 $\pm$ 3.17	2.86 $\pm$ 0.38	0.567 $\pm$ 0.131	20.40 $\pm$ 7.27

According to the Gaussian models (Equation 4-15), both $\text{NO}_3^-_{\text{nit}}$ and $\text{NO}_3^-_{\text{denit}}$ increased with temperatures, from 5 °C up to an optimum of 32.5°C and 47°C, respectively (**Figure 4-1**). Similarly to our observations at 20°C, $\text{NO}_3^-_{\text{nit}}$ and $\text{NO}_3^-_{\text{denit}}$ remained within the same order of magnitude of values in the range of temperatures explored, the Gaussian fit leading to an optimum activity twice as high for denitrification, with $A_{\text{max}}(\text{NO}_3^-_{\text{denit}}) = 2.8 \mu\text{g N g}^{-1} \text{ dw h}^{-1}$ ($R^2 = 0.7$) and $A_{\text{max}}(\text{NO}_3^-_{\text{nit}}) = 1.4 \mu\text{g N g}^{-1} \text{ dw h}^{-1}$ ($R^2 = 0.4$) (**Table 4-5**). The nitrification

potential rate showed a decrease at temperatures higher than 32.5°C, but for denitrification the range of temperature tested (5- 45°C) was too low to observe a decrease, but the optimum temperature (47°C) was predicted by the curve fitting.

For temperatures lower than 20°C, no or low NO_2^- production was observed for both processes (**Figure 4-2**). Nitrite production started to be observed at 20°C but rapidly increased at higher temperatures. According to the adjustment to the Gaussian model, optimum temperatures of NO_2^- potential production rates occurred at 35°C for nitrification, equaling $0.2 \mu\text{g N g}^{-1} \text{dw h}^{-1}$, whereas for denitrification the optimum temperature was around 50°C with a rate of $1.8 \mu\text{g N g}^{-1} \text{dw h}^{-1}$. Ranges of NO_2^- potential production rates were ten times higher for denitrification than for nitrification.

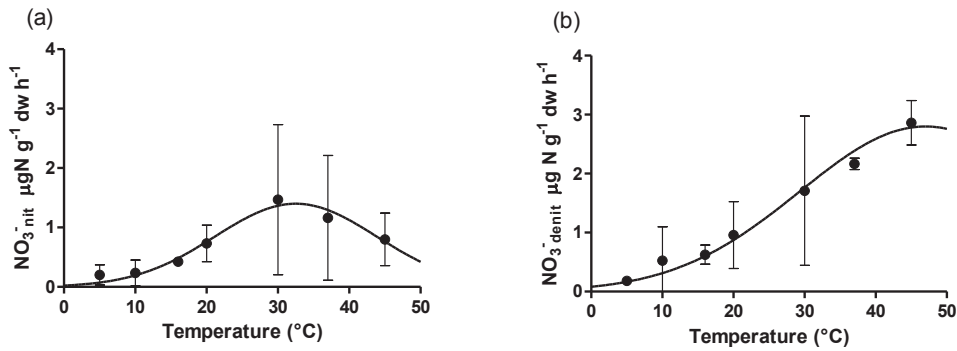


Figure 4-1. Nitrification (a) and denitrification (b) potential rates, i.e., nitrate production from nitrite oxidation during nitrification and nitrate reduction into nitrite from denitrification, with means and standard deviation (SD). The curve corresponds to the best fit of a sigmoid function based on all data observed, with $R^2 = 0.4$ for nitrification and $R^2=0.7$ for denitrification.

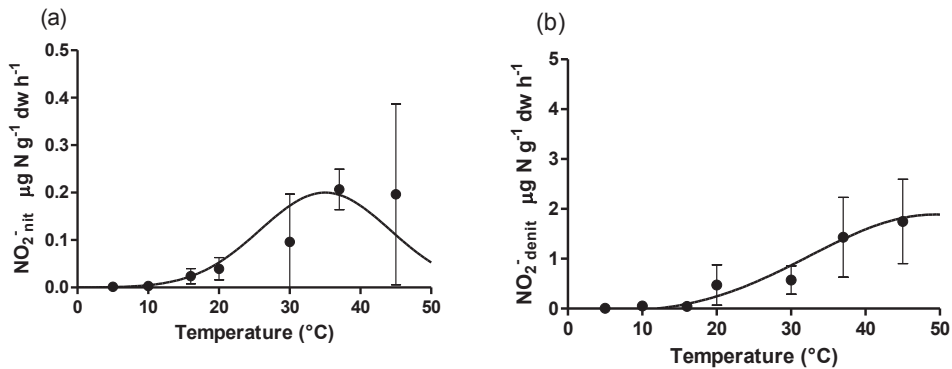


Figure 4-2. Nitrite net productions from ammonium oxidation during nitrification (a), i.e., n_T-m_T , and from nitrate reduction during denitrification (b), i.e., a_T-b_T , with means and SD. The curve corresponds to the best fit of a sigmoid function based on all data observed, with $R^2 = 0.3$ for nitrification (a) and $R^2=0.5$ for denitrification (b). Scale for denitrification is ten times greater than for nitrification

Nitrous oxide potential production rates also increased up to an optimal temperature both for nitrification and denitrification. For nitrification, the optimum temperature of N_2O_{nit} was 42.0°C, whereas for denitrification, the optimum of N_2O_{denit} reached 54.8°C, according to the Gaussian model (Table 4-3). For nitrification, N_2O was essentially produced starting at 30°C, below which no production was observed, in contrast to denitrification, which was associated with N_2O production for lower temperatures of 15–20°C. Indeed, N_2O_{nit} occurs in narrower range (dti = 11.5) than N_2O_{denit} (dti = 26.8) (Table 4-5). Starting from a difference by a factor of 100 at 20°C, the increase in the N_2O potential production rate was only higher by a factor of about 10 for denitrification (up to 0.64 $\mu g N g^{-1} dw h^{-1}$, $R^2=0.9$) than for nitrification (up to 0.06 $\mu g N g^{-1} dw h^{-1}$, $R^2=0.7$) at their respective optimum temperatures, as provided by the Gaussian adjustment.

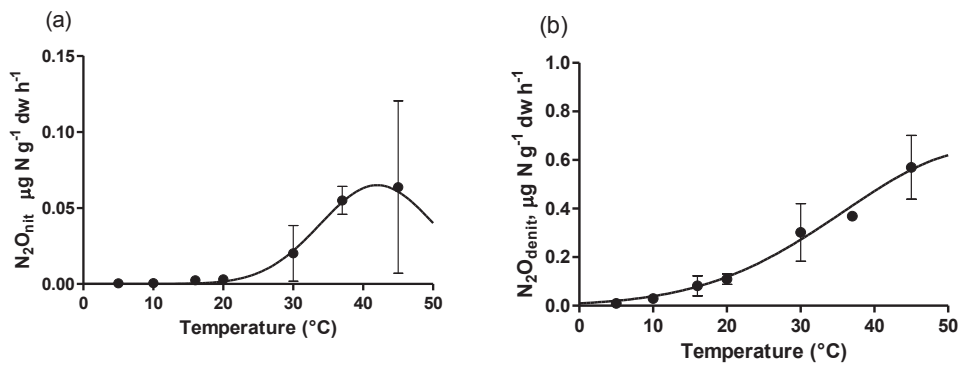


Figure 4-3. Nitrous oxide (N_2O) potential production rates from nitrification (a), i.e., p_T , and denitrification (b), i.e., b_T-c_T , with means and SD. The curve corresponds to the best fit of a sigmoid function, based on all data observed, with $R^2=0.7$ for nitrification (a) and $R^2=0.9$ for denitrification (b). Scale for N_2O produced by denitrification is ten times greater than N_2O produced from nitrification.

Table 4-5. Fitted parameters of the sigmoid Gaussian relationship to temperature of nitrification and denitrification: optimum activity (A_{max}), optimum temperature (T_{opt}) and σ value (dti). RMSD and R^2 of the fitted relationships.

Nitrification	A_{max} $\mu g N g^{-1} dw h^{-1}$	T_{opt} $^{\circ}C$	dti $^{\circ}C$	RMSD $\mu g N g^{-1} dw h^{-1}$	R^2
N_2O_{nit}	0.06	42.0	11.5	0.02	0.7
$NO_3^-_{nit}$	1.4	32.5	16.0	0.08	0.4
Denitrification	A_{max} $\mu g N g^{-1} dw h^{-1}$	T_{opt} $^{\circ}C$	dti $^{\circ}C$	RMSD $\mu g N g^{-1} dw h^{-1}$	R^2
N_2O_{denit}	0.64	54.8	26.8	0.06	0.9
$NO_3^-_{denit}$	2.8	47.0	25.0	0.13	0.7

The proportion of N_2O production compared to NO_3^- produced ($\text{NO}_3^-_{\text{nit}}$) or NO_3^- reduced ($\text{NO}_3^-_{\text{denit}}$) followed the same pattern as their respective N_2O emissions (**Figure 4-4**). The $\text{N}_2\text{O}/\text{NO}_3^-$ ratio of nitrification varied between 0.22 to 0.36% below 30°C and from 1.92 to 7.11% from 30°C to 45°C. For denitrification, the ratio increased from 7.55 to 24.40 % from 5° to 30°C and decreased above (**Table 4-4**).

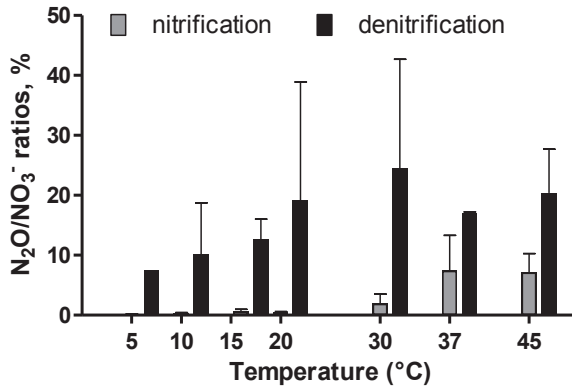


Figure 4-4. Mean ($\pm$ SD) of nitrification and denitrification $\text{N}_2\text{O}/\text{NO}_3^-$ ratios (%) for various temperatures from 5 to 45°C.

4.5. Discussion

4.5.1. Role of experimental conditions on nitrification and denitrification potentials

In our experiments, oxygen concentrations, pH, soil moisture and substrate conditions were set initially at the optimum values for nitrification and denitrification and remained so during the course of the experiment. In particular, oxic conditions were maintained in the slurry during the whole experiment, as shown by the measurements of O_2 at the end of the incubation time (showing a decrease by only 6 to 20 % after 4 h incubation at 20 and 45°C respectively, oxygen concentration remaining above $257.2 \text{ mgO}_2.\text{l}^{-1}$). Their respective substrates were in excess at initial conditions and throughout the incubation time, for all temperatures. During nitrification, the increase in NO_3^- was accompanied by a decrease in NH_4^+ , which never reached limiting concentrations, however. Similarly, NO_3^- was in excess during the denitrification process and was consumed during the experiment without reaching limiting concentrations. Soil pH was between 6 and 7 during the entire experiment, which is the optimum value found for nitrification activity ([Bramley and White 1989](#)) and denitrification ([Saleh-Lakha et al. 2009](#)).

Soil was taken in the first 20 cm, because this is where denitrification enzyme activity (DEA) and Nitrification Enzyme Activity (NEA) are at the highest concentration ([Dhondt et al. 2004](#)). We did not add carbon substrate, considering a sufficient initial stock, especially because slurries favor substrate availability to bacteria ([Laverman et al. 2010](#)) and organic matter was relatively high ($\text{SOM} > 2\%$). Again, linearity in the responses concerning the appearance or disappearance of NO_3^- and N_2O support the absence of a limitation.

The Gaussian model fitted well the responses of nitrification and denitrification as a function of temperature i.e., a similar increase with temperature up to an optimum and a decrease at higher temperatures. However, the three parameters (A_{max} , T_{opt} , d_{ti}) of the kinetics of each process strongly differed, with optimal temperatures and the sigma value of the responses to temperatures higher for denitrification ($T_{opt} = 47^{\circ}\text{C}$; $d_{ti} = 25^{\circ}\text{C}$) than for nitrification ($T_{opt} = 32.5^{\circ}\text{C}$; $d_{ti} = 16^{\circ}\text{C}$), probably due to the diversity of the bacterial communities, widely phylogenetically spread for denitrifying bacteria (Balser et al. 2006). However, for both communities a decrease in the activity at a high temperature ($\sim 45^{\circ}\text{C}$) would probably result from the integrity of the enzymatic reaction being affected. Similar optimal temperature have been found in different soils for nitrification potential around 30°C (Saad and Conrad 1993) or denitrification around 45°C (Fischer and Whalen 2005), although some studies mentioned the optimum temperature higher for denitrification, up to 60°C in silt loam soil (Keeney et al. 1979). Low temperatures ($5\text{--}10^{\circ}\text{C}$) led to very low $\text{NO}_3^-_{\text{nit}}$ and $\text{NO}_3^-_{\text{denit}}$ ($< 0.5 \mu\text{g N g}^{-1} \text{dw h}^{-1}$), as previously found by (Stanford et al. 1975).

4.5.2. Influence of temperature on nitrite production and nitrous oxide emissions

Both reactions of nitrification and denitrification produce NO_2^- as an intermediate, which rarely accumulate in the environment but can be enhanced by high temperatures (Kim et al. 2008). According to Saad and Conrad (1993), no NO_2^- production should occur at 20°C , which was nearly the case during our experiments ($\text{NO}_2^-_{\text{nit}} (20^{\circ}\text{C}) < 0.01 \mu\text{g N g}^{-1} \text{dw h}^{-1}$ and $\text{NO}_2^-_{\text{denit}} (20^{\circ}\text{C}) < 1 \mu\text{g N g}^{-1} \text{dw h}^{-1}$). Many studies demonstrate a faster turnover from nitrite oxidizing bacteria (NOB) than ammonia oxidation bacteria (AOB) at 20°C , which explains the rare accumulation in the terrestrial environment (Gelfand and Yakir 2008). However, the increase in $\text{NO}_2^-_{\text{nit}}$ at high temperatures may be due to the NOB activity decrease when temperature increased (Jones and Hood 1980; Kim et al. 2008) or even NOB inhibition, which consequently led to higher nitrous oxide emissions (Chalk and Smith 1983).

The levels of N_2O production we found at 20°C , were within the same range as those previously obtained for nitrifying ($0.001 \mu\text{g N g}^{-1} \text{dw h}^{-1}$) and denitrifying ($0.1 \mu\text{g N g}^{-1} \text{dw h}^{-1}$) in a similar luvisolic soil in the area (Vilain et al. 2014). Regarding the effect of temperature, specifically explored here, the high production of N_2O from denitrification, within the large range explored (cf. a high d_{ti}) is consistent with a higher adaptation of denitrifying bacteria to high temperatures, as reported in (Saad and Conrad 1993). In nitrification conditions, the response of bacterial activities in terms of N_2O was much narrow around the optimal temperature, exactly as nitrification itself, with a shift however: as shown by the Gaussian model, above $\sim 40^{\circ}\text{C}$ the potential production of N_2O decreased, whereas the potential rate of nitrification ($\text{NO}_3^-_{\text{nit}}$) decreased already from $\sim 30^{\circ}\text{C}$. As fully oxic conditions are still ensured at that temperature in the experiment, this indicates that the N_2O production as a byproduct of NH_4 oxidation is enhanced with respect to NO_2^- production. However, according to Khalil et al. (2004) and Kool et al. (2011) nitrifiers denitrification should not be completely excluded at the highest temperature due to the 20 % decrease of O_2 . At total, we show here that, similarly to denitrification, the ratio of N_2O emission to nitrification increases with high temperatures (from 0.2% at 5°C to 7.1% at 45°C). This results seems to differ to what has been observed by other authors, but their experiments were carried out in a much lower range of temperatures ($5\text{--}20^{\circ}\text{C}$) (Maag and Vinther 1996).

An important result gained here, with the exploration of a wide range of temperatures, is the relative increase of N_2O production in both nitrifying and denitrifying conditions at temperature above 20°C . Whereas up to $\sim 20^\circ\text{C}$, denitrification is the major producer of N_2O , a factor of 100 higher than for nitrification, the difference between the two processes is considerably reduced, to a factor of less than 10, beyond 20°C , suggesting possible high N_2O production either in wet (anaerobic) and dry (aerobic) conditions during heat wave events which have been occurring more frequently in the last decades (Eckard and Cullen 2011). Besides results obtained under laboratory fully controlled conditions, more investigations are needed for documenting long-term physiological adaptation of the microbial communities to changing temperature, in order to integrate spatial and temporal variability at the field scale.

4.6. Conclusion

Focusing on N_2O production associated to potentials of nitrification and denitrification in a gradient of temperatures and under optimal substrate conditions, we have determined the parameters of their relationship with temperature. Besides contributing more detailed knowledge on the production of N_2O , these relationships are necessary for taking into account nitrification, denitrification, N_2O emissions and nitrite accumulation in modeling approaches.

In nitrification and denitrification conditions, we found similar rates of potential activity for nitrate reduction and production, but wide differences are obtained regarding N_2O emissions. Whereas N_2O was essentially produced by denitrification below 20°C rather than by nitrification, the ratio of N_2O emitted per unit of nitrate reduced or produced, steadily increases with temperature. Above 20°C , the differences in N_2O production relatively to the two processes are reduced, from a factor of 100 to a factor of less than 10. Therefore, an increase of temperature could enhance N_2O emissions from ecosystems, due to the increase of absolute nitrification and denitrification rates and of their ratio $\text{N}_2\text{O}/\text{NO}_3^-$.

Acknowledgments

The study was supported by a PhD scholarship from the DIM Astrea of the Ile-de-France Region. The PIREN-Seine and EFFEMAIR- N_2O (Ademe) programs are also acknowledged for providing funding for the analysis. We also sincerely thank Benjamin Mercier and Abdelkader Azougui for their kind laboratory and field technical assistance. Thanks are also due to the farmer, Eric Gobard, who allowed us to work on his farm.

5

Curative vs. preventive management of nitrogen transfers in rural areas: lessons from the case of the Orgeval watershed (Seine River basin, France)

Garnier J^{1,2}, Billen G^{1,2}, Vilain G¹, Benoit M¹, Passy P^{1,2}, Tallec G³, Tournebize J³, Anglade J¹, Billy C³, Mercier B¹, Ansart P³, Azougui A¹, Sebilo M⁴, Kao C⁵

Ce chapitre est un article publié en juin 2014 dans la revue *Journal of Environmental Management* (doi: 10.1016/j.jenvman.2014.04.030).

¹CNRS UMR 7619 Metis, BP 123, Tour 56-55, Etage 4, 4 Place Jussieu, 75005 Paris, France

²UPMC, UMR 7619 Metis, BP 123, Tour 56-55, Etage 4, 4 Place Jussieu, 75005 Paris, France

³IRSTEA, UR HBAN, 1 rue Pierre-Gilles de Gennes, CS 10030, 92761 Antony, France

⁴UPMC UMR 7618 IEES, BP 120, Tour 56-66, Etage 4, 4 Place Jussieu, 75005 Paris, France

⁵AgroParisTech Centre de Paris – 19 avenue du Maine, 75732 Paris cedex 15, France

5.1. Abstract

The Orgeval watershed (104 km²) is a long-term experimental observatory and research site, representative of rural areas with intensive cereal farming of the temperate world. Since the past few years, we have been carrying out several studies on nitrate source, transformation and transfer of both surface and groundwaters in relation with land use and agriculture practices in order to assess nitrate (NO₃⁻) leaching, contamination of aquifers, denitrification processes and associated nitrous oxide (N₂O) emissions. A synthesis of these studies is presented to establish a quantitative diagnosis of nitrate contamination and N₂O emissions at the watershed scale. Taking this watershed as a practical example, we compare curative management measures, such as pond introduction, and preventive measures, namely conversion to organic farming practices, using model simulations. It is concluded that only preventive measures are able to reduce the NO₃⁻ contamination level without further increasing N₂O emissions, a result providing new insights for future management bringing together water-agro-ecosystems.

5.2. Introduction

In the early 20th century, the invention of the Haber-Bosh process allowing industrial production of mineral nitrogen (N), mostly used as fertilizers after World War II, profoundly changed agricultural practices (Davidson et al. 2012). Although agricultural productivity increased, providing food to the growing human population, the nitrogen cycle was widely opened, leading to severe environmental degradation (Sutton et al. 2011). The control of nitrogen pollution is therefore a major challenge in agricultural river basins (Billen et al. 2007; Grizzetti et al. 2012). Continental water masses (from lentic to lotic and from surface-to groundwater) are often substantially contaminated by nitrate (NO_3^-), causing major problems for drinking water supply (Ward et al. 2005) as well as for aquatic biodiversity (James et al. 2005). Moreover, nitrogen fluxes mostly originating from diffuse sources are delivered to the coastal zones in excess with regard to other major nutrients such as silica and phosphorus, possibly participating in eutrophication problems caused by harmful algal blooms with damage to various economic activities (fisheries, tourism, etc.) (Cugier et al. 2005; Howarth et al. 2011; Lancelot et al. 2011; Romero et al. 2013).

In many intensive agricultural areas, such as the Paris Basin, inorganic nitrogen applied as fertilizers to arable soil exceeding the amount exported by crop harvesting, are leached to surface water and aquifers. NO_3^- can also be denitrified in soils and riparian zones (Haycock and Pinay 1993; Billen and Garnier 1999; Burt et al. 2002; Rassam et al. 2008) as well as river and pond sediments (Garnier et al. 2000; Tomaszek and Czerwieniec 2000; David et al. 2006; Gruca-Rokosz and Tomaszek, 2007; Garnier et al. 2010; Passy et al. 2012) before ultimately reaching the coastal zone. The process of denitrification, at every stage of the nitrogen cascade, thus represents a natural mechanism of elimination of NO_3^- contamination, re-injecting nitrogen into the pool of inert atmospheric di-nitrogen. However, during this process, nitrous oxide (N_2O) is produced as an intermediate, which is emitted into the atmosphere, particularly under suboptimal conditions of carbon (C) and nitrogen substrate concentrations (Knowles 1982; Tallec et al. 2006; Saggar et al. 2012). A budget made at the scale of the Seine Basin showed that agricultural soils are dominant contributors of the overall N_2O emission budget (Garnier et al. 2009). N_2O is a powerful greenhouse gas, also contributing to the destruction of the stratospheric ozone layer, and the increase of its emission, possibly related to increased NO_3^- use in agriculture or to remediation actions aimed at eliminating NO_3^- from water through denitrification, is a matter of serious concern. Whereas the application of Urban Wastewater Directive (UWWD, 1991) and Water Framework Directive (WFD, 2000) have already contributed to a quite significant reduction in phosphorus load, much is expected for nitrogen reduction from changes in the Common Agricultural Policy (CAP) encouraging "greening" practices (EU, 2013). The small Orgeval watershed ($\approx 100 \text{ km}^2$) is representative of the dominant landscape of the central Seine Basin ($\approx 76,000 \text{ km}^2$ at the entrance of the estuary) characterized by an intensive cereal crop belt surrounding the large Paris conurbation, which has completely shaped its hinterland during historical periods (Billen et al. 2009a, 2013; Barles, 2010).

The Orgeval watershed is a long-term experimental observatory and research site initiated in the early 1960s by IRSTEA, the French National Research Institute of Science and Technology for the Environment and Agriculture. Whereas early research was mostly dedicated to the issues of hydrology and agricultural drainage, with the intensification of cereal cropping at the expense of cattle breeding, attention has been progressively paid to

water quality issues, especially because the aquifers of the Orgeval watershed contribute to the production of drinking water for the city of Paris.

In this paper, we present a synthesis of the long-term field and modelling research carried out in this watershed, with the aim of making a diagnosis of the sources of nitrogen contamination, its transfer and transformation processes at the catchment scale. We then explore, using the GIS-based modelling approach developed for the Seine basin (Seneque-RiverStrahler, Ruelland et al. 2007; Billen et al. 2009b), several management options for decreasing nitrogen contamination of surface and groundwater, with particular emphasis on the risk of pollution swapping between water NO_3^- contamination and increased N_2O emission.

Although we use the Orgeval watershed as a practical well documented case study in which a fully detailed modeling exercise can be carried out, the scope of the results obtained, largely encompasses this particular study site and the conclusions are of general relevance for all rural areas with intensive industrial crop farming.

5.3. Site studied and methods

5.3.1. Characteristics of the Orgeval watershed

The Orgeval watershed is located 70 km East of Paris (France) and is a small sub-catchment covering 104 km² in the Marne sub-basin of the Seine River upstream from Paris (**Figure 5-1**).

The climate is semi-oceanic, with annual rainfall about 700 mm and a mean annual air temperature around 10°C (varying from 0.6 to 18° seasonally).

The Orgeval watershed is highly homogenous in terms of pedology, climate and topography (mean altitude, 148 m, with few slopes except in the valleys). The Orgeval watershed is covered with a 10-m loess layer, under which two tertiary aquifer formations are separated by a discontinuous grey clay layer (Mégnyen, 1979). The shallowest aquifer of the Brie Limestone Oligocene formation, with more interactions with surface waters, has a relatively shorter water residence than the deepest Champigny Limestone Eocene aquifer. The lower layer of the surface loess cover is enriched with clay, resulting in waterlogged soils in the winter. For this reason, up to 90% of the arable soils of the Orgeval watershed have been artificially tile-drained since the early 1960s. Land use is mostly agricultural land (82%), dominated by cereal crops (wheat, maize, barley and pea), with conventional practices, mainly based on mineral nitrogen fertilization. The remaining surface is covered by woods (17%) and urban zones or roads (1% of the surface) (**Figure 5-1**).

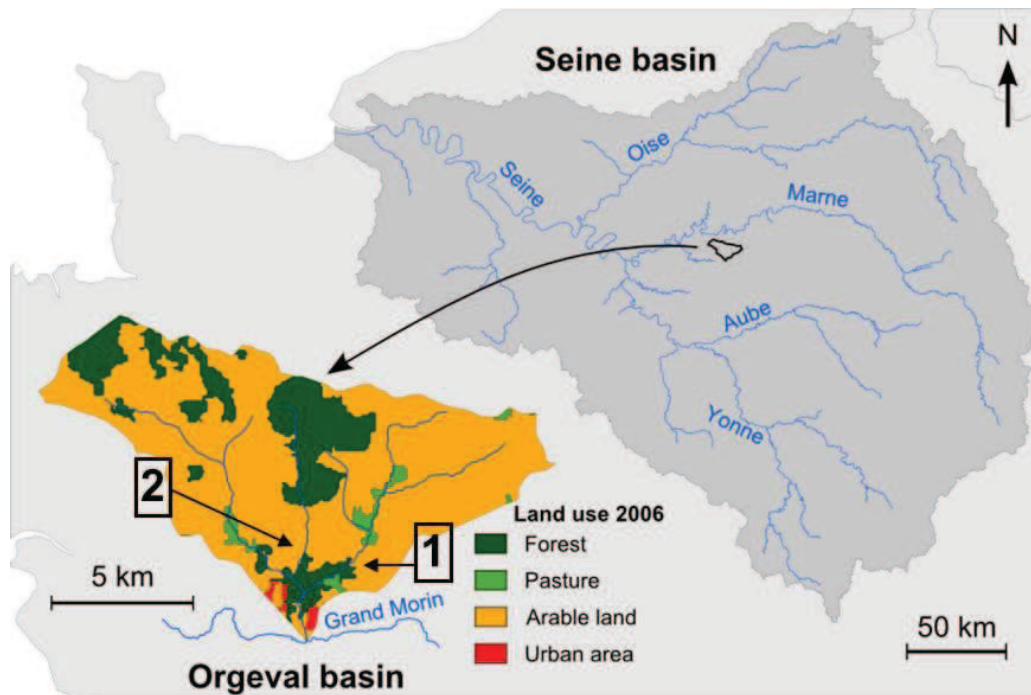


Figure 5-1. Location of the Orgeval Basin in the Seine Basin and the two sites studied.

5.3.2. Sampling and field studies, lab experiments and chemical analysis

Within the Orgeval watershed, series of nitrogen measurements (mainly nitrate as well as dissolved N_2O) have been carried out at least since 2005 on surface waters. Two specific sites have been equipped (Site 1 since 2007, Site 2 since 2011) for water table NO_3^- and N_2O dissolved concentration and for N_2O emissions from agricultural soils. A farm drainage pond was also sampled.

Surface water. NO_3^- concentrations were weekly measured since 1975 at the Mélarchez station (order 1) and, since 2005 at the outlet of the Avenelles sub-watershed and the Orgeval one (Le Theil station) in the framework of IRSTEA routine programme. Dissolved N_2O in surface water have been measured from 2006 to 2008 at monthly intervals at the same three sampling stations (partly in Vilain et al. 2010; Vilain et al. 2012c) (**Figure 5-1**).

Water table. On site 1 (**Figure 5-1**), three piezometers were installed along a slope from the plateau to the riparian zone in January 2007. This 6% inclination slope oriented northwestward reaches the Avenelles River. This site is typical of the whole Orgeval watershed both in terms of agricultural practices (grain crop with wheat, barley and maize as the main rotation) and fertilizer applications (from 120 to 160 $kgNha^{-1}$ for wheat/barley, to 180 $kgNha^{-1}$ for maize). Three piezometers were also installed in July 2011 in site 2. The piezometers were sampled for NO_3^- and N_2O determination in the Brie aquifer since their installation.

Agricultural soils. Suction ceramic cups were also installed on site 1 (**Figure 5-1**) during two winter drainage periods (January to March 2010 and December 2012 to April 2013) to quantify the sub-root NO_3^- concentrations for a conventional agricultural system. Other data were obtained at site 2 (in the winters 2012 and 2013) for an organic agricultural system and are used for the characterisation of organic agriculture scenarios (see below).

On site 1 along the piezometric slope, hermetically closed chambers (open bases measuring $50 \times 50 \times 30$ cm) allowed quantifying N_2O emissions (see [Vilain et al. 2010](#)) from cropping soil according to the methodology described by [Hutchinson and Livingston \(1993\)](#) and [Livingston and Hutchinson \(1995\)](#). Measurements were taken at different topographical landscape positions from the uphill to the riparian position from May 2008 to July 2009; a forested soil was investigated for comparison.

$\delta^{15}\text{N}$ -isotopic measurements in the soil organic matter were taken along two transects at six different locations on one occasion in March 2007 ([Billy et al. 2011](#)). For each transect, soil was sampled at 10-cm intervals from the surface to 90 cm deep. Air-dried and sieved (2 mm), the soil samples were homogenized prior to organic N isotopic composition analysis. These measurements were used as an integrated estimator of long-term soil denitrification processes.

To pursue the determination of the source of N_2O emissions in greater detail, soils sampled between 2009 and 2011 at several periods of the season, from the same site 1 cropped slope were incubated in batch experiments under optimal laboratory conditions (nutrients, temperature). Since N_2O is known to originate from nitrification and denitrification, both processes were investigated. As described in [Garnier et al. \(2010\)](#) and [Vilain et al. \(2012b\)](#), batch experiments were run and the NO_3^- , NO_2^- , NH_4^+ concentrations followed during a short incubation time (4–6 h), to avoid any confinement in the flasks, in triplicate and in the dark. For nitrification assays, ammonium was added and the flasks were flushed with ambient air to ensure aerobic conditions, while for denitrification assays, NO_3^- was added and the flask was flushed with N_2 in order to produce anaerobic conditions. Production of N_2O associated with the processes was also measured.

Farm drainage pond. A drainage farm pond on site 2 (**Figure 5-1**) was also investigated over 3 years for NO_3^- concentrations (2007 to 2010) in order to evaluate the pond's potential for eliminating nitrogen leached from agriculture ([Passy et al. 2012](#)). N_2O concentrations dissolved in the water column were determined seasonally in 2010, allowing to estimate emissions ([Garnier et al. 2009](#)).

Analytical methods

Analytical methods for NO_3^- and N_2O concentrations in water are described in [Jones \(1984\)](#) and [Garnier et al. \(2009\)](#), respectively. N_2O concentrations in gas sample were analysed by gas chromatography, as described by [Vilain et al. \(2010\)](#). Measurement of organic N isotopic composition of the soil is described by [Billy et al. \(2010\)](#).

5.3.3. Simulating N reduction measures

The biogeochemical model (RiverStrahler) describing the ecological functioning of aquatic systems ([Billen et al. 1994](#); [Garnier et al. 2002](#)), currently implemented at the scale of the Seine Basin embedded in the GIS-Seneque interface tool ([Ruelland et al. 2007](#); [Thieu et al.](#)

2009; Passy et al. 2013) has been used here for exploring scenarios of mitigating measures at the scale of the Orgeval watershed. The principle of the model is illustrated in **Figure 5-2**.

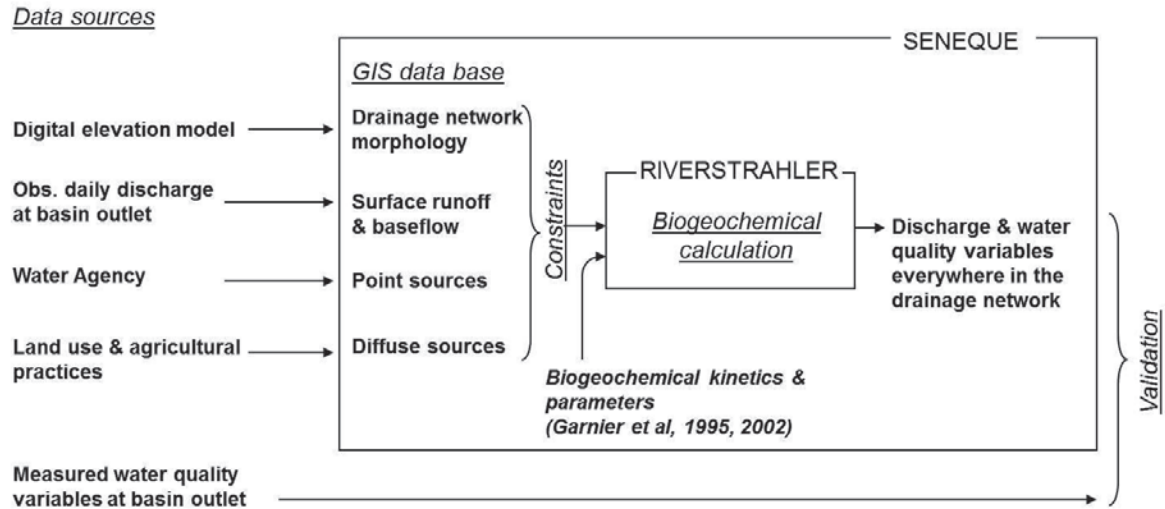


Figure 5-2. Representation of the Seneque/Riverstrahler model.

5.4. Quantifying the N cascade through the Orgeval watershed

5.4.1. N leaching from agricultural soils to sub-root water, tile-drains and aquifers

Wheat, maize, pea and barley cover around 44, 14, 6 and 4%, respectively, of the cultivated area in the Orgeval watershed (RGA-Recensement Général Agricole, 2000). The main crop rotations are wheat-pea-wheat (28%) and maize-winter wheat-spring barley (20%), with a mean crop yield of about 5500 kg cereal equivalent per ha, corresponding to about 100 kgN ha⁻¹ yr⁻¹. The fertilizer application rate ranges from 120 to 180 kgNha⁻¹ yr⁻¹. Atmospheric deposition of N adds around 15 kgN ha⁻¹ yr⁻¹ and atmospheric N₂ fixation (through non-symbiotic fixation and by legume crops in some rotations) about 10 kgN ha⁻¹ yr⁻¹ (Billy et al. 2010). The soil N balance thus reveals a long-term surplus of about 50 kgNha⁻¹ yr⁻¹.

Sub-root concentrations measured from 2010 to 2013 with suction cups installed 1 m deep under representative arable plots average 22 mgNO₃-N L⁻¹ (SD=15). This value is close to the average concentration observed in tile drains in the same area (26 mgNO₃-N L⁻¹) (**Figure 5-3**). These sub-root concentrations are quite similar to those observed elsewhere in the Seine Basin in the 1990s. Indeed, in the chalky Champagne, East of Paris, the concentrations obtained were 27.2 mgNO₃-N L⁻¹ for a 10-year wheat/beet rotation but significantly less with the introduction of alfalfa in the rotation (20.8 mgNO₃-N L⁻¹) (Beaudoin et al. 1992). Similar figures were found in the Northern or Western sectors of the Seine Basin, i.e., respectively, 19 mgNO₃-N L⁻¹ (Machet and Mary, 1990) and 29 mgNO₃-N L⁻¹ (Arlot and Zimmer, 1990).

With an average discharge of $0.36 \text{ m}^3 \text{ s}^{-1}$ at the outlet of the Orgeval watershed, a yearly N leached flux can be estimated to $2400 \text{ kg km}^{-2} \text{ yr}^{-1}$ (50% variation).

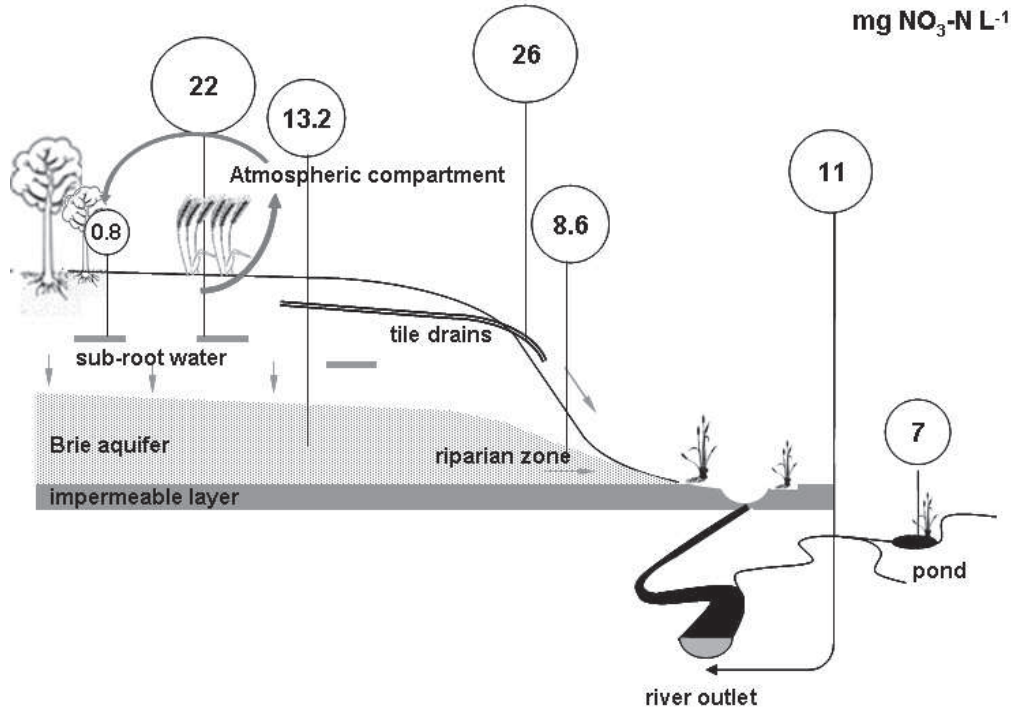


Figure 5-3. Concentrations of nitrate cascading within the Orgeval basin (see text for explanations, unit in mg N L⁻¹).

NO₃⁻ concentrations in the Brie aquifer, measured from samples collected in the piezometers installed uphill, are around $13.2 \text{ mg NO}_3\text{-N L}^{-1}$. Samples collected midslope or below the riparian buffer strip show 35–40% lower concentration, down to $8.6 \text{ mg NO}_3\text{-N L}^{-1}$ (**Figure 5-3**), probably because of denitrification processes occurring when the water table reaches the biogeochemically active upper soil layers. In the pond studied, the average annual concentration was even lower ($7 \text{ mg NO}_3\text{-N L}^{-1}$), compared to the average concentration entering the pond ($15 \text{ mg NO}_3\text{-N L}^{-1}$). At the outlet of the Orgeval watershed, the average river water concentration is $11 \text{ mg NO}_3\text{-N L}^{-1}$.

5.4.2. Denitrification and N₂O emissions in soils along a cropped slope

Both nitrification and denitrification in soil are able to produce the greenhouse gas N₂O, particularly under suboptimal conditions (limitation by substrates, oxygen tension, pH, temperature, etc.) (Firestone and Davidson 1989), although several other microbial processes are able to consume the N₂O emitted (e.g. nitrifier denitrification (Wrage et al. 2001), dissimilatory NO₃⁻ reduction to ammonia (Burgin and Hamilton 2007), anammox in specific conditions (Dalsgaard et al. 2005; Dalsgaard et al. 2013). In the same line as the

research on wastewater treatment plants (Tallec et al. 2006), the relative magnitude of nitrification or denitrification in the emission of N_2O was experimentally explored in Orgeval watershed soil samples (Vilain et al. 2012b; Vilain et al. 2014). It appeared that potential rates of NO_3^- production (nitrification) and NO_3^- reduction (denitrification) were, on average, within the same range ($0.8\text{--}0.9 \mu\text{g } NO_3\text{-N g}^{-1} \text{ dw h}^{-1}$), but the associated potential N_2O production was much lower (by a factor of 100) for nitrification than denitrification (Table 5-1), corroborating previous findings by Tallec et al. (2006). The ratio of N_2O production to NO_3^- reduction was up to 20% for the denitrification potential, while the ratio of N_2O emission to NO_3^- production by nitrification was only about 0.2%.

Table 5-1. Average potential values for agricultural soils in denitrification and nitrification in experimental conditions (batch experiments at 20°C), and associated N_2O production (SD for Standard Deviation, 7 experiments). Percentages of N_2O production are also given for comparison.

	Potential NO_3^- production/reduction rates $\mu\text{g } NO_3\text{-N g}^{-1} \text{ dw h}^{-1}$	Potential N_2O production rates $\mu\text{g } N_2O\text{-N g}^{-1} \text{ dw h}^{-1}$	N_2O/NO_3^- ratios %
Denitrification	0.89 (SD=0.47)	0.15 (SD=0.08)	24.4 (SD=20.7)
Nitrification	0.81 (SD=0.271)	0.002 (SD=0.001)	0.18 (SD=0.16)

Direct in situ measurements of N_2O emissions by agricultural and forest soil using closed chambers were taken on 21 dates from May 2008 to August 2009 (Vilain et al. 2010; Vilain et al. 2012a). For uphill plateau sites, a value equalling $0.29 \text{ mg } N_2O\text{-N m}^{-2} \text{ d}^{-1}$ was estimated for cropland, higher than the average one found for forested soils: $0.15 \text{ mg } N_2O\text{-N m}^{-2} \text{ d}^{-1}$.

Higher values, close to $0.41 \text{ mg } N_2O\text{-N m}^{-2} \text{ d}^{-1}$ were measured in downslope sites, with the level of the water table closer to the soil surface. N_2O emissions, averaged for footslope and riparian zone was $0.61 \text{ mg } N_2O\text{-N m}^{-2} \text{ d}^{-1}$ (Figure 5-4 a). These results show increasing transformation of nitrogen (denitrification mainly) along the slope, and concomitant increasing N_2O emission.

$\delta^{15}\text{N}$ fractionation values of soil organic nitrogen along a cropped slope and averaged over a 1-m soil profile, were higher than the primary nitrogen (N) sources from which they are derived, such as mineral nitrogen fertilizers, atmospheric deposition and symbiotic N_2 (all characterized by $\delta^{15}\text{N}$ values close to zero), indicate indeed the existence of a long-term denitrification process (Billy et al. 2010; Vitousek, 2013). Based on a modelling approach of the isotopic composition of the soil N compartment, Billy et al. (2010) estimated that a 1‰ $\delta^{15}\text{N}\text{-Norg}$ increase above that of the primary N sources corresponds to a denitrification of $\sim 10 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ (e.g. $2.7 \text{ mg N m}^{-2} \text{ d}^{-1}$) which confirm the prevalence of denitrification.

The distribution of $\delta^{15}\text{N}$ of the bulk soil N pool from the uphill plateau down to the riparian zone of the river shows a regular increase from 2.4‰ in plateau forested soils and 5.8‰ in crop soil, to 7.4‰ in the downslope arable soil and in the buffer strip results well in agreement with N_2O emission from denitrification (Figure 5-4 b).

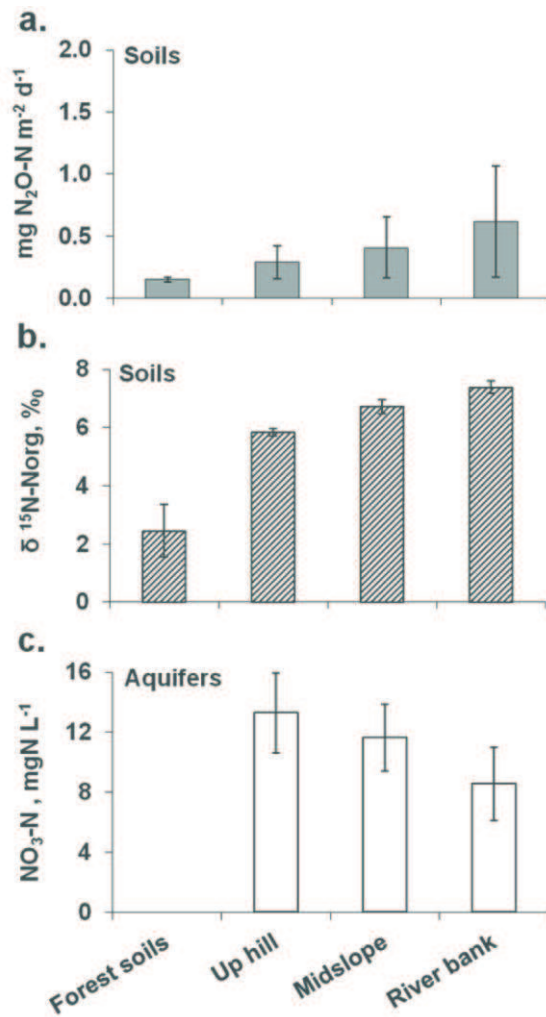


Figure 5-4 a. Seasonal average of N_2O emission from soils in a forest area and an agricultural slope, redrawn from Vilain et al. (2010). b. Variations of $\delta^{15}\text{N}$ of nitrogen organic matter average over a 1-m soil profile (b), recalculated from Billy et al. (2010). c. Seasonal averages of $\text{NO}_3\text{-N}$ concentrations in the water of the brie aquifer as sampled in the piezometers along the slope, modified from Vilain et al. (2012c).

N_2O concentration in the aquifer was also measured by sampling the piezometers. The values found were largely over-saturated ($20 \mu\text{g N}_2\text{O-N L}^{-1}$ on average), taking into account that N_2O saturation in water with respect to the atmospheric level of 330 ppb varies from 0.35 to $0.5 \mu\text{g N}_2\text{O-N L}^{-1}$ depending on the temperature (Figure 5-4 c). We interpreted these high N_2O values in the aquifer as resulting from leaching from the root zone, although denitrification and N_2O production in the aquifer itself is not fully excluded, critical oxygenation around $2\text{--}3 \text{ mg O}_2 \text{ L}^{-1}$ being occasionally observed (Vilain et al. 2012a). The lower N_2O concentrations in the downslope sites can be explained by microbial transformation into N_2 , e.g. again corroborating a complete denitrification along the slope. N_2O degassing from the aquifer along the underground flow, i.e. indirect N_2O emissions, is not excluded.

5.4.3. In-stream N elimination processes

Direct measurement with bell-jars allowed estimating the rate of benthic denitrification in river sediments. Consumption rates on the order of 3.1 ($SD=1.1$) $\text{mg N m}^{-2} \text{ h}^{-1}$ were observed (Thouvenot-Korppoo et al. 2009; Billy et al. 2011). Considering a river bottom area of about $175,830 \text{ m}^2$ for the Orgeval watershed as a whole, this leads to a maximum estimate of $3000\text{--}6000 \text{ kgN yr}^{-1}$ for benthic denitrification ($30\text{--}60 \text{ kgN km}^{-2} \text{ yr}^{-1}$ at the watershed scale), showing that in-stream processes represent a marginal value in the nitrogen elimination of the $2400 \text{ kgN km}^{-2} \text{ yr}^{-1}$ found at the base of the root zone.

Accordingly, N_2O concentrations, above saturation, observed in small rivers of the Orgeval watershed, are inherited from the groundwater feeding them, instead of being produced through in-stream processes. Indeed, these concentrations rapidly decrease from the spring downwards until reaching saturation (Garnier et al. 2009).

5.4.4. A synthetic budget of N transfers in the Orgeval watershed

Based on the data summarized in the above paragraphs, a tentative budget of nitrogen transfer at the scale of the Orgeval watershed was established (Figure 5-5), describing the fate of NO_3^- mostly coming from the surplus nitrogen left by agricultural soils. Denitrification in the soil profile and in the downslope areas (where a temporarily or permanently shallow water table comes in contact with the upper biogeochemically active layers of the soil) eliminates more than 40% of the nitrogen leaving the root zone.

The various denitrification figures in this budget are in good agreement with the values found (i) for soil denitrification (Pinay et al. 1993; Hefting et al. 2006), (ii) for the riparian zones (Billen and Garnier 1999) and (iii) for in-stream benthic denitrification at the scale of the whole Seine hydrographic network (Thouvenot-Korppoo et al. 2009).

On the basis of (i) the N_2O emissions from soils together with a fine resolution of the topography and land use in the watershed, (ii) the N_2O fluxes from rivers and groundwater deduced from concentration measurements (Garnier et al. 2009; Vilain et al. 2010; Vilain et al. 2012c), the total N_2O emissions for the whole Orgeval watershed were estimated at $142 \text{ kgN}_2\text{O-N km}^{-2} \text{ yr}^{-1}$ (Vilain et al. 2012a). This represents about 10% of the sum of the denitrification rates occurring in soils, footslopes and riparian zones and in-stream sediments (see Figure 5-5 a). This N_2O percentage emission is in agreement (within a factor of 2) with the potential values found experimentally for denitrification.

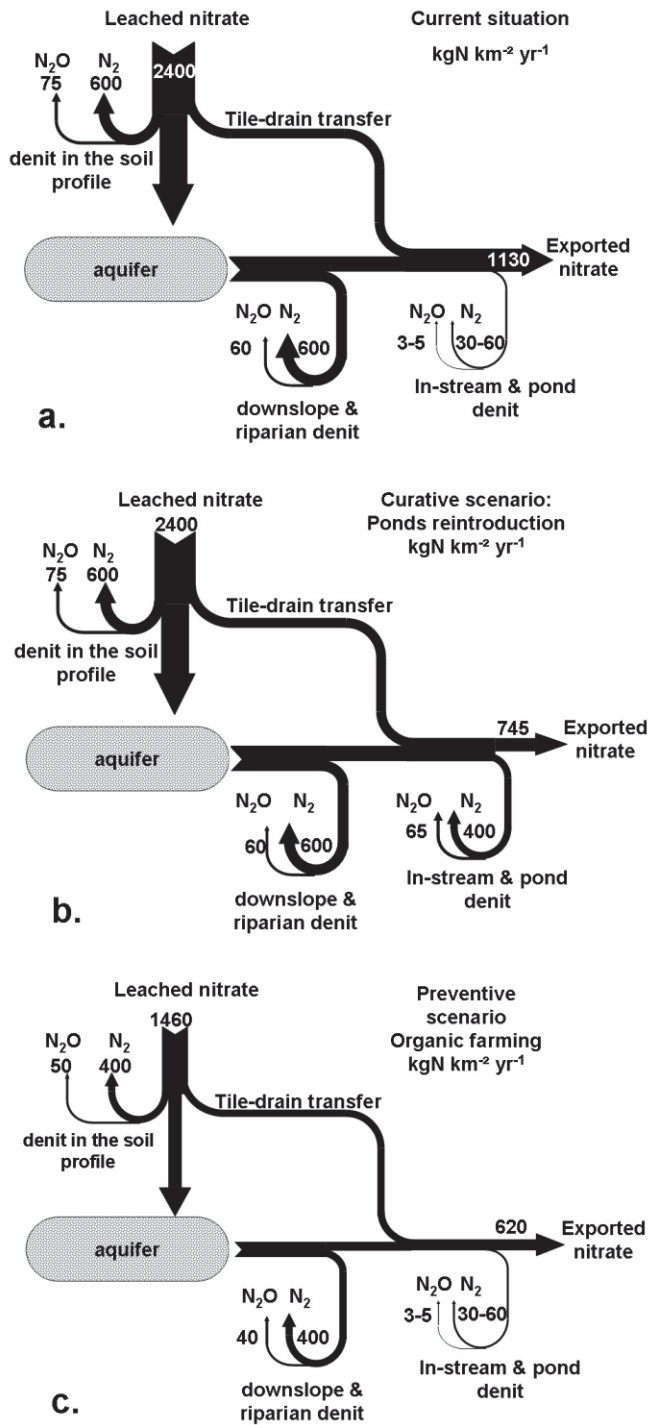


Figure 5-5. Summarizing budget of nitrate transfer and transformation and associated nitrous oxide emissions in the Orgeval Basin. Calculations are based on the average hydrology from 2006 to 2012. a) current situation based on measurements; b) scenario of pond reintroduction; c) scenario of organic farming.

5.5. Curative management measures to reduce NO_3^- contamination

Drainage or irrigation water retention ponds are often seen as buffer interfaces where N elimination is effective. The creation of such systems is often considered within the framework of compensatory measures, possibly included in the wetland status (Dahl 2011). In addition, these waterbodies can be viewed as anthropogenic refuge for biodiversity (Chester and Robson 2013).

5.5.1. NO_3^- and N_2O concentrations in an artificial pond

We investigated such a pond established at the outlet of a tile drain collector draining 35 ha of cultivated land. Its surface area is 3700 m², with a volume of 8000 m³ (i.e. a mean depth of about 2 m). The concentrations at the entrance of the pond averaged 13.5 mg $\text{NO}_3\text{-N L}^{-1}$ (Figure 5-6 a) over the period studied, close to the value found for the concentration in the Brie aquifer (see Figure 5-3). NO_3^- concentrations in the pond show a systematic summer decrease, down to 1.5 mg $\text{NO}_3\text{-N L}^{-1}$ in late summer (annual mean, 7 mg $\text{NO}_3\text{-N L}^{-1}$). These values are accurately reproduced by a simplified model of stagnant water (Garnier and Billen 1993; Garnier et al. 2000; see also Passy et al. 2012) (Figure 5-6 a).

Regarding N_2O concentrations, the values averaged 3.8 $\mu\text{g N}_2\text{O-N L}^{-1}$, i.e. a tenfold over-saturation (with extreme concentrations of 8.4 and 1.1 $\mu\text{g N}_2\text{O-N L}^{-1}$ for a data series in 2010, $n=14$). Based on the saturation concentration (Weiss and Price 1980) and the gas transfer coefficient of 0.4 m h⁻¹ (Wanninkhof 1992; Borges et al. 2004), the annual mean N_2O emissions at the pond surface can be estimated at 3.4 mg $\text{N}_2\text{O-N m}^{-2} \text{ d}^{-1}$, a value similar to the emission at the cropped downslope (see Figure 5-4).

The observed decrease in NO_3^- concentrations in the pond during the period of high biological activity suggests that such ponds could effectively be used as curative management infrastructures for NO_3^- reduction in surface water. However, the concomitant outgassing of N_2O represents a serious limitation, as it can result in the simple swapping from one type of pollution to another.

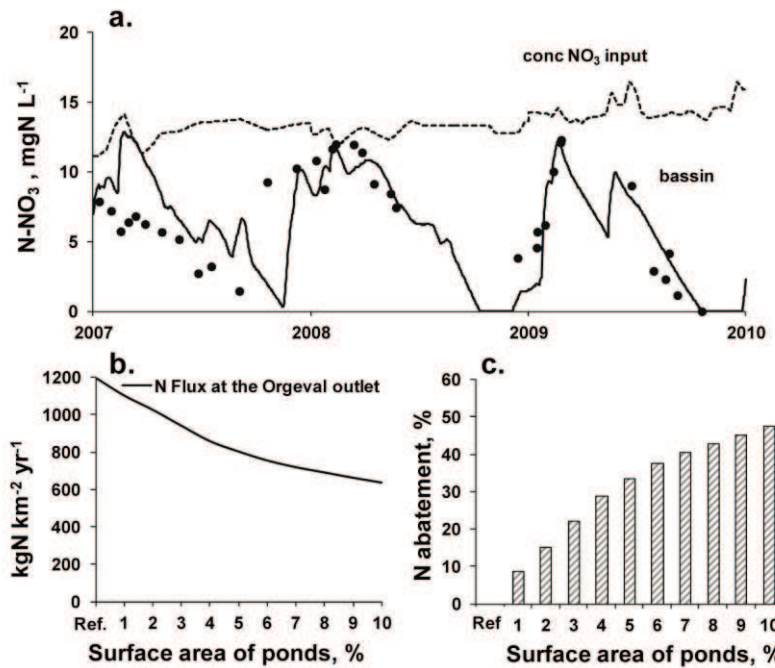


Figure 5-6 **a.** Interannual $\text{NO}_3\text{-N}$ concentrations in a drainage pond in the Orgeval watershed. Dotted line: $\text{NO}_3\text{-N}$ concentration at the entrance; solid line: simulated $\text{NO}_3\text{-N}$ concentrations in the pond; black dots are the measured NO_3 concentrations. **b.** Simulated N fluxes at the outlet of the Orgeval watershed with a range of surface area of ponds (from the reference situation to 10 % of the total surface area of the Orgeval watershed); **c.** Associated N abatement is shown in comparison (recalculated from [Passy et al. 2012](#)).

5.5.2. Simulation of the effect of pond creation at the scale of the Orgeval watershed

Interestingly, historical maps of the Orgeval area (e.g. the so-called Cassini map, dating back to the middle of the 18th century) reveal that the traditional landscape of the Brie region was characterized by a large number of ponds established on the headwaters, both for driving mills and for pisciculture. In the Orgeval watershed, the number of ponds was in the range of 60, and their surface area amounted to 1% of the total surface area of the watershed ([Passy et al. 2012](#)). Most of these ponds were dried and converted to cropland during the first half of the 19th century.

In order to explore the role of pond implementation in the Orgeval watershed as a measure to reduce the nitric contamination of surface water, the Seneque/RiverStrahler model ([Ruelland et al. 2007](#); [Thieu et al. 2009](#); [Passy et al. 2013](#)) was run, and connected drainage ponds were virtually introduced at different surface areas ([Passy et al. 2012](#)). The results showed that a 34% and 47% reduction of the N flux at the outlet of the Orgeval watershed can be expected with a total surface area of ponds equalling 5% and 10% of the watershed, respectively, compared to 9% abatement with the 1% pond coverage of the Cassini map (**Figure 5-6 b, c**). Reintroducing ponds in the landscape necessarily increases the residence

time of the water masses, increases the primary production providing more carbon for denitrification, for example. However, although possibly a refuge for biodiversity, e.g. for fish to feed and spawn, a shift from lotic to lentic species can be damageable.

Whereas the process of denitrification could be used for mitigation measures in combatting nitric contamination in the hydrosystems by creating or restoring wetlands, caution must be taken to limit a shift from nitric to N_2O pollution. Considering the N_2O emitted in the experimental pond studied, an increase of the N_2O emission to about $60 \text{ kgN}_2\text{O-N km}^{-2} \text{ yr}^{-1}$ by the Orgeval catchment could be expected in the case of 5% pond area, close to the emission by agricultural soils (see **Figure 5-5 b**). However due to contradictory results (cf. [Welti et al. 2012](#)), a comprehensive assessment of ecosystem services and disservices in agricultural landscapes remains a challenge ([Burgin et al. 2013](#)).

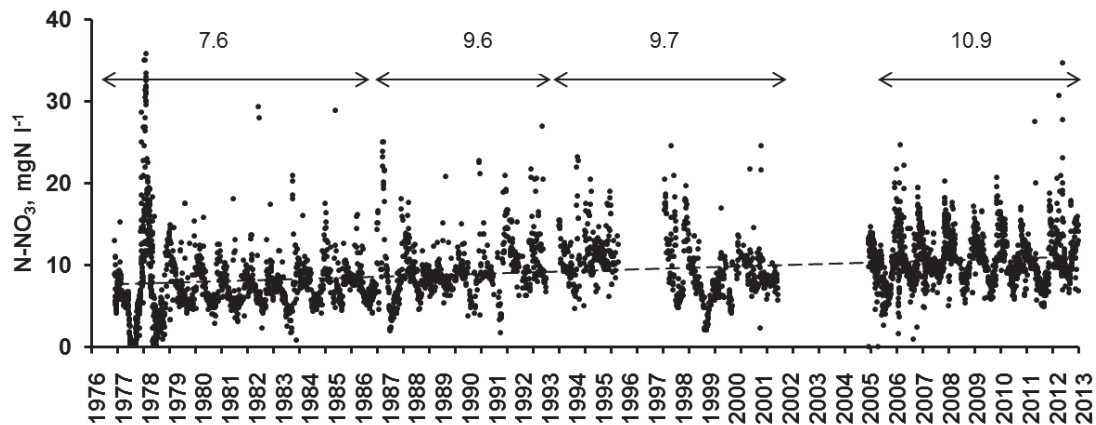


Figure 5-7. Long-term chronicle of observed $\text{NO}_3\text{-N}$ concentrations in the Melarchez River, a headwater stream in the Orgeval watershed.

5.6. Preventive management measures to reduce nitrogen contamination

5.6.1. Good Agricultural Practices

Good Agricultural Practices, consisting in lowering and fractionation of N fertilization, return of crop residues to the soil and introduction of catch crops, were promoted in the 1990s. When correctly applied, these measures are able to significantly reduce N leaching ([Beaudoin et al. 2005](#)). The long-term chronicle of NO_3^- concentrations in a headwater stream of the Orgeval watershed, available since 1976 from IRSTEA, however shows that NO_3^- concentration has only levelled off in the 1990s to $9.7 \text{ mgNO}_3\text{-N L}^{-1}$ on average, and reached $10.9 \text{ mgNO}_3\text{-N L}^{-1}$ in the 2000s (**Figure 5-7**). No trend toward a reduction is in fact observed for the Orgeval catchment. It appears that the current agricultural practices, although they involve careful calculation of the nitrogen fertilization with respect to the requirement of

crop growth during the vegetative period, are not able to further reduce the nitrogen surplus which is leached during the winter period. Alternative agricultural systems are therefore probably required for reducing NO_3^- leaching.

A few farms in the Orgeval watershed have been converted to organic farming practices. These farms use long crop rotations (8 yrs), established on small plots (<10 ha), starting with 2 or 3 years of alfalfa, then alternating cereals and legumes (peas or horse bean). External inputs of organic nitrogen, partly in the form of composted manure, are extremely limited. Although the cereal yield of these exploitations is about 15–20% lower than the conventional yield, their overall nitrogen surplus is much lower. Preliminary measurements (Benoit et al. unpublished) of sub-root NO_3^- concentrations measured with suction cups under the different plots of one such farm (site 2, Figure 5-8) shows values of about $13.4 \text{ mgNO}_3\text{-N L}^{-1}$ (SD =4.8), i.e. about half the value found for conventional farming. Note that the value found is higher than the range of the values reported by (Thieu et al. 2011) for organic farming based on literature data.

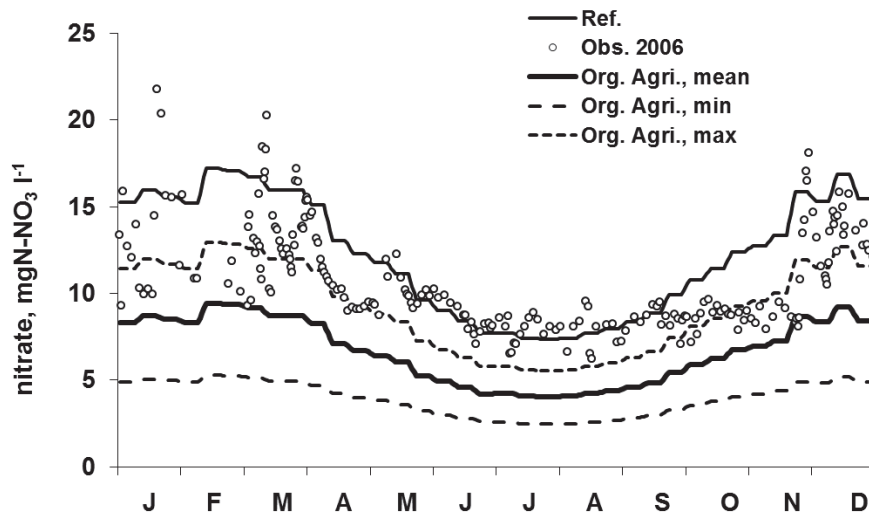


Figure 5-8. Seasonal variations of nitrate concentrations at the outlet of the Orgeval Basin, the year 2006 taken as example. A rather good agreement is obtained between the observations and the simulation for 2006. Compared to the reference simulation, the organic agricultural scenario shows a 45% decrease in annual mean nitrate concentrations (Org. Agri., mean). The amplitude of the response is shown with the exploration of the SD range (Org. Agri., min and max).

5.6.2. Modelling NO_3^- contamination resulting from GAP and generalized organic farming

The Seneque/RiverStrahler model has been run for exploring the effect of changes in agricultural practices at the scale of the Orgeval watershed. The current situation, modelled by considering a mean sub-root water concentration of $22 \text{ mgNO}_3\text{-N L}^{-1}$ under arable land, was compared with that corresponding to a concentration of $13.4 \text{ mgNO}_3\text{-N L}^{-1}$ (SD =4.8) (organic farm, see above). An average decrease of 45% (25–68%) of the annual nitrogen

concentrations at the outlet of the watershed is obtained (**Figure 5-8**). Such a preventive measure would not increase N_2O emissions, a result corroborated by our own experimental measurements in the Orgeval watershed (Benoit et al. unpublished) and could even reduce them (Aguilera et al. 2013). **Figure 5-5 c** compares the implication of this preventive scenario to the curative one (**Figure 5-5 b**) and the current situation (**Figure 5-5 a**).

5.7. Discussion and conclusions

The introduction of reactive nitrogen into the biosphere by modern agriculture has drastically increased, and the sequence of effects it causes in the atmosphere, in terrestrial ecosystems, in freshwater and marine systems, and on human health, is known as the nitrogen cascade (Galloway et al. 2003). In a river network with a continuous unidirectional transport of water and elements, the N cascade superimposed on the N spiraling, a concept defined as the travel distance of a water N atom before returning to the water downstream (Howard-Williams 1985).

A front-line question for the near future is: Can we change agricultural practices to re-equilibrate the nutrient stoichiometry of surface water, preventing eutrophication, and still satisfy the needs of the population (in food and drinking water) with sustainable agriculture? Considering that more than 50% of terrestrial reactive nitrogen is now from Haber-Bosch mineral nitrogen 'industrial production' (mostly in the food system or a consequence of it), to overcome environmental problems of N pollution in the next 50 years, suggestions for future research should focus on new approaches for analysing water-agro-food systems (Billen et al. 2013b), based on the concepts of socio-ecological trajectory (Fischer-Kowalski and Rotmans, 2009) and territorial ecology (Barles, 2013). The territorial watershed scale would be a suitable scale to initiate new directions in agricultural systems. Many discussions are converging to request a tightening of the feedback loop between production and consumption so as to achieve sustainability (Sundkvist et al. 2001; Davis et al. 2012). A political consensus on this matter is very difficult to achieve (Leridon and de Marsily, 2011; Swinnen and Squicciarini, 2012), but the regional scale allows a good level of coherence for decision and management, i.e. a level at which implementation of measures appears relatively possible.

The Orgeval watershed is nowadays one of the long-surveyed watershed case study areas that has been subjected to biogeochemical investigations in addition to the 50 years of study in hydrology. The facilities offered for monitoring have made it possible to determine a comprehensive budget of nitrogen transfer and transformations at the scale of this territory. Specific nitrogen fluxes delivered at the outlet of the Orgeval watershed has been estimated at $1130 \text{ kg N km}^{-2} \text{ yr}^{-1}$ and is on the order of that delivered at the outlet of the Seine Basin as a whole ($1600 \text{ kg N km}^{-2} \text{ yr}^{-1}$ for the 2002–2007 period; see Passy et al. 2013). A similar observation can be made for the N_2O emission, $\approx 140 \text{ kg N}_2\text{O-N km}^{-2} \text{ yr}^{-1}$ for the Orgeval watershed compared to the $180 \text{ kg N}_2\text{O-N km}^{-2} \text{ yr}^{-1}$ obtained at the scale of the Seine watershed (Garnier et al. 2009).

The studies conducted in the Orgeval watershed, reveal that denitrification, mostly in waterlogged soils in slope shoulders and riparian zones, is a major process for nitrogen elimination along its cascade from agricultural soil to the river outlet, already reducing the

fluxes of leached nitrogen between the base of the root zone and their discharge into the river system by 40–50% (see **Figure 5-3**). Globally, at least 10% of the total denitrification flux ends as greenhouse gas N₂O emissions.

Among the measures which can be envisaged to further reduce nitrogen contamination of surface water, the creation of shallow ponds can be valuable, especially in many traditional landscapes, which were once characterized by numerous ponds. Historical land use situations are indeed recognised useful for planning measures to achieve environmental targets (Glavan et al. 2013). Many authors have stressed the value of such landscape management, especially when other ecological functions can be associated, such as conservation of the biodiversity, connectivity in the landscape, etc. (Ruggerio et al. 2008; Le Viol et al. 2012; Armitage et al. 2012). However, ponds, often promoted as compensation measures or even for wastewater management (Howard-Williams, 1985), should not be implemented excessively or inconsistently: the connectivity of pond networks should be considered at the territorial landscape scale so that they remain favorable to biodiversity. Bronner et al. (2013), for instance, report that in the US, the policy of environmental compensation measures has led to a strong decrease of high-quality forested wetlands at the expense of low-quality wetland area, such as many isolated freshwater ponds. Using the Seneque/RiverStrahler model, we have shown that a 30–40% reduction of NO₃⁻ at the outlet of the watershed could be obtained by introducing drainage ponds, up to 5% of the total surface area of the watershed. However, this would increase N₂O emissions by about 50%.

A more effective, preventive reduction measure would be the conversion of agriculture to organic farming practices with low fertilization, which has been shown to allow significant reduction of NO₃⁻ concentration at the base of the root zone with respect to current conventional practices. This type of measure not only reduces nitrogen contamination at the source, thus also acting on groundwater contamination, but is the only one which allows reducing instead of increasing overall N₂O emissions by the watershed. The generalization of organic farming which requires local supply in organic manure as well as an outlet for its fodder production would be facilitated by the reintroduction of livestock farming in this specialized cereal cropping area. Clearly, meeting the objectives of the Water Framework Directive requires deep structural changes in the agriculture towards more sustainable and efficient systems (EU, 2013), rather than simple adjustments of farming practices (Volk et al. 2009; Glavan et al. 2012).

The combination of local studies together with an adapted modelling tool has proved here to be a relevant approach for quantifying nitrogen transformations and transfers at the watershed scale, even allowing the exploration of mitigation measures prior to field applications of ecological engineering investigations. Although several other process-based models might have been used (e.g. SWAT, Arnold et al. 1998, Neitsch, 2005; INCA, Whitehead et al. 1998; Wade et al. 2002), Seneque/RiverStrahler, was preferably used here, especially because it is currently used by the Seine Water Agency for WFD reporting. Other models based on regression approaches (e.g. GREEN, Grizzetti et al. 2005; MONERIS, Behrendt et al. 2002; NEWS-DIN, Dumont et al. 2005), would not have been able to explore scenarios like those tested here, because they would be too far from the calibrating data sets.

Acknowledgements

The FIRE-FR3020 research federation is greatly acknowledged for its interdisciplinary research framework and for funding the site's equipment. We extend our thanks to the PIREN-Seine program for providing funding for the analysis. François Gilloots and Eric Gobard are sincerely acknowledged for having allowed us to conduct this research in their fields. Thanks are due to the IRSTEA research institution for opening their experimental watershed (Orgeval watershed) to other scientific communities. This work was partly carried out in the scope of the ANR-ESCAPADE and ADEME-EFEMAIR projects.

Partie III.

La lixiviation de l'azote dans des exploitations du bassin de la Seine

Rapport-gratuit.com 
LE NUMERO 1 MONDIAL DU MÉMOIRES

Dans la précédente partie, nous avons pu appréhender les fuites d'azote à l'échelle d'une exploitation agricole représentative d'un petit bassin versant (l'Orgeval, 104 km²). Les concentrations d'azote dans les bougies poreuses ont aussi permis de modéliser les fuites d'azote à l'exutoire du bassin de l'Orgeval. Pour le bassin de la Seine, afin de réduire la contamination nitrique à la zone côtière, les mesures agro-environnementales ne montrant pas d'effet, une simulation de conversion à l'AB avait été explorée (Thieu et al. 2011). Une nette amélioration de la qualité de l'eau était observée, toutefois ces simulations ne s'appuyaient pas sur des résultats d'agrosystèmes biologiques spécifiques du bassin de la Seine, mais sur quelques données de la littérature en Europe du Nord. Afin de réaliser des scénarios réalistes, il fallait donc pallier à ce manque de données et c'est dans cet objectif que le réseau ABAC a été créé (AESN-DIM Astrea), pour acquérir les données de terrain nécessaires. La mise en place du réseau a débuté à l'automne 2012 avec huit systèmes de cultures dans trois pôles pédoclimatiques (cf. **Chapitre 6**). Du fait de l'intérêt général (scientifiques, gestionnaires et agriculteurs) pour ces résultats, le réseau s'est développé à l'automne 2013 dans dix-huit systèmes de cultures du bassin de la Seine, répartis en six pôles pédoclimatiques du bassin de la Seine (cf. **Chapitre 7**). Dans chaque pôle, au moins un système biologique et un système conventionnel ont été équipés pour s'affranchir des conditions pédoclimatiques, les comparaisons dans les six pôles permettant d'évaluer les performances agronomiques des systèmes et pratiques dans des conditions environnementales variées. Les sollicitations de la part des acteurs du territoire, nous ont conduits à équiper plusieurs systèmes au sein d'un même pôle, ce qui a permis d'enrichir nos résultats aussi bien en AB qu'en AC. Nous n'avons toutefois pas pu répondre à toutes les invitations, surtout celles en dehors du périmètre du bassin de la Seine, mais avons toujours cherché à rendre autonomes les demandeurs, afin que le jeu de données soit le plus complet possible.

6

Nitrate leaching from organic and conventional arable crop farms in the Seine basin (France)

Benoit M¹, Garnier J^{1,2}, Anglade J¹, Billen G^{1,2}

Ce chapitre constitue un article qui a été publié en Octobre 2014 dans la revue *Nutrient Cycling in Agrosystems* (DOI : 10.1007/s10705-014-9650-9).

¹UPMC, UMR 7619 Metis, Box 105, 4 place Jussieu, 75005 Paris, France

²CNRS, UMR 7619 Metis, Box 105, 4 place Jussieu, 75005 Paris, France

6.1. Abstract

In the Seine Basin, characterised by intensive arable crops, most of the surface and groundwater is contaminated by nitrate (NO_3^-). The goal of this study is to investigate nitrogen leaching on commercial arable crop farms in five organic and three conventional systems. In 2012–13, a total of 37 fields are studied on eight arable crop rotations, for three different soil and climate conditions. Our results show a gradient of soil solution concentrations in function of crops, lower for alfalfa (mean, $2.8 \text{ mg NO}_3\text{-N l}^{-1}$) and higher for crops fertilised after legumes ($15 \text{ mg NO}_3\text{-N l}^{-1}$). Catch crops decrease nitrate soil solution concentrations, below $10 \text{ mg NO}_3\text{-N l}^{-1}$. For a full rotation, the estimated mean concentrations is lower for organic farming, $12 \pm 5 \text{ mg NO}_3\text{-N l}^{-1}$ than for conventional farming $24 \pm 11 \text{ mg NO}_3\text{-N l}^{-1}$, with however a large range of variability. Overall, organic farming shows lower leaching rates ($14\text{--}50 \text{ kg NO}_3\text{-N ha}^{-1}$) than conventional farms ($32\text{--}77 \text{ kg NO}_3\text{-N ha}^{-1}$). Taking into account the slightly lower productivity of organic systems, we show that yield-scaled leaching values are also lower for organic ($0.2 \pm 0.1 \text{ kg N kg}^{-1} \text{ N yr}^{-1}$) than for conventional systems ($0.3 \pm 0.1 \text{ kg N kg}^{-1} \text{ N yr}^{-1}$). Overall, we show that organic farming systems have lower impact than conventional farming on N leaching, although there is still room for progress in both systems in commercial farms.

6.2. Introduction

Nitrogen (N) is an essential element for plant growth and its use in agriculture as a mineral fertiliser based on the Haber-Bosch process sharply increased after World War II. The current world fertiliser application mean is presently 133 kg N ha⁻¹ yr⁻¹, with strong heterogeneity in time and space. In France, the use of fertilisers reached a maximum of 200 kg N ha⁻¹ yr⁻¹ in the 2000s and has recently decreased to a mean of 150 kg N ha⁻¹ yr⁻¹ (www.faostat.fao.org), due to both price increase (+ 60% between 2000 and 2013; www.bdm.insee.fr) and to National and European regulations aiming at protecting water resources. Nitrate pollution of groundwater from agriculture is an issue of major concern for the European Union (EU) ([Addiscott et al. 1991](#); [Sutton et al. 2011](#)), which has identified vulnerable zones and promoted good agricultural practices since 1991, in the scope of the Nitrate Directive (no. 91/676/CEE). The whole Seine Basin, with intensive crop production, has been classified as a vulnerable zone, because 68% of its drinking water intakes are contaminated by pesticides and 30% by nitrate (NO₃⁻) ([AESN, 2013](#)).

Organic farming (OF) is already recognised as a good alternative to combat pesticide pollution and maintain biodiversity ([Pelosi et al. 2014](#); [Henneron et al. 2014](#)), but its impact on NO₃⁻ contamination is still controversial. Only a few studies (34 to our knowledge and none in France) have compared NO₃⁻ leaching in OF and conventional farming (CF) from arable crops in the EU. Comparisons in terms of area-scaled leaching values are often in favour of OF, with 30–40% lower values ([Berg et al. 1999](#); [Hansen et al. 2000](#); [Korsaeth and Eltun 2000](#); [Haas et al. 2002](#); [Stopes et al. 2002](#)), but in some studies around 20% more leaching in OF than in CF has been measured ([Kristensen et al. 1994](#); [Torstensson et al. 2006](#); [Sapkota et al. 2012](#)). On the other hand, when expressed in yield-scaled units, the differences in leaching rates either are not significant ([Kirchmann and Bergstrom 2001](#); [Mondelaers et al. 2009](#)) or disfavour OF ([Tuomisto and Helenius 2008](#); [Korsaeth 2008](#)).

The major question we address here is the capacity of OF systems in the Seine Basin to preserve surface and groundwater from NO₃⁻ contamination in the watershed, while producing arable crops. Launched to fill the gap of knowledge on NO₃⁻ leaching in the different cropping systems of the Seine Basin, the ABAC regional project (DIM-Astrea, Ile-de-France Region & AESN) is designed as a farmer-centred approach, referring to the “bottom-up” approach from the agroecology concept ([Altieri, 2002](#)), also named as “the next wave of innovation” according to [MacMillan and Benton \(2014\)](#). We choose to observe real practices of farmers, typical and representative of the cropping systems in their respective sub-region, since experimental plots and commercial farms can give different results, as shown for carbon sequestration by [Aguilera et al. \(2013\)](#).

Although our approach is based on data from a single year, instead of a number of continuous years, we compared systems of arable crop successions in organic and conventional commercial farms. Such data are essential for water managers seeking to protect groundwater quality, compatible with drinking water production, for the 16 million inhabitants of the Seine Basin. In this context, area-scaled leaching values are pertinent indicators for assessing the impact of agriculture on water quality, while yield-scaled leaching values (i.e. expressed per unit calorie or N harvested) reflect the trade-off of production vs. environmental contamination.

Therefore, soil solution concentrations are measured below the root zone, with vertical porous ceramic cups ([Stopes et al. 2002](#)). The vertical ceramic cups can be set up quickly

without destruction of the soil's horizons and sampled water in the soil solution zone can be directly analysed for NO_3^- concentrations.

Also, determining the diffuse sources associated with different agricultural practices, as an input to models of biogeochemical nutrient fluxes, will be an additional perspective of this study. As an example, the Seneque-Riverstrahler model, developed for calculating water quality of large river basins, such as the Seine watershed, is able to explore possible agricultural scenarios (Thieu et al. 2011) and their impact at the coastal zone in terms of eutrophication (Garnier et al. 2010; Passy et al. 2013).

6.3. Materials and methods

6.3.1. Localisation and characteristics of the main areas

The Seine Basin, with a surface of 78,650 km², has substantial agricultural activity. The climate is humid and temperate, with a large gradient of temperature and rainfall. Arable crops cover around 60% of the utilised agricultural land (UAL) with wheat as a dominant crop (29% UAL). In this area, OF accounts for only 1.5% UAL, compared with 3.8% UAL in France in 2012.

A total of eight agricultural systems was studied in three sectors of the Seine Basin, with different rainfall patterns, pedology and agriculture practices. The first group is located in the East of the Seine Basin, in Seine & Marne (S&M), with deep loamy soil (luvisol) generally drained due to hydromorphic conditions, over a calcareous substratum. The mean annual rainfall is around 700 mm and the mean annual temperature 9.7°C (40 years of measurements at the Boissy-le-Châtel weather station). Most of agriculture in S&M is characterised by winter wheat, alternating with maize or faba beans, with only 1.1% UAL in OF. The second group, located in the North of the Seine Basin, in Oise, is characterised by a chalky substratum, present in nearly all the periphery of the Seine Basin. The annual averages in rainfall and temperature are similar to those of the S&M, 697 mm and 9.7°C, respectively (30 years of data measured in the Saint Quentin, Météo France weather station). Oise agriculture is dominated by arable crops like wheat (> 50% UAL), rape seed and sugar root production, with only 1.3% UAL in OF. The third group is located in the South of the Seine Basin, in Yonne, with average annual rainfall and temperature of 880 mm and 10.7°C, respectively (30 years of data measured in Cruzy, Météo France weather station) and a chalky substratum. Percentages of oat and oilseed rape are higher in Yonne than in the other sectors, as OF which accounts for 4.1% UAL.

6.3.2. Agricultural practices studied

Five OF and three CF cropping systems were studied, characterised by different practices and regions. In S&M, four cropping systems were studied, including two CFs and two OFs. The CF rotations last two years with tillage (CF1) or without tillage (CF2), whereas one OF rotation lasts five years with low exogenous fertilisation (OF1) and the other lasts nine years with exogenous fertilisation (OF2). In Oise, the three systems studied count two OFs, one with a three crops rotation (OF3) and the second with a six year rotation (OF4), both using vinasse on cereals; the CFs, is characterised by a three years rotation (CF3). The last OF located in Yonne is autonomous, a six years rotation with no exogenous fertilisation (OF5). Except CF2, all the systems use standard tillage (Table 6-1). The conversion time to organic

systems ranges from 3 (OF1, OF3, OF5) to 10 years (OF2, OF4). In this region, because of the very low livestock density, arable crop farms in OF replace mineral fertilisers by the introduction of legume crops and a low proportion of exogenous organic fertiliser application (manure, vinasse, poultry droppings, etc.). In order to compensate herbicide applications, OF requires tillage, harrow, hoe, crops diversification and introduction of forage crops such as alfalfa, which, besides nitrogen symbiotic fixing, considerably reduces self-propagating weeds.

Table 6-1. Main characteristics of the cropping systems studied in the three regions (S&M, Oise, Yonne) in OF and CF with the number of the fields studied, the exogenous fertiliser types used and the tillage presence (X) or absence (No).

Region	Systems	Number of fields studied	Fertiliser types	Tillage
S&M	CF1	2	Mineral or organic	x
S&M	CF2	2	Mineral	No
Oise	CF3	3	Mineral	X
S&M	OF1	5	Vinasse and horse manure	X
S&M	OF2	9	Poultry, horse manure and vinasse	X
Oise	OF3	3	vinasse	X
Oise	OF4	6	vinasse	X
Yonne	OF5	6	-	X

All OF rotations were rather long (mean, 7 years) including alfalfa in the beginning. For OF systems, 30 fields were equipped with ceramic cups including nine different crops: alfalfa (eight fields), wheat (nine fields), flax (two fields), faba beans (three fields) and lentils (one field), and rye, oat or triticale and maize (one field each) (**Table 6-2**). In CF, with shorter rotation (2-3 years), seven fields were instrumented for five different crops: wheat (four fields), rapeseed, faba beans and maize (one field each). Exogenous fertilisation in OF is between 8-200 kg N ha⁻¹ and 74-238 kg N ha⁻¹ in CF. Organic fertilisers, applied during the study, were sugar beet vinasse (17 kg N t⁻¹), horse manure (7 kg N t⁻¹) or poultry manure (43 kg N t⁻¹). Mineral fertilisation was generally a combination of calcium ammonium nitrate (CAN, 27% N) and urea (35% N). During the study, catch crops (CC) as mixed seeds (60% vetch, 15% clover, 15% lacy phacelia, 10% mustard) were grown from September to December on CF1 and OF1.

Table 6-2. Data of the fields instrumented in the eight systems in OF and CF including crops, previous crops with straw buried (b.) or exported (e.) and its net BNF inputs, N exogenous inputs (organic or mineral fertilisers) in 2012-13, SMN before fall tillage (Oct; Nov; Dec), soil solution concentrations (soil solution conc.) and N harvested.

Farm system	Crops (2012-13)	Previous crops (2011-12)	Net BNF previous crops	N exogenous inputs	Crop Harvest (2012-13)	SMN after fall tillage (2012)	Soil solution conc.	Fertiliser types
			kg N ha ⁻¹	kg N ha ⁻¹	kg N ha ⁻¹	kg N ha ⁻¹	mg N l ⁻¹	
CF1	CC* - faba	wheat (b)	0	74	221	44.5	2.4	vinasse
CF1	wheat	pea	19	165	175	52.1	23.2	ammonitrate
CF2	wheat	faba- maize (b.)	154	212	162	25.4	28.7	N, P, K (2 -2 - 1)
CF2	faba - maize	wheat (b.)	0	175	125	10.4	6.5	N, P, K (2 -2 - 1)
CF3	wheat	wheat (e.)	0	238	168	24.7	24.5	N, P, K (2 -2 - 1.8)
CF3	rapeseed	wheat (e.)	0	130	116	23.5	30.9	N, P, K (2 -2 - 1.8)
CF3	wheat	peas	39	136	215	41.9	61.1	N, P, K (2 -2 - 1.8)
OF1	alfalfa 2	alfafa 1 (e.)	137	0	304	14.9	1	
OF1	alfalfa 2	alfafa 1 (b.)	173	0	304	16.1	4	
OF1	wheat	faba beans	28	80	72	n.d	10.4	vinasse, horse manure
OF1	CC* - flax	wheat (b.)	0	50	31	n.d	9.2	vinasse
OF1	CC* - beans	wheat (b.)	0	0	52	n.d	6.2	
OF2	alfalfa 1	triticale (e.)	0	0	75	5.4	2.4	
OF2	alfalfa 2	alfalfa 1 (e.)	157	0	403	12.8	3	
OF2	wheat	alfalfa 2 (e.)	190	0	125	37.5	17	
OF2	flax	wheat (e.)	0	0	39	24.1	23	
OF2	wheat	flax (b.)	0	200	86	43.1	42	poultry & horse manure
OF2	baresoil - lentils	wheat (e.)	0	0	59	24.1	17	
OF2	triticale	lentils	8	50	96	23.5	23	vinasse
OF2	wheat	faba beans	55	200	95	19.5	26	horse & poultry manure
OF2	oat	wheat (e.)	0	140	120	16.8	8.7	horse manure
OF3	alfafa 2	alfalfa 1 (e.)	146	0	310	8.4	18	
OF3	baresoil - wheat	triticale (b.)	0	0	98	n.d	31	
OF3	wheat	maize (b.)	0	48	80	25.6	25	vinasse
OF4	alfafa1	wheat (b.)	0	0	81	8.4	7	
OF4	wheat	alfafa 2 (e.)	102	0	70	6.2	13	
OF4	wheat	maize (b.)	0	0	95	12.8	5	
OF4	maize	wheat (b.)	0	0	78	11.3	16	
OF4	baresoil - faba	spelt (b.)	0	0	66	4.8	2	
OF4	wheat	vegetables	0	32	70	4.8	1	vinasse
OF5	alfalfa 2	alfalfa 1 (e.)	170	0	397	0.9	2	
OF5	alfalfa 3	alfalfa 2 (e.)	187	0	217	13.1	7	
OF5	wheat	wheat (e.)	0	0	30	10.2	9	
OF5	baresoil - faba	sunflower	0	0	74	8.3	9	
OF5	wheat	faba beans	31	0	38	13.3	18	
OF5	rye	wheat spring (b.)	0	0	24	11.9	5	

* CC ploughed in December 2012

6.3.3. Field measurements

6.3.3.1. Ceramic cups

Ceramic cups have been used, since 1904, to measure NO_3^- in groundwater (Briggs and McCall 1904). Their cost, effectiveness and ease of installation have made them the most commonly used devices for collecting soil solution water. Because our experimental study is farmer-centred, all the ceramic cups were installed vertically to avoid soil damage in the fields. Furthermore, this approach is supported by the comparison between results from both horizontal and vertical installations, which did not show significant differences ($P\text{-value} \geq 0.05$) (Bowman et al. 2002). Although the vertical installation is flexible, the porous ceramic cups must be removed before plowing, which can make a long-term monitoring campaign difficult. All the soils of the ABAC farm network were at least 90 cm deep, favourable for installing vertical ceramic cups taking into account 80–90% of the root density. A total of 37 fields were equipped with ceramic cups measuring 85 cm in length (SDEC, France, SPS Ø 31mm), implemented with a manual auger of the same diameter, with the head placed 5 cm below the ground surface, allowing shallow tillage. We assumed that six ceramic cups per field (i.e. 37×6 cups) would make it possible to determine the local variability; other studies generally using at least three (Eriksen et al. 1999) and up to ten ceramic cups (Stopes et al. 2002). The ceramic cups were arranged on a line parallel to the soil tillage, a minimum of 14 m from the edge of the field, in order to avoid any side effect. After 48 h of vacuum setting, samples were taken weekly at the beginning of the rainy hydrological season, when the soil is water saturated (first month) and fortnightly for the rest of the drainage period. Many of the farmers were involved in the sampling process. The period of sample collection lasted 6 months, from December 12th, 2012 to May 22nd, 2013, with an average of 10 sampling dates (≈ 2500 samples for analysis).

6.3.3.2. Soil samples

For each field, right after tillage (from October to December), the six samples of soils are extracted over 90 cm depth. Soil samples were collected with the auger and pooled for the three layers [0–30], [30–60] and [60–90] cm. The fresh soil samples were stored for a few days at 4°C, until analysis of humidity and soil mineral nitrogen (SMN) and then frozen at –18°C for further analysis (texture and C, N content).

6.3.4. Analyses

6.3.4.1. Soil analysis

Triplicates of soil were weighed (30 g) to determine moisture and soil organic matter (SOM) using the loss-on-ignition method. Soil samples were heated at 105 °C (48 h) for the former, and calcined at 450 °C (4 h) for the latter, and re-weighed after each step. Soil mineral nitrogen concentrations were determined after KCl extraction, with 5 g of soil in 20 ml of KCl (2 M) for 2 h on a shaking table. The suspensions were centrifuged for 10 min at 3000 rpm and the supernatant is frozen at –18 °C until analysis in the autoanalyzer (Quattro, Bran & Luebbe). The rest of the sample is freeze-dried to determine the particle size distribution (without decarbonation), soil organic carbon (SOC) (after decarbonation), and loss of ignition at 1000°C, and total N (LAS, INRA Arras).

6.3.4.2. Soil solution analysis

The soil solution taken from each ceramic cup is frozen until analysis of the N concentrations using an autoanalyzer Quattro (Bran & Luebbe). The method used to measure ammonium (NH_4^+) is based on the reaction of the blue indophenol (Slawyk and MacIsaac 1972). Nitrite (NO_2^-) and NO_3^- were measured using the sulphanilamide method (Jones 1984) and NO_3^- concentrations were determined after reduction to nitrite.

6.3.5. Calculations

6.3.5.1. Percolating water flow

The daily percolating water flow (W_i , mm d^{-1}) was calculated using climate data from the nearest weather station (Irstea or Météo France) in each sector (S&M, Oise, Yonne) located in Boissy-le-Châtel (48°49'15"N 3°08'19"E), Mesnil-sur-Bul (48°04'42"N 3°35'00"E) and Arces (48°04'42"N 3°35'00"E), respectively. We used daily rainfall (R_i , mm) and daily potential evapotranspiration (ETP_i , mm) to calculate W_i during the sampling period.

The daily water storage (WS_i , mm d^{-1}) was incremented by the daily previous water storage (WS_{i-1} , mm d^{-1}), R_i , ETP_i multiplied by a crop coefficient (k) fixed at 0.5 during the winter period (Perrier et al. 1980; Katerji and Perrier 1985; Allen, 2000) and the previous water inflow (W_{i-1} , mm d^{-1}) (Equation 6-1). Then W_i was determined by the difference between the WS_i and the water holding capacity up to its field capacity (WHC_{FC} , mm) (Equation 6-2), which was determined from soil characteristics (depth, texture and structure) (Bruand et al. 2004).

Equation 6-1. $\text{WS}_i = \max (\text{WS}_{i-1} + R_i - k \cdot \text{ETP}_i - W_{i-1}; 0)$

Equation 6-2. $W_i = \max (\text{WS}_i - \text{WHC}_{\text{FC}}; 0)$

The N leached flow was calculated between each collecting date, by multiplying the average concentration by the amount of infiltrated water. Leaching was measured for a six months sampling period, from December to May and is assumed to represent the total leaching of the year, as vegetation uptake and evaporation prevent leaching during the rest of the year.

6.3.5.2. N inputs: fertilisers and biological nitrogen fixation

The calculation of total N inputs takes into account exogenous inputs, via the application of organic and mineral fertilisers and biological N fixation (BNF) by previous legume crops.

The amount and N content of fertilisers (organic and mineral) and the straw management (buried or exported) were documented by farmers. The BNF is estimated from yields, using the relations established in Anglade et al. (submitted) for six legume species commonly grown in Northern Europe namely alfalfa, faba bean, field pea, lentil and white/red clover. Highly significant linear relationships were found between total N accumulation in shoot (N_y) and the amount of fixed N_2 derived from atmosphere (BNF), with different regression coefficients depending on species (α_{cult} ; β_{cult}). In order to take into account below-ground contributions (BGN), comprising N associated with roots, nodules and rhizodeposition via exudates and decaying root cells and hyphae, multiplicative factors (BG) derived from a literature review were attributed, amounting 1.3 and 1.7 for grain and forage legumes, respectively (Equation 6-3).

Equation 6-3. $\text{BNF (kg N ha}^{-1}\text{)} = \text{BG} * [\alpha_{\text{cult}} * \text{Ny}_{\text{cult}} + \beta_{\text{cult}}]$

Then, net input by BNF (net BNF) from the preceding crop is obtained by subtracting N harvested, in grain or in herbage, from the estimated total N input (including BGN) derived from N_2 fixation. We assumed an above-ground N harvest index (NHI) value of 0.75 for grain legumes. For alfalfa, different values were used depending on cutting regimes, e.g., 3 cuts at a height of 10 cm (3-inches), with one left in the field as green manure, is common.

6.3.5.3. Statistical analysis

All the statistical analyses were performed using R software. Differences within data sets were analysed using the Kolmogorov-Smirnov test to verify normality ($P > 0.05$). Analysis of variance tests (ANOVA) were determined for the data sets from the different areas, cropping systems and depths. Significance is accepted at $P < 0.05$.

6.4. Results

6.4.1. Soil properties in OF and CF

6.4.1.1. Textures

Composition in silt, sand (coarse and fine) and clay were determined for each field for the three layers. In all the fields, the clay percentages increased significantly with depth (P -value = $5.5\text{e-}07$ ***), whereas the silt percentages significantly decreased (P -value = 0.0002 ***). Average clay was around 25% and maximum in CF3 (31%) and OF4 (29%), and minimum in OF5 (21%). The mean percentage of sand was around 10%, maximum in OF5 (24%) and minimum in OF2 (4.5%). The mean WHC_{FC} was 200 mm in Oise, 180 mm in S&M and 160 mm in Yonne.

6.4.1.2. Nutrients

Nutrients as SOM, SOC and total N decreased with depth (P -value = $6.5\text{e-}14$ ***), with no significant differences between the OF and CF systems in this network. The SOM values were significantly different within depths and between systems (P -value = 0.003 **), but not between systems (OF, CF) (P -value = 0.2). Soil mineral N before the period of drainage was measured in the eight systems. Ammonium concentrations were stable in relation to depth, with a mean of 10 ± 0.6 kg $\text{NH}_4\text{-N ha}^{-1}$ for all systems. Nitrate concentrations were significantly higher in the first layer (0–30 cm) (P -value = $6.5\text{e-}14$ ***), with high variations between fields, e.g., within a range from 1 to 17 kg $\text{NO}_3\text{-N ha}^{-1}$. Soil mineral N were not significantly different between the OF and CF systems, due to the high variability between fields in terms of crops, and the preceding crop, and hence agricultural practices (Table 6-2).

6.4.2. Soil solution concentrations for the typical OF and CF crop successions

Soil solution concentrations showed variations along the drainage period in all fields. The percentage variation from the six ceramic cups at one date averaged 47%, mainly because of

the high micro-heterogeneity of the soils in terms of their composition, texture and therefore physical and biogeochemical processes. In the following, for each field, we will use the mean soil solution concentration, measured during the drainage period (~ 6 cups $\times$ 10 sampling dates). In all the equipped fields, soil solution concentrations increased as a function of SMN integrated over the soil profile (**Figure 6-1 a**). Soil mineral N can therefore be used in a first approach, as an indicator of soil solution concentrations. No relation was found between soil solution concentrations and SOM (**Figure 6-1 b**).

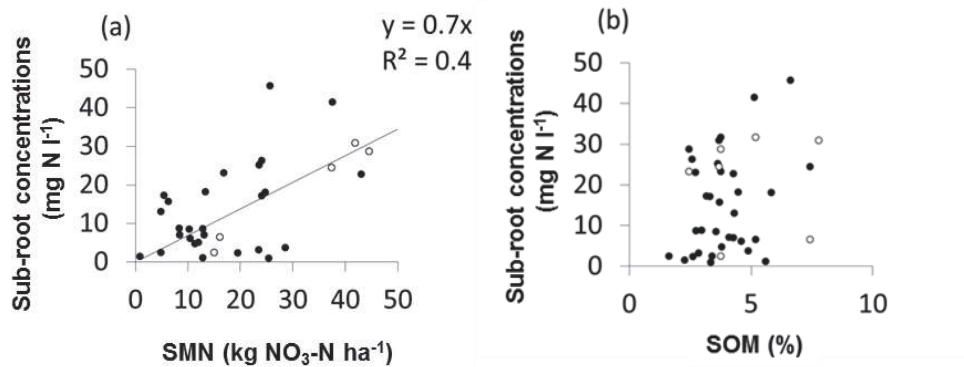


Figure 6-1. Relations between (a) sub-root concentrations and SMN before the drainage period and (b) sub-root concentrations and SOM, over the profile in all the fields instrumented in ceramic cups: black circles (organic) and open circles (conventional) fields.

6.4.2.1. Variations in organic rotations

Regarding OF, the lowest soil solution concentration was found for fields cultivated with legumes (5 ± 4 mg $\text{NO}_3\text{-N l}^{-1}$). The first year after alfalfa was ploughed, the mean soil solution concentration was 15 ± 2 mg $\text{NO}_3\text{-N l}^{-1}$, due to mineralisation and 12 ± 9 mg $\text{NO}_3\text{-N l}^{-1}$ for the second year (**Figure 6-2 a**). Grain legumes without fertilisation (faba beans, lentils in the fourth position in the rotation) had a mean soil solution concentration of 9 ± 6 mg $\text{NO}_3\text{-N l}^{-1}$. Crops after the legumes, with a mean 68 kg $\text{NO}_3\text{-N ha}^{-1}$ net BNF, had a mean soil solution concentration of 21 ± 6 mg $\text{NO}_3\text{-N l}^{-1}$. At the end of the rotation, cereals with low N input showed a mean soil solution concentration of 12 ± 8 mg $\text{NO}_3\text{-N l}^{-1}$. The use of mixed-seeds as CCs led to low concentrations, 5 mg $\text{NO}_3\text{-N l}^{-1}$ in OF1. Overall, the mean N soil solution concentration for the 7-years theoretical OF rotation was 12 ± 5 mg $\text{NO}_3\text{-N l}^{-1}$.

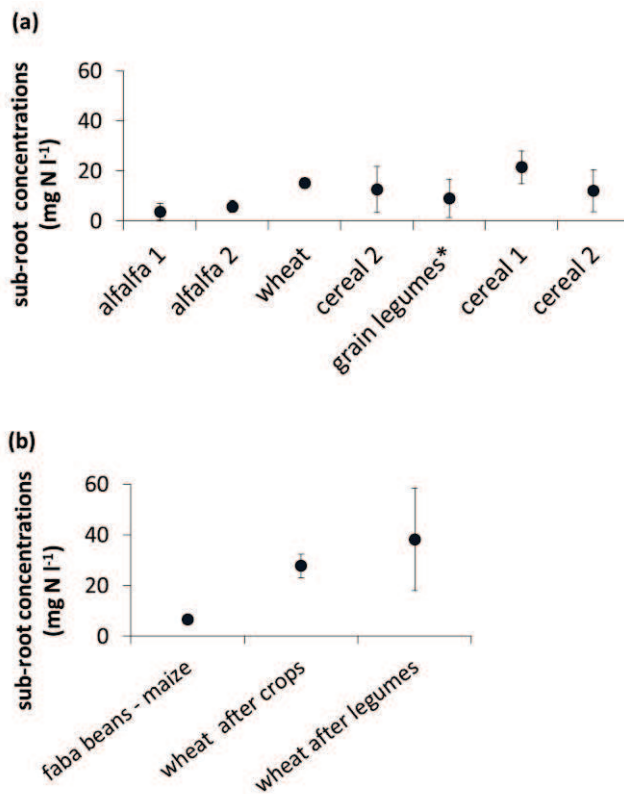


Figure 6-2. Means ($\pm$ SD) sub-root concentrations (a) from organic rotation with succeeding crops : alfalfa 1 (n=3), alfalfa 2 (n=5), wheat post-alfalfa 2 (n=3), cereals 2 (n=3), grain legumes (n=3), cereal 1 (n=5), cereal 2 (n=5); (b) from conventional rotation with maize (n=1), wheat after crops (n=2) and wheat after legume (n=3). (*) sampling following CC ploughing in December 2012 or bare soil.

6.4.2.2. Variations in conventional rotations

For conventional rotations, N soil solution concentrations were measured in S&M and Oise (Figure 6-2 b). Maize succeeding faba-beans led to the lowest concentrations ($6 \text{ mg NO}_3\text{-N l}^{-1}$); fertilised crops resulted in a mean concentration of $28 \pm 4 \text{ mg NO}_3\text{-N l}^{-1}$, whereas the concentration for wheat after legumes reached $38 \pm 20 \text{ mg NO}_3\text{-N l}^{-1}$. Green manure used as a biannual CC resulted in considerable soil solution concentration, as for wheat on CF2 ($30 \text{ mg NO}_3\text{-N l}^{-1}$), higher than crops after legumes when exported, and as for wheat post-peas on CF1 ($23 \text{ mg NO}_3\text{-N l}^{-1}$). Mean soil solution concentration, for the 3-years theoretical CF rotation (wheat, legumes with CC, wheat) was $24 \pm 11 \text{ mg NO}_3\text{-N l}^{-1}$.

At total, considering the typical crop successions for OF (7 crops) and CF (3 crops) in the studied region, soil solution concentrations were on average lower for OF than for CF ($12 \text{ mg NO}_3\text{-N l}^{-1}$ against $24 \text{ mg NO}_3\text{-N l}^{-1}$), although this difference is not statistically significant given the high variability within each system.

6.4.2.3. Impact of N inputs on organic crops

Crops on the fourth or fifth position in the OF rotation were generally fertilised with vinasse, poultry droppings or compost, leading to various N soil solution concentrations. For organic fertilisers as poultry manure or vinasse, the soil solution concentrations were directly impacted by the total N amount applied (**Figure 6-3**). For example, the addition of poultry manure to horse manure or crops after legume (200 kg N ha^{-1}) led to soil solution concentrations over $26 \text{ mg NO}_3\text{-N l}^{-1}$. However in presence of CC or horse manure alone, soil solution concentrations were lower than $11 \text{ mg NO}_3\text{-N l}^{-1}$ despite high amounts of organic N applied. For a vinasse application (50 kg N ha^{-1}) without CC, the soil solution concentration was $25 \text{ mg NO}_3\text{-N l}^{-1}$, whereas in presence of CC, the soil solution concentrations decreased to $6 \pm 5 \text{ mg NO}_3\text{-N l}^{-1}$. Also, for crops after grain legumes, the application of compost before vinasse decreased the soil solution concentrations from $22.8 \text{ mg NO}_3\text{-N l}^{-1}$ to $10.4 \text{ mg NO}_3\text{-N l}^{-1}$. Concerning legumes, although net BNF was five times higher for crops after two years of alfalfa (150 kg N ha^{-1}) than for crops after grain legumes (30 kg N ha^{-1}), their corresponding soil solution concentrations means were $15 \pm 3 \text{ mg NO}_3\text{-N l}^{-1}$ and $20 \pm 3 \text{ mg NO}_3\text{-N l}^{-1}$ respectively (**Table 6-2**).

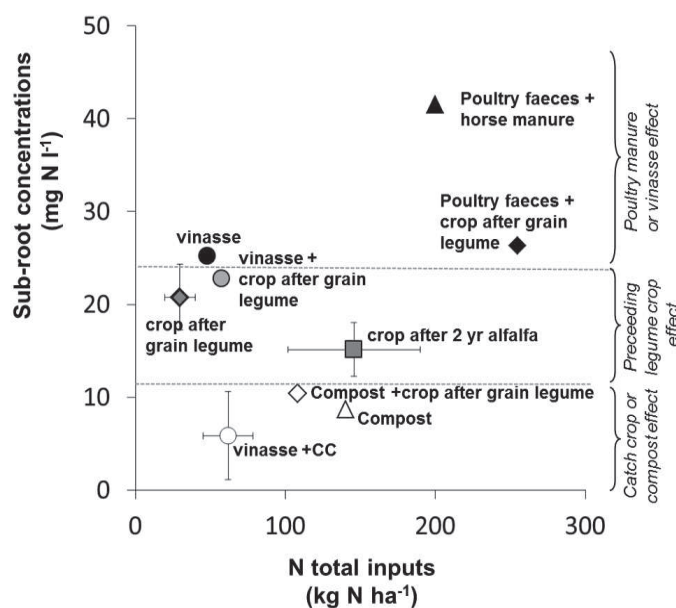


Figure 6-3. Effect of N inputs (amounts and types : organic fertilisers, legumes) and CC on sub-root concentrations in OF fields.

6.4.3. Influence of crop management, soil and climate on nitrate losses

Differently from the previous section where we have analysed soil solution concentrations for typical crop successions for OF and CF, here we analysed soil solution concentrations and N leaching by cropping systems.

6.4.3.1. Relation between N inputs and soil solution concentrations

Mean soil solution concentrations and N inputs have been calculated for each systems (Figure 6-4 a). In S&M, OF1 had the lowest mean soil solution concentration with 6 ± 5 mg $\text{NO}_3\text{-N l}^{-1}$, covering three legumes on a five crops rotation which account two thirds of the N inputs. The lowest mean soil solution concentration observed in conventional was 13 ± 7 mg $\text{NO}_3\text{-N l}^{-1}$ in CF1, in relation to low N inputs ($129 \text{ kg N ha}^{-1} \text{ yr}^{-1}$). For the full OF2 rotation, soil solution concentrations mean was 18 ± 9 mg $\text{NO}_3\text{-N l}^{-1}$, using both exogenous inputs ($66 \text{ kg N ha}^{-1} \text{ yr}^{-1}$) and net BNF ($45 \text{ kg N ha}^{-1} \text{ yr}^{-1}$), a figure close to that found on the no-till system, CF2, amounting 19 ± 10 mg $\text{NO}_3\text{-N l}^{-1}$, also coming from fertilisers and net BNF ($348 \text{ kg N ha}^{-1} \text{ yr}^{-1}$). In Oise, the soil solution concentrations showed considerable variations between cropping systems. The highest soil solution concentrations means have been measured in CF3 (39 ± 15 mg $\text{NO}_3\text{-N l}^{-1}$) for $181 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ total inputs, coming at 87% N inputs from synthetic fertilisers, which is typical of conventional practices in this area and in OF3 (25 ± 8 mg $\text{NO}_3\text{-N l}^{-1}$) for $89 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ N total inputs (83% N inputs from net BNF). In contrary, in OF4 the mean soil solution concentration on the rotation was 7 ± 6 mg N l^{-1} , in relation to low total inputs ($22 \text{ kg N ha}^{-1} \text{ yr}^{-1}$) with nearly no exogenous inputs ($5 \text{ kg N ha}^{-1} \text{ yr}^{-1}$). In Yonne, the mean soil solution concentration was 8 ± 3 mg N l^{-1} for OF5 (no CF investigation in this area), in reference to no exogenous input and three legumes on six year rotation providing $65 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ net BNF (Table 6-2).

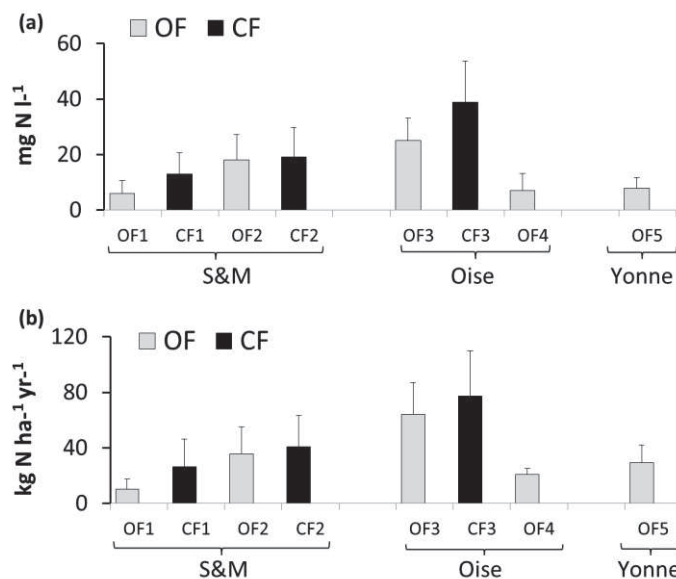


Figure 6-4. Sub-root concentrations (a) and leaching (b) from the different cropping systems in each area (S&M, Oise, Yonne) with OF in grey and CF systems in black. Standard deviations represent the range of variations within each rotation.

6.4.3.2. Hydrological conditions and leaching

Water holding capacity at field capacity equalled 180 mm in S&M, 190 mm in Oise and 160 mm in Yonne. Finally, the Wi cumulated during the drainage period was 235 mm in S&M, 209 mm in Oise and 239 mm in Yonne.

In most cases, the conversions from concentration to leaching (concentration x infiltrated water) did not change the final ranking of the cropping systems in terms of NO_3^- leaching, except in Yonne (**Figure 6-4 b**). Indeed, OF5 contributed to a higher leaching ($37 \text{ kg NO}_3\text{-N ha}^{-1} \text{ yr}^{-1}$) than OF4 in Oise ($13 \text{ kg NO}_3\text{-N ha}^{-1} \text{ yr}^{-1}$), despite their similar soil solution concentrations, due to its higher Wi in 2012-13, e.g. in Yonne (239 m) than in Oise (209 mm). In S&M or Oise, leaching for full rotations, were lower, for OF (13 to $37 \text{ kg NO}_3\text{-N ha}^{-1} \text{ yr}^{-1}$) than for CF (32 to $77 \text{ kg NO}_3\text{-N ha}^{-1} \text{ yr}^{-1}$) but the difference is not significant.

6.4.3.3. Relation between N total inputs, harvest and N leaching

Considering the entire span of the rotations studied, the yearly average total N input, including total BNF over the rotation, was 20% higher in CF ($211 \text{ kg N ha}^{-1} \text{ yr}^{-1}$) than in OF ($167 \text{ kg N ha}^{-1} \text{ yr}^{-1}$). Moreover, the total input was distributed differently between total BNF and exogenous fertilisation for OF and CF. Mean total BNF integrated over the whole rotations in OF and CF systems was 132 and $38 \text{ kg N ha}^{-1} \text{ yr}^{-1}$, respectively, whereas the exogenous fertilisation in OF and CF was conversely $23 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ and $160 \text{ kg N ha}^{-1} \text{ yr}^{-1}$, respectively. As a result, the soil solution concentrations tended to be lower for the OFs ($13 \pm 6 \text{ mg N l}^{-1}$) than for the CFs ($24 \pm 10 \text{ mg N l}^{-1}$). However, a gradient of soil solution concentrations exists in both systems.

The mean harvest was 24% higher in CF (169 kg N ha^{-1}) than in OF (129 kg N ha^{-1}), with 20% more total inputs. Thus the means N use efficiency (i.e. the ratio of harvested N on total N inputs) is similar in OF and CF, 78% and 81% respectively. Looking at leaching per unit kg N harvested (yield-scaled leaching), OF systems still show slightly lower values than CF, ranging from 0.1 to $0.3 \text{ kg N kg}^{-1} \text{ N}$ and 0.2 to $0.5 \text{ kg N kg}^{-1} \text{ N}$ respectively (**Table 6-3**).

Table 6-3. Yearly means of N inputs: N exogenous (fertilisers), total BNF and atmospheric deposition, harvest and N losses: sub-root concentrations, leaching per ha or per protein for OF and CF rotations studied in S&M, Oise and Yonne.

Farms	N exogenous input	BNF input previous crop	Atmospheric deposition	Mean N harvest (% N from legumes)	Soil solution concentration	Leaching per ha	Leaching per N harvested
	$\text{kg N ha}^{-1} \text{ yr}^{-1}$	$\text{kg N ha}^{-1} \text{ yr}^{-1}$	$\text{kg N ha}^{-1} \text{ yr}^{-1}$	$\text{kg N ha}^{-1} \text{ yr}^{-1}$	mg N l^{-1}	$\text{kg N ha}^{-1} \text{ yr}^{-1}$	$\text{kg N kg}^{-1} \text{ N yr}^{-1}$
CF1	119	10	12	198 (56%)	13	32	0.2
CF2	194	154	12	143 (0%)	19	51	0.3
CF3	168	13	12	166 (0%)	39	77	0.5
OF1	26	293	12	153 (77%)	6	14	0.1
OF2	66	152	12	122 (42%)	18	37	0.3
OF3	16	127	12	162 (64%)	25	50	0.3
OF4	5	63	12	77 (32%)	7	13	0.2
OF5	0	207	12	130 (79%)	8	37	0.3

6.5. Discussion

6.5.1. Experimental design and advantages of studying commercial farms

Herein we compared alternative systems, each with its own logic but having enough features in common for a useful comparison. Our sample of cropping systems is predominated by organic systems because of the lack of available references for OF, comparing with CF. We found a broad diversity of practices existing in both OF and CF, with a wide range of leaching values.

Classical agronomical research is often conducted in factorial experiments deconstructing a complex system in order to isolate specific components and identify cause-and-effect relationships. Factorial experiments are particularly relevant in agronomical studies when they are conducted in the absence of an ecosystem context. However, studies of intact agroecosystems have established the importance of both long-term and landscape-scale effects, especially in assessing alternative and innovative practices used by farmers on watersheds (Sharpley et al. 1994; Nguyen et al. 1995).

6.5.2. Effect of climate and soil on leaching variability

The three pedo-climatic regions covered in our study (Yonne, S&M and Oise) are representative of the substantial variability in water infiltration in the Seine Basin (Ledoux et al. 2007). It is well known that the amount of infiltrated water is a major determining factor of leaching, and depends on rainfall and soil texture. In this study, Yonne is the most affected by leaching, due to high Wi and the highest sand percentage. In our study, as in many others, the highest leaching is related to the highest sand percentage (Nieder et al. 1995; Beaudoin et al. 2005), for both conventional and organic systems (Hansen et al. 2000).

6.5.3. Key management practices controlling N leaching

Variations in soil solution N concentrations over the rotations are associated with crop type. We conclude here that leaching values have the same ranking in relation to crops (legumes < crops with CC < winter crops < crops after legumes) as those measured in organic systems in Norway, with legumes (6 kg N ha⁻¹), undersown grain (13 kg N ha⁻¹), vegetables (17 kg N ha⁻¹), grain without undersown legumes (30 kg N ha⁻¹) and potatoes (33 kg N ha⁻¹) (Solberg 1995). The soil cover during winter, which ensures incorporation of SMN, as well as the period of implantation and the root depth of the crop explain this ranking.

In this study, we observe similar and very low soil solution concentrations (5.5 mg NO₃-N l⁻¹) during the second year of alfalfa although net BNF from the previous year of alfalfa is high (between 137 kg N ha⁻¹ and 170 kg N ha⁻¹). This can be explained by the fact that alfalfa is very effective at accessing deep-leached NO₃⁻ (1 m below the soil surface) and rhizodeposition very low during the crop development in winter. After alfalfa ploughing, the amount of net BNF has an effect on soil solution concentrations, with mean concentrations 15 ± 2 mg N l⁻¹ for N inputs 102-190 kg N ha⁻¹. Differences of around 30 % of the soil solution concentrations (from 13 to 17 mg N l⁻¹) can be explained by differences in net BNF, depending on biomass yields (9 t ha⁻¹ with 2 cuts vs. 13 t ha⁻¹ with 3 cuts). Such an increased input leads to a 66 % increase in wheat yields (from 3 t ha⁻¹ to 5 t ha⁻¹). As a whole, the

incorporation of alfalfa into the soil did not lead to a massive loss of NO_3^- , probably due to its low rate of mineralisation. However we would recommend alfalfa exportation and its date of destruction should be as late as possible (e.g., spring or late winter), in order to reduce NO_3^- leaching (Francis et al. 1992). Moreover, for the two following years after alfalfa ploughing, soil should remain covered during fall-winter season with CC or winter cereal.

6.5.4. The importance of catch crops

Imposed by the Nitrate Directive (91/676/EEC), in 2012, the use of CCs should have reached 100% of the bare soil surface, however some organic fields are exempted due to late spiked chain harrows in order to reduce weeds propagation. In this study, the mixed-seed CCs reduce N contamination of 60%, in accordance with most studies which conclude in a positive effect of CCs on NO_3^- leaching, e.g., reductions from 38 to 70% for mustard (Hooker et al. 2008), from 50 to 79% for radish (Justes et al. 1999), of 71% for chicory and of 67% for ryegrass (Sapkota et al. 2012). However, very few studies have discussed the effect of mixed-seed CCs, which has become more frequent, especially in organic agriculture, showing greater benefits on crop biomass and NO_3^- leaching reduction, than with a single seed (Rinnofner et al. 2008). Moreover some farmers may wish to substitute CCs with green manure (CF2), which is a matter of particular concern. Indeed the long-term effect of green manures would increase the risk of leaching (Moller et al. 2008) and even for CCs such as mustard, an increase from 9 to 26 kg N ha⁻¹ yr⁻¹ is shown by Constantin et al. (2011) based on 13 years of model simulations. As for legumes, in order to minimise the risk of N leaching, several studies suggest incorporating green manure in spring rather than in autumn (Kankanen et al. 2008), but with a possible negative effect on the yield that needs to be examined, as for ryegrass incorporation (Aronsson et al. 2007). Results are still lacking for irrevocable recommendations and further studies on NO_3^- leaching are needed in order to optimise N management in organic rotations.

6.5.5. Room for improvement in conventional and organic systems

In conventional systems, a variety of N management methods are also possible, such as no-tillage, low N input and CCs... Studies on no-tillage systems have shown a decrease of NO_3^- leaching by a factor of two to four (Angle et al. 1993; Drury et al. 1993). However, as observed on CF2, the association of exogenous N inputs and BNF in a no-tillage system can still lead to substantial NO_3^- leaching. Moreover, the values obtained on CF3 (18–46 mg $\text{NO}_3\text{-N l}^{-1}$), are close to the measurements obtained for experimental trials in North of France in the 1990s (between 18 and 33.5 mg $\text{NO}_3\text{-N l}^{-1}$: Arlot and Zimmer, 1990; Machet and Mary, 1990; Chapot 1990; Denys, 1990). In contrast, to protect a drinking water spring, the association of low N input and CCs has reduced the N contamination from 16 to 9 mg $\text{NO}_3\text{-N l}^{-1}$ (Beaudoin et al. 2005). In the organic cropping systems studied, most organic fertilisers are applied in fall, due to rainy conditions in spring (especially in S&M), preventing field work with farm machines. Together with the period of application, the types of organic fertilisers used can control soil solution concentrations and leaching during the drainage period. For example, poultry manure (OF2) and vinasse (OF1, OF3, OF4) are highly mineralised in contrast to manure, which can, however, lead to a long-term impact (Bergstrom and Kirchmann 1999). On the other hand, we have shown that alfalfa N management can be improved, especially by exporting part of the harvest or by ploughing it as late as possible within the drainage period.

This study shows that there is room for progress in both systems regarding N management to reconcile good water quality and sustainable agriculture in a single area.

6.6. Conclusion

For CF and OF, substantial variations in soil solution N concentrations and N leaching stem from management practices, in terms of fertiliser application timing, quantity and quality, e.g., the combination of different sources of N such as legumes in green manure, as well as mineral and organic fertiliser. The soil solution concentrations were proportional to the amount of organic fertilisers applied, however the relationship is no longer valid in presence of catch crops, horse manure or for crops after legumes. Appropriate N management is an objective to ensure crop growth and to limit nitrogen leaching. In literature, most studies on NO_3^- leaching from organic systems do not include alfalfa (2 or 3 years), as we experiment here in the Parisian basin. Furthermore, reported results generally concern short rotations (4 years), whereas they are rather long in the Seine Basin (7 years). We have shown that in addition to agricultural practices and N management, various other factors such as soil properties and climate contribute to NO_3^- leaching, so that no significant difference between organic and conventional systems in terms of flux of N leaching has been evidenced to date, but a gradient of leaching between the terms of the rotations clearly appears (alfalfa < crops with CC < legumes < crops without fertilisation < crops fertilised in fall or after legumes). In terms of soil solution concentrations of infiltrating water, however, organic cropping systems in a given pedo-climatic context show better performance than their conventional counterparts. When yield-scaled, leaching appear to be in a similar range for both OF and CF. At last, more studies are needed at the system scale, to improve N management in order to reduce NO_3^- concentration in infiltrating water and thus protect the quality of the water resource.

Acknowledgements

The study is funded by a PhD scholarship and an equipment sponsorship from the DIM Astrea of the Ile-de-France Region and the Water Agency of the Seine-Normandy Basin. The CNRS-ZA-Seine (CNRS-Zone Atelier-Seine) and the ANR Escapade programs are also acknowledged for providing funding for the analysis. Météo- France and Irstea stations are gratefully acknowledged for providing weather data. Many thanks are extended to the farmers (Mr Gobard, Mr Collin, Mr Dupuis, Mr Lefevre, Mr Leturq and Mr Gilloots), who allow us to work on their farms and help us when possible. We also sincerely thank Benjamin Mercier and Abdelkader Azougui for their kind laboratory and field technical assistance. We also greatly acknowledge the referees for their detailed and constructive revision.

7

A large variability of leaching in organic and conventional farms

Benoit M¹, Garnier J^{1,2}, Billen G^{1,2}

¹ UPMC, UMR 7619 Metis, Box 105, 4 place Jussieu, 75005 Paris, France

² CNRS, UMR 7619 Metis, Box 105, 4 place Jussieu, 75005 Paris, France

Ce chapitre est un article en préparation qui fera prochainement l'objet d'une soumission probablement dans la revue scientifique *Agricultural Systems*.

7.1. Abstract

In the Seine basin, characterized by intensive arable crops, most of the surface and groundwaters are contaminated by nitrate (NO_3^-). The goal of this collaborative study involving a network of volunteer farmers is to investigate NO_3^- leaching with ceramic cups (90 cm deep) on the whole crop rotations of organic (OF) and conventional (CF) commercial farms in the Seine Basin. In total, 10 CF and 8 OF systems were studied in six different soil and climate conditions, taking into account a wide diversity of practices (including low or no exogenous N inputs, systematic catch-crops implementation, no-tillage,...) and different organic fertilisers (including biogas residue, slurry, poultry, cow and horse manures).

Overall, OF farms (rotations with alfalfa) led to a lower average sub-root concentration ($8.7 \pm 2.8 \text{ mg N l}^{-1}$), than CF systems ($12.6 \pm 6.7 \text{ mg N l}^{-1}$). The calculated water inflows ranged between 92 and 164 mm for the four months drainage period. The amounts of N leached in OF and CF were between 3.7-16.9 and 4.5-33.6 kg N ha^{-1} respectively, according to the different farming practices, soil and climate conditions. The main controlling factors were the use of organic fertiliser in fall, the lack of catch-crops before spring crops and the proportion of legumes in the rotation. Overall, this wide collaborative network highlights good and innovative practices by CF and OF farmers in order to reduce NO_3^- in groundwater.

7.2. Introduction

Nitrate (NO_3^-) pollution in groundwater from agriculture is a major concern for the European Union (Nitrate Directive, n°91/676/CEE). Leached N represents a risk for the drinking water quality and a higher cost for the producers. The first studies on N leaching in the Seine basin (France) were performed in the 90's for conventional farming (CF) systems (Arlot and Zimmer, 1990; Machet and Mary, 1990; Chapot, 1990; Denys, 1990; Lemaire and Nicolardot 1997) and concentrations exceeded the drinking water standard (11 mgN l^{-1} or $50 \text{ mg NO}_3 \text{ l}^{-1}$). Nowadays, most of the studies dedicated to N leaching in the Seine basin are testing “good agricultural practices” (GAP) in experimental trials (Constantin et al. 2010) but a few in commercial farms (Beaudoin et al. 2005) and generally not in commercial organic farming (OF) systems. However at a regional scale, farming practices in commercial farms are wide and diversified based on different systems, techniques and rotations. In the international literature, studies on OF and CF agro-environmental performances have led to the controversial conclusions regarding the respective merit of both systems, with wide variations according to regional conditions (Mondelaers et al. 2009; Tuomisto et al. 2012). We already compared OF and CF systems in term of N leaching in the Seine basin but with a low diversity on CF systems, showing 46 % less leaching in OF than in CF systems from the same regions (Benoit et al. 2014).

In this new study, our aim is to quantify the N leached in a wider panel of CF and OF systems from different climate and soil conditions within the Seine basin. To catch the variability of nitrate losses, 10 CF and 8 OF systems including several rotations were equipped with porous ceramic cups, which can be set up quickly without any destruction of the soil horizons of the plots (Stopes et al. 2002). Sampled water from the soil solution at 90 cm deep during the autumn and winter drainage period can be directly analysed for NO_3^- concentrations, and considered representative of sub-root water concentrations, infiltrating to the hydrosystem. The results are interpreted in terms of crops and/or practices during the on-going and preceding cropping period. The role of legumes as a precedent crop was particularly interesting to document due to the impact of biological N fixation (BNF) on leaching. The effect of catch crops, the implementation of which is mandatory (but not always applied) before spring crops on vulnerable zone since 2012, is also an important issue. Such data are essential to quantify the effects of possible agricultural changes in terms of ground- and surface water nitrate contamination and coastal eutrophication in the Seine watershed (Garnier et al. 2010; Thieu et al. 2011).

7.3. Materials and methods

7.3.1. Localisation and characteristics of the main areas

The Seine basin climate is humid and temperate, with a large gradient of temperature and pluviometry. Agricultural land covers more than 60% of the total surface. Arable crops occupy 60% of the utilised agricultural land (UAL) with winter wheat as the dominant crop (29% UAL). 18 agricultural systems, 8 in OF and 10 in CF, were studied in 15 farms (three of those are mixed CF and OF systems), in six different pedoclimatic conditions in the Seine Basin (Figure 7-1).



Figure 7-1. Localisation of the farms studied in 2013-14 in the Seine basin.

The first area studied is located in the East of Paris in Seine & Marne (S&M) (**Figure 7-1**, n°1), characterized by a deep loamy soil (luvisol) generally drained, due to hydromorph conditions. The mean annual rainfall is 676 mm and the mean annual temperature 11.3°C (Meteo France weather station, Melun). In S&M, farming is based on cereals, sugar beet, maize and faba beans productions and OF counts for 1.1% UAL ([Agreste, 2012](#)). In this area, seven farms have been studied, including three OF and five CF systems.

The second area is located in the North of the basin in the Oise department (**Figure 7-1**, n°2), characterized by a chalky substratum, which is present in almost all the periphery of the Seine basin. The annual average rainfall is 697 mm and temperatures 9.7°C (Meteo France weather station, Saint Quentin). In this area, the agriculture is known for cereals and sugar roots production (40% of national production with S&M) and OF counts for 1.3% UAL ([Agreste, 2012](#)). In this area, three farms have been followed, including two CF and two OF systems.

The third area is in the South-East of the Seine basin in Yonne (**Figure 7-1**, n°3), and is characterized by a chalky substratum, with an average annual rainfall of 880 mm and mean temperature of 10.7°C (Meteo France weather station, Cruzy). The percentages of oat and oilseed rape are higher in Yonne than in other areas and OF represents 4.1% UAL ([Agreste, 2012](#)). In this area, two farms has been investigated, including each system, OF and CF.

Three additional sites have been studied in the West side of Paris, where agriculture is also mainly dominated by wheat, maize and oleaginous crops and OF under 2% UAL. One OF system is located in Val d'Oise (**Figure 7-1**, n°4), characterized by a mean rainfall of 670 mm, 10.7°C mean temperature and a deep clay-loamy soil (Meteo France weather station, Villiers-Adam). In this area, agriculture is dominated by wheat (59%), maize (14%) and oleaginous (17%) and OF represents around 1% UAL ([Agreste, 2012](#)). A conventional farm has been also studied, located in Yvelines (**Figure 7-1**, n°5), with 675 mm mean rainfall, 11.2°C mean temperature and a deep loamy-clay soil (Meteo France weather station, Villacoublay).

Agriculture is dominated by wheat (53%), barley (12%) and oleaginous (23%), whereas OF represents about 2% UAL (Agreste, 2012).

Finally, a mixed farm (OF+CF) is located in the Centre region (Figure 7-1, n°6), characterized by an annual rainfall of 599 mm, a mean temperature of 11°C and a clay soil on a chalky substratum (Chartres, weather station). Agriculture is dominated by 60% wheat, 32% oleaginous, 20% barley and 8% maize and OF represents below 0.5% UAL (Agreste, 2012).

7.3.2. Description of the organic and conventional systems

In total, 8 OF systems and 10 CF systems have been studied according to different farming practices and locations. In OF, the rotations last generally seven to nine years, starting with two to three years of alfalfa, wheat, cereal, legumes, wheat, cereal (Table 7-2 a). In S&M, the three OF also used exogenous N inputs as vinasse (OF1), poultry manure (OF2) and horse manure (OF3). In Oise, the two systems studied usually used vinasse (OF4 and OF5), but during the year of experimentation no vinasse was applied. For the organic farms studied in Yonne (OF6) and Val d'Oise (OF7), the systems are only based on legume BNF. In Centre, three fields have been instrumented (OF8).

The 10 CF systems have been studied in parallel in the same areas (Table 7-2 b). The rotations studied lasted 2 to 5 years. In S&M, CF1 present legumes in its rotation (wheat – faba beans – wheat); CF2 is a two-year rotation (wheat – maize with faba beans as CC) with no-tillage; CF3 operates a common rotation of the region (sugar beet-wheat-maize) fertilised both with mineral and cow manure; CF4 has a common rotation (wheat-rapeseed-maize) with systematic and diverse CC; CF5 (rapeseed-wheat-sugar beet) has low mineral fertilisation. In Oise, two CF systems are studied, one using only mineral fertilisation on wheat-flax-rapeseed (CF6) and the other one, using both mineral and biogas residue on wheat-barley-sugar beet (CF7). In Yonne, the rotation instrumented (wheat-oat-rapeseed or maize) is common in the region and uses mineral and manure fertilisation (CF8). In Yvelines, the farm studied uses low mineral inputs, slurry and compost as exogenous N input on wheat-barley-rapeseed and maize (CF9). In Centre, one field has been instrumented (wheat after maize) (CF10).

Concerning exogenous N inputs, the OF systems used in average 77 kg N ha⁻¹ of organic fertiliser whereas CF systems, applied around 170 kg N ha⁻¹ yr⁻¹. Organic fertilisers used both by OF and CF systems are horse manure, poultry manure, slurry and biogas residue. Mineral fertilisation in CF is generally a combination of ammonitrate and solutions of urea ammonium nitrate.

In total, 49 fields were instrumented in OF and 30 in CF, due to the shorter CF rotation. The CF crops counted wheat (50%), barley (14%), oilseed rape (14%), maize (18%) and leys (4%), in exactly the same proportion as in the Seine basin.

7.3.3. Field measurements

7.3.3.1. Ceramic cups

Ceramic cups have been used since 1904 to measure NO_3^- in groundwater (Briggs and McCall 1904). Their recognized advantages are their cost, effectiveness and ease of installation which have made them the most commonly used devices for collecting water. The 79 fields are equipped with ceramic cups of 85 cm length, (SDEC, France, SPS Ø 31mm), all the soils of the network being at least 90 cm deep. Ceramic cups (6 per plots) were installed with a manual auger of the same diameter, arranged on a line parallel to the soil tillage with the head place below the ground surface (-5 cm) to allow shallow tillage. They are placed at a minimum of 14 m distance from the edge of the field in order to avoid any side effect. Samples are taken weekly, after 48 h of vacuum setting, at the beginning of the rainy hydrological season, when the soil is water saturated (first month) and fortnightly for the rest of the sampling period.

7.3.3.2. Soils samples

During the implementation of the ceramic cups, soil samples are extracted before the sampling period in each field. The six samples of soils are extracted with the auger and homogenized following three layers [0-30], [30-60] and [60-90] cm. The fresh samples are stored at 4°C until analysis of humidity and then frozen at -18°C for further analysis (texture and C, N content).

7.3.4. Analyses

7.3.4.1. Soils analysis

Triplicates of soil are weighed (30 g) to determine moisture and soil organic matter (SOM) using the loss-on-ignition method. Soil samples are heated at 105 °C (48 h) for the former, and calcined at 450 °C (4 h) for the latter, and re-weighed after each step. Soil mineral nitrogen concentrations are determined after KCl extraction, with 5 g of soil in 20 ml of KCl (2 M) for 2 h on a shaking table. The suspensions are centrifuged for 10 min at 3000 rpm and the supernatant is frozen at -18 °C until analysis in the autoanalyzer (Quatro, Bran & Luebbe). The rest of the sample is freeze-dried to determine the particle size distribution (without decarbonation), soil organic carbon (SOC) (after decarbonation), and loss of ignition at 1000°C, and total N (LAS, INRA Arras).

7.3.4.2. Water analysis

The water taken from each ceramic cup is frozen until analysis of the N concentrations using an autoanalyzer Gallery (Thermofisher). The method used to measure ammonium (NH_4^+) is based on the reaction of the blue indophenol (Slawyk and MacIsaac 1972). Nitrite (NO_2^-) and NO_3^- are measured using the sulphanilamide method (Jones 1984) and NO_3^- concentrations are determined after reduction to nitrite.

7.3.5. Percolating water flux

We used a simple water balance modeling of a one layer reservoir in order to calculate the daily water percolation (W_i , mm d⁻¹) during the sampling period.

The soil water content at field capacity (WHC_{FC} , mm) over 0-90 cm was defined using a pedotransfer function (Bruand et al. 2004). The daily water storage (WS_i) was calculated using the daily precipitation (R_i , mm) and the reference evapotranspiration (ETP_i , mm) multiplied by a crop coefficient (k_1) (Morizet et al. 1984), fixed at 0.5 during the winter period (Perrier et al. 1980; Allen, 2000). The model operates at a daily time step to calculate the water storage (WS_i , mm d⁻¹) (Equation 7-1). Once the WS_i reaches the field capacity (WHC_{FC}) the excess water is lost as percolation (Equation 7-2) with a velocity coefficient (k_2) based on Benoit et al. (2014).

Equation 7-1. $WS_i = \max (WS_{i-1} + R_i - k_1 * ETP_i - W_{i-1}; 0)$

Equation 7-2. $W_i = \max (k_2 * (WS_i - WHC_{FC}); 0)$

The N leached flow was calculated between each collection date, by multiplying the average concentration by the amount of percolated water. N leached was measured for a six months sampling period, from December to May and is assumed to represent the total leaching of the year, as vegetation uptake and evaporation prevent leaching during the rest of the year.

7.3.6. Statistical analysis

All the statistical analyses are performed using R software. Differences within data sets are analysed using the Kolmogorov-Smirnov test to verify normality. Analysis of variance tests (ANOVA, Student, *P-value* < 0.10) are determined for the data set, due to the important variations between sample conditions.

7.4. Results

7.4.1. Climate and soil conditions

The soil parameters have been established for each field layer. However in order to characterize the soil of each region, regional means were used (**Table 7-1**). Most soils are silty clay loam, except in Yonne and in Centre area where the sand or the clay contents are higher. Nutrients (SOC, N tot) are higher in Yvelines and lower in Yonne. The bulk density (BD) was 1.5 and the WHC_{FC} were determined in S&M (189 mm), Oise (199 mm), Yonne (168 mm), Val d'Oise (209 mm), Yvelines (167 mm), Centre (183 mm) based on [Bruand et al. \(2004\)](#).

Table 7-1. Soils characteristics in the regions studied in terms of textures (clay, sand), soil organic carbon (SOC), N total (N tot), bulk density (BD) and water holding capacity at field capacity (WHC_{FC}).

Soil parameters	Depths (cm)	S&M	Oise	Yonne	Val d'Oise	Yvelines	Chartres
Clay	0-30	20	24	19	18	24	30
g 100 g ⁻¹	30-60	23	27	24	24	24	30
	60-90	25	32	31	27	27	29
Sand	0-30	11	9	24	8	13	8
g 100 g ⁻¹	30-60	10	7	21	7	14	9
	60-90	10	8	22	6	12	14
SOC	0-30	1.07	1.37	0.99	0.86	1.89	1.39
g 100 g ⁻¹	30-60	0.55	0.83	0.52	0.52	0.97	0.77
	60-90	0.43	0.52	0.35	0.44	0.55	0.46
N tot	0-30	0.10	0.12	0.08	0.08	0.15	0.13
g 100 g ⁻¹	30-60	0.06	0.08	0.05	0.05	0.08	0.08
	60-90	0.05	0.05	0.03	0.05	0.04	0.04
BD g cm⁻³	0-90	1.53	1.54	1.53	1.53	1.53	1.58
WHC_{FC} mm	0-90	189	199	168	209	167	183

7.4.2. Variations of sub-root concentrations

7.4.2.1. Variations according to crops

In order to determine the crop impact on water contamination, all the crops studied in OF have been integrated into a typical nine years rotation, as alfalfa 1, alfalfa 2, alfalfa 3, wheat, cereal, legumes, wheat, cereal, cereal (**Figure 7-2 a**). Alfalfas show the lowest sub-root concentrations (SRC), $4.35 \pm 4.15 \text{ mg N l}^{-1}$ during the three years. Wheats after alfalfa ploughing have the highest SRC ($21.74 \pm 8.52 \text{ mg N l}^{-1}$), due to an important BNF inputs. Crops cultivated two years after alfalfa show SRC mean of $12.07 \pm 7.24 \text{ mg N l}^{-1}$. Legumes have a SRC mean of $7.51 \pm 4.75 \text{ mg N l}^{-1}$. Wheats after legumes show SRC $12.21 \pm 10.58 \text{ mg N l}^{-1}$. At the end of the rotation, the SRC for crops without organic fertilisation are lower ($< 10 \text{ mg N l}^{-1}$) than after fertilisation ($> 14 \text{ mg N l}^{-1}$).

Likewise, the CF crops were classified in two typical rotations from this region: rotation 1 (rapeseed, wheat, barley) and rotation 2 (maize/sugar beet – wheat – cereals) (**Figure 7-2 b, c**). In the first rotation, the lowest concentrations were found for rapeseed and barley with average SRC of $6.29 \pm 2.33 \text{ mg N l}^{-1}$ and $4.42 \pm 1.64 \text{ mg N l}^{-1}$, respectively. Those two crops have the particularity to be sown early and to have an early growth. In the second rotation, the lowest SRC was measured on CC ($4.62 \pm 2.58 \text{ mg N l}^{-1}$) and the highest on wheat after spring crops (maize or sugar beet) with CC (including legumes) with a mean of $21.54 \pm 14.06 \text{ mg N l}^{-1}$.

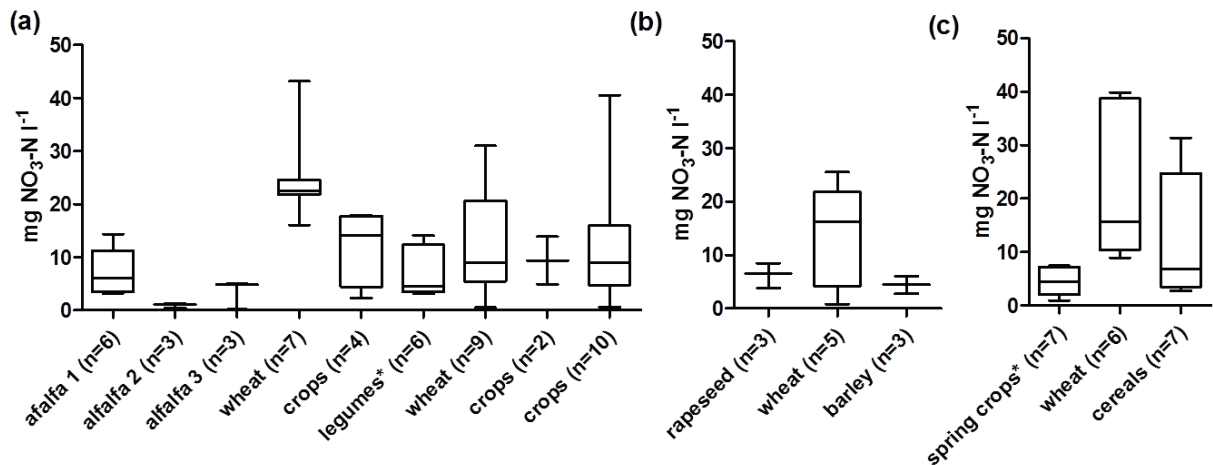


Figure 7-2. Sub-root concentrations means for crops in organic (a) and conventional (b and c) rotations studied in 2013-14 in all the farms. (*) stands for spring crops (maize, sugar beet) or legumes with or without catch-crops (CC) (n= number of field studied).

Table 7-2. Summary of the organic (a) and conventional (b) fields instrumented with ceramic cups in 2013-14 with their N inputs and sub-root concentrations (SRC). Fertiliser types: biogas residue (BGR), cow manure (CM), horse manure (HM), mineral (Min), poultry manure (PM), slurry (S). (*) stands for spring crops with catch-crops and (**) stands for spring crops with bare soil.

(a)

system	field	crops in 2013-14	previous crops	fertiliser type	N inputs kg N ha ⁻¹	SRC mg N l ⁻¹
OF1	1	wheat	alfalfa 2			18
OF1	2	wheat	beans			6
OF1	3	wheat	alfalfa 2			24
OF1	4	alfalfa 1	wheat			5
OF1	5	beans *	flax*			5
OF2	6	oat	wheat	C	140	14
OF2	7	wheat	alfalfa 2			23
OF2	8	triticale	wheat	C	140	15
OF2	9	flax**	wheat			17
OF2	10	wheat*	alfalfa 1			5
OF2	11	wheat	flax	C + PM	215	39
OF2	12	wheat	lentils**	C	140	27
OF2	13	faba beans**	triticale			16
OF2	14	wheat*	oat			0
OF3	15	lentils**	wheat			11
OF3	16	trefoil 3	trefoil 2			0
OF3	17	triticale	wheat			17
OF3	18	wheat	lentils	HM	70	9
OF3	19	wheat	faba beans**			5
OF3	20	sainfoin 2	sainfoin 1			1
OF4	21	alfalfa 3	alfalfa 2			5
OF4	22	alfalfa 2	alfalfa 1			0
OF4	23	spelt	oat			13
OF4	24	peas-oat	wheat			14
OF5	25	wheat	lentils_rye			10
OF5	26	alfalfa 2	alfalfa 1			1
OF5	27	wheat	faba beans**			1
OF5	28	peas-trit	wheat			4
OF5	29	wheat	vegetables			7
OF5	30	faba beans*	wheat			4
OF5	31	maize**	wheat			3
OF5	32	faba beans	maize			3
OF6	33	wheat	faba beans**			9
OF6	34	alfalfa 3	alfalfa 2			5
OF6	35	wheat	alfalfa 3			15
OF6	36	alfalfa 1	wheat			7
OF6	37	soya*	wheat			4
OF6	38	oat	rye			5
OF7	39	wheat	alfalfa 2			24
OF7	40	alfalfa 2	alfalfa 1			1
OF7	41	faba beans	sunflower			4
OF7	42	sunflower	wheat			14
OF7	43	wheat	faba beans			4
OF7	44	alfalfa 1	oat			18
OF7	45	wheat	peas+trit			5
OF7	46	peas+trit	wheat			5
OF8	47	wheat	faba beans*	C	70	31
OF8	48	wheat	alfalfa 3			39
OF8	49	alfalfa 1	oat			4

(b)

system	field	crops in 2013-14	previous crops	fertiliser type	N inputs kg N ha ⁻¹	SRC mg N l ⁻¹	SMN in fall 2013 kg N ha ⁻¹	SMN in spring 2014 kg N ha ⁻¹
CF1	1	wheat	faba beans	Min	200	14.0	46	49
CF1	2	wheat	wheat	Min	150	13.4	84	100
CF2	3	wheat	maize*	Min	180	34.0	129	151
CF2	4	maize*	wheat	Min	190	7.4	41	48
CF3	5	wheat	sugar beet	Min	200	9.3	14	84
CF3	6	wheat	wheat	Min	200	25.8	31	121
CF3	7	wheat	maize	Min	200	30.9	39	59
CF4	8	wheat*	wheat	Min	180	7.6	22	95
CF4	9	rapeseed	wheat	Min	180	8.0	20	15
CF4	10	maize*	wheat	Min	180	2.1	13	225
CF5	11	wheat	rapeseed	Min	170	0.9	14	37
CF5	12	rapeseed	row barley	Min	150	5.2	10	60
CF5	13	sugar beet*	wheat	Min	170	7.2	20	101
CF6	14	wheat	rapeseed	Min	230	27.2	48	132
CF6	15	wheat	wheat	Min	230	19.1	52	83
CF6	16	flax**	wheat	Min	230	4.8	17	14
CF7	17	wheat	peas*	BGR	40	35.0	78	45
CF7	18	barley	wheat	BGR	20	6.8	30	18
CF7	19	sugar beet*	oat	BGR	80	8.7	29	54
CF8	20	maize*	oat	Min + CM	138	1.3	17	20
CF8	21	rapeseed	oat	Min	135	4.5	30	153
CF8	22	wheat	rapeseed	Min	80	17.2	74	90
CF8	23	barley	wheat	Min	185	6.6	21	132
CF9	24	wheat	rapeseed	S	110	7.4	15	82
CF9	25	wheat	rapeseed	S	110	20.1	78	33
CF9	26	wheat	maize*	Min + CM	177	16.6	40	52
CF9	27	wheat	alfalfa 2	Min	92	19.6	25	227
CF9	28	barley	wheat	S	177	52.0	150	153
CF9	29	maize*	wheat	S	90	3.9	32	18
CF10	30	wheat	maize	Min	80	11.5	51	138

7.4.2.2. The role of catch-crops

For spring crops, SRC were measured either on bare soil, on growing CC or after CC ploughing (**Table 7-3**). On bare soil, SRC mean was significantly higher ($10.95 \pm 5.85 \text{ mg N l}^{-1}$) than with CC (4.16 ± 2.49) (t-test, $P\text{-value} = 0.025$). Different CC varieties were grown from one to ten species with two different ploughing dates (spring, fall) (**Figure 7-3**). The diverse CC combinations did not have any impact on SRC. Fields sampling were either carried out after CC ploughing or during CC growth in CF and OF, however no significant difference was observed between both treatments ($P\text{-value} = 0.53$), with SRC mean for early ($4.86 \pm 3.09 \text{ mg N l}^{-1}$) or late ploughing ($3.56 \pm 1.88 \text{ mg N l}^{-1}$).

Table 7-3. Catch-crops diversity in organic and conventional fields (and farms) and associated sub-root concentration (SRC).

Field	Farm	sowing -harvest dd/mm - dd/mm	catch crops variety	SRC mg N l ⁻¹
4	CF2	01/09 - 10/03	faba beans	6.74
8	CF4	15/08 - 26/10	faba, beans, flax, mustard, sunflower, rape	7.96
10	CF4	20/08 - 01/04	faba, beans, phacelie, oat, clover, oat, vetch, snflower, sarazin, rape	2.11
13	CF5	01/09 - 15/12	vetch, radish, oat, phacelia	7.19
19	CF7	01/09 - 01/12	white mustard	7.42
20	CF8	01/09 - 01/04	aot-vetch	0.95
29	CF9	01/09 - 01/03	oat, vesce, clover, placelia	3.50
5	OF1	01/09 - 01/04	flax, oat, clover	4.99
10	OF2	01/09 - 01/02	alfalfa	3.59
14	OF2	01/09 - 03/03	white clover	0.58
29	OF5	27/08 - 15/12	white clover	3.66
30	OF5	05/09 - 15/12	faba, clover, oat	2.31
37	OF6	01/09 - 30/02	rye	3.06

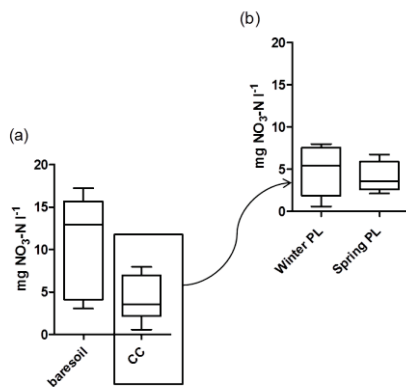


Figure 7-3. (a) Means sub-root concentration of spring crop fields on bare soil (n=6) and with CC (n = 13), (b) the latter being split into winter ploughing (PL) (n=6) and spring ploughing (PL) (n=5).

7.4.2.3. Influence of fall fertilisation

Fields with organic fertiliser application in fall have been observed in organic ($n = 6$) and conventional fields ($n = 5$). The SRCs depend on the type and the amount of fertiliser applied (**Figure 7-4**). The SRC increased immediately for organic fertiliser with a C/N < 15, as biogas residue, slurry or poultry manure. However, when the organic fertilisers had C/N > 15, the SRC measured were lower, as for cow manure or horse and cow manure. Furthermore, the presence of CC after the biogas residue application (80 kg N ha^{-1}) reduced the SRC by 85%.

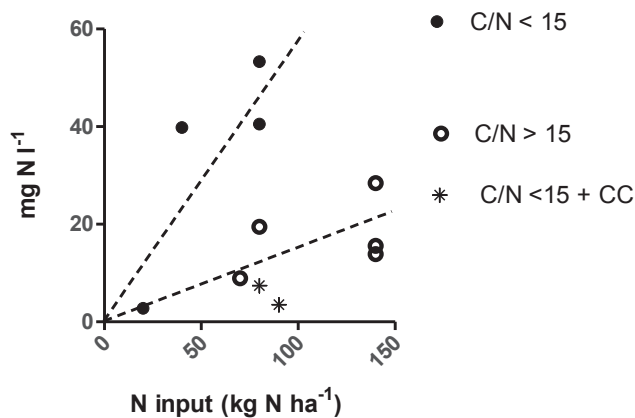


Figure 7-4. Sub-root concentration (SRC) in relation to N exogenous input for different type and amount of organic fertiliser applied in fall in organic or conventional fields and the impact of CC after organic fertiliser.

7.4.2.4. Variations among CF and OF systems

The SRC at the rotation scale showed important variations from 4.1 to 22 mg N l^{-1} , according to the farms systems (**Figure 7-5**).

In S&M, the SRC were the lowest in OF3 (7.1 mg N l^{-1}), CF4 (5.9 mg N l^{-1}) and CF5 (4.4 mg N l^{-1}), in relation to perennial crops (OF3), low inputs (CF5) or systematic used of CC (CF5). For the other farms in S&M, SRC means are over 10 mg N l^{-1} . However in OF1 (11.4 mg N l^{-1}), the incomplete rotation contained 60% of legumes ploughing. The highest SRC in OF was measured in OF2 (17.3 mg N l^{-1}) mainly due to poultry manure application in fall. The highest SRC in CF rotation were observed in CF2 (20.7 mg N l^{-1}) and CF3 (22.0 mg N l^{-1}), due to the systematic introduction of green manure (faba beans before maize) or important N input, respectively.

In Oise, the OF systems studied present lower SRC than in CF. The OF4 (4 years) and OF5 (8 years) SRC means were 10.7 and 4.1 mg N l^{-1} respectively. The CF rotations showed higher SRC, as in CF6 (17.0 mg N l^{-1}) and in CF7 (16.6 mg N l^{-1}), mainly due to important mineral (230 kg N ha^{-1}) or biogas residue application in fall, respectively.

In Yonne, the SRC means were quite low in both systems, with 6.8 mg N l⁻¹ (OF6) and 7.4 mg N l⁻¹ (CF8), in relation to no exogenous input and good N management respectively. In the West region (OF7, CF9), the SRC were 9.4 and 19.9 mgN.l⁻¹ respectively, mainly due to no N exogenous input and slurry application in fall respectively. The fields instrumented from Centre region have not been considered here due to a lack of data on the full rotation, but all data are summarized in **Table 7-4**. Overall the SRC measured in OF and CF systems are 11.9 and 13.3 mg N l⁻¹ in S&M, 7.4 and 16.9 mg N l⁻¹ in Oise and 6.8 and 7.7 mg N l⁻¹ in Yonne, respectively.

Table 7-4. Means of sub-root concentrations (SRC), water inflow and N leached on every farm.

Region	Farms	SRC <i>mg N l⁻¹</i>	Water inflow <i>mm</i>	Leaching <i>kg N ha⁻¹</i>
S&M	OF1	11.4	96	11.7
S&M	OF2	17.3	97	16.9
S&M	OF3	7.1	92	6.7
S&M	CF1	13.7	96	11.3
S&M	CF2	20.7	96	20.9
S&M	CF3	22.0	96	21.1
S&M	CF4	5.9	96	5.9
S&M	CF5	4.4	97	4.5
Oise	OF4	10.7	96	9.8
Oise	OF5	4.1	96	3.8
Oise	CF6	17.0	96	16.8
Oise	CF7	16.8	96	15.9
Yonne	OF6	6.8	132	7.7
Yonne	CF8	7.4	133	9.0
Val d'Oise	OF7	9.4	30	3.7
Yvelines	CF9	19.9	157	33.6
Centre	OF8	24.5	104	26.8
Centre	CF10	11.5	104	12.3

7.5. N Leached in conventional and organic systems

The leached NO₃⁻ results from SRC value multiplied by water inflow (Wis) from each region (**Table 7-4**). In S&M and Oise, the cumulative Wis were between 92-97 mm during the sampling periods. In consequence, the mean NO₃⁻ leached in OF and CF are 11.8 kg N ha⁻¹ and 12.7 kg N ha⁻¹ in S&M and 6.8 kg N ha⁻¹ and 16.3 kg N ha⁻¹ in Oise, respectively (**Figure 7-5**). In Yonne, the NO₃⁻ leached was slightly lower in OF6 (7.7 kg N ha⁻¹) and in CF8 (9.0 kg N ha⁻¹), for a mean water percolation of 132.5 mm.

Then in the West region, the amount of water is much higher in CF9 (157 mm) than in OF7 (30 mm), in consequence the amount of NO₃⁻ leached is ten times higher in CF9 (33.6 kg N ha⁻¹) than OF7 (3.7 kg N ha⁻¹).

As a major result of this study, taking into account the OF and CF rotations means in each regions (S&M, Oise, Yonne), leaching was lower in OF (8.7 ±2.6 kg N ha⁻¹ yr⁻¹) than in CF (12.7 ± 3.6 kg N ha⁻¹ yr⁻¹) (*P-value* = 0.14).

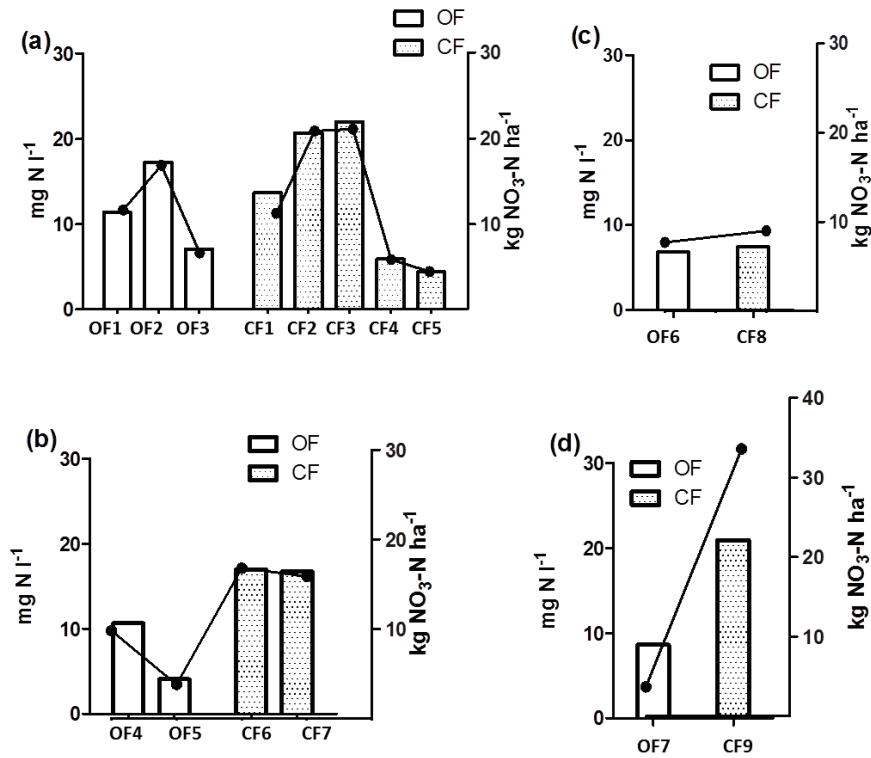


Figure 7-5. Sub-root concentrations means (in histograms) and N leached (black lines) in OF and CF systems studied in S&M (a), Oise (b), Yonne (c) and the West (Val d'Oise, Yvelines) (d).

7.6. Discussion and conclusions

7.6.1. Farmers strategies

In this network, we observed three main farmers strategies regarding fertilisation management. The first strategy focused on “optimizing crop effect”, as introducing diversified CC before every crop or only using the BNF as N inputs. This strategy showed the lowest concentrations, in accordance to most studies referring to CC efficiency (Justes et al. 2012) and the systems based on alfalfa (Beaudoin et al. 1992). The second strategy focused on “optimizing N use efficiency” (low input, only one type of fertiliser). Their mean N leached was between 4.5 to 11.3 kg N ha⁻¹ yr⁻¹, in accordance to the GAP in the Seine basin (Beaudoin et al. 2005). Then the third strategy focused on the “N inputs combination”, cumulating the use of legumes with the input of organic and/or mineral fertilisers. This last strategy showed the highest NO₃⁻ concentrations (up to 33.6 7 kg N ha⁻¹ in CF9), but was rarely studied in the literature which usually compares one treatment separately from the others.

7.6.2. Representativeness of the network

The CF rotations studied took into account a large panel of farming practices, similarly to the situation in the Seine basin. However, even if the crops types' proportion is respected in our sample with respect that of the whole basin, the mean sub-root concentrations found may be underestimated, as our sample, particularly for CF, possibly over-represents good agricultural practices. The conventional farms were indeed selected among volunteer farmers, pioneers for applying new practices (e.g. catch crop regulation) or even keen on biological systems. Typically the lowest CF concentrations were measured in farms with agro-environmental measures (CF4, CF5). Instead in the Seine basin, the proportion of farmers using CC represents no more than 50% UAL of spring crops (Agreste, 2013), a lower proportion than in our sample (70 %) in spite of the recommendations made in the scope of the Nitrate Directive in vulnerable zones.

On the other hand, our OF rotations sample was well representative of the OF in the Seine basin. OF rotations much longer than the CF ones are indeed mostly of 9 years (with 2 or 3 years of alfalfa) and often located in the deep soil sectors of the Seine basin (50% OF rotation, Agence Bio). Therefore the thin soils (depth < 90 cm) especially present in the South-West part of the Seine basin were not sampled, either in OF or CF system.

7.6.3. Organic and conventional farming leaching

As a major result of this study, taking into account the OF and CF rotations means in each regions (S&M, Oise, Yonne), the leaching means in OF and CF are 8.7 ± 2.6 and 12.7 ± 3.6 kg N ha⁻¹ yr⁻¹ respectively. In addition to previous studies in the Seine basin, in one mixed farm (OF/CF) followed during three years (Benoit et al, submitted, cf. chap. 3) and in 3 CF and 4 OF systems in two regions (S&M, Oise) (Benoit et al. 2014), the whole SRC means in OF and CF farms are 10.1 ± 3.4 and 16.6 ± 10.3 mg N l⁻¹ respectively (- ~40% in OF) (Table 7-5 a). Overall, N leaching in OF and CF systems are 15.3 ± 9.7 and 27.0 ± 24.7 kg N ha⁻¹ with variations in relation to different climatic years in different regions. In final, leaching in OF is significantly 43% lower than in CF systems (*P*-value = 0.09) (Table 7-5 b).

Table 7-5. Synthesis of three studies on sub-root concentrations (a) N leaching (b) means for OF and CF systems (weighted by the regions) in the Seine basin. Standard deviation (SD) stands for the difference between farms from the different regions studied.

	(a) Mean sub-root concentrations (mg N l⁻¹)			
	<i>One mixed farm</i> ¹	<i>7 systems</i> ²	<i>14 systems</i> ³	Mean (SD)
OF mean (SD)	7.8	14.1 (8.9)	8.7 (2.8)	10.1 (3.4)
CF mean (SD)	10.8	27.4 (13.6)	12.6 (4.6)	16.6 (10.3)
-(OF-CF)/CF (%)	-28%	-49%	-31%	-39%

	(b) Leaching (kg N ha⁻¹ an⁻¹)			
	<i>One mixed farm</i> ¹	<i>7 systems</i> ²	<i>14 systems</i> ³	Mean (SD)
OF mean (SD)	14.6	28.7 (17.9)	8.7 (2.6)	15.3 (9.7)
CF mean (SD)	19.5	59.5 (22.6)	12.7 (3.6)	27.0(24.7)
-(OF-CF)/CF (%)	-25%	-52%	-31%	-43%

¹ chapter 3 ² chapter 6 ³ chapter 7

In conclusion, this network revealed a wide diversity OF and CF systems with different impacts on NO₃⁻ leaching in different regions. The N leached depends mainly on farmers' strategies, including the length of the rotations, crops variety, N inputs management but also on climate conditions. The interaction between farmers, researchers and stakeholders made possible owing to the network established during our study is full of promises for innovative practices which should improve water quality taking into account climate constraints and farmers strategies (MacMillan and Benton 2014).

Acknowledgement

The study is funded by a PhD scholarship and an equipment sponsorship from the DIM Astrea of the Ile-de-France Region and the Water Agency of the Seine-Normandy Basin. The PIREN-Seine and the ANR Escapade programs are also acknowledged for providing funding for the analysis. Météo-France and Irstea stations are gratefully acknowledged for providing weather data. Many thanks are extended to the farmers who allowed us to work on their farms and helped us when possible. We also sincerely thank Afonso G., Ansart P., Azougui A. and Mercier B. for their kind laboratory and/or field technical assistance.

Conclusion générale et perspectives

L'**objectif principal** de cette thèse était de déterminer la dynamique de l'azote au sein des exploitations de grandes cultures dans le bassin de la Seine, et de comparer les performances environnementales, vis-à-vis de la lixiviation et des émissions de GES, des systèmes biologiques et conventionnels. Aucune étude sur les pertes d'azote dans les systèmes biologiques très spécifiques du bassin de la Seine n'avait encore été publiée, ce qui limitait la crédibilité des programmes de conversion à l'AB mis en place dans le cadre de la protection de la qualité des captages d'eau potable. Pour aborder cette question, nous avons dû d'emblée faire certains **choix méthodologiques**, qui méritent a posteriori d'être discutés. Des **conclusions robustes** sur les performances comparées des deux systèmes de culture peuvent être tirées de nos travaux: nous les résumerons succinctement. Enfin des **perspectives** prometteuses s'ouvrent à l'issue de nos trois ans de recherche à la fois pour mieux comprendre les mécanismes qui régissent, à diverses échelles, les processus de pertes environnementales d'azote dans les agrosystèmes, et pour mieux interagir avec les agriculteurs en vue d'améliorer leurs pratiques.

Les choix méthodologiques et leurs limites

Pour l'essentiel, nos travaux ont porté sur la mesure des flux dans des exploitations commerciales du bassin de la Seine. Pour ce faire, il a fallu contacter des agriculteurs afin qu'ils nous « accueillent » dans leurs champs et nous donnent accès à leurs itinéraires techniques (pratiques, rendements et analyses).

L'échantillonnage des fermes que nous avons suivies n'est donc pas exempt d'un biais lié au nécessaire volontariat des exploitants, qui peut impliquer que des agriculteurs déjà sensibilisés aux bonnes pratiques agricoles (très bas intrants, cultures intermédiaires systématiques) aient répondu positivement à nos sollicitations.

D'autre part, la nature participative du réseau ABAC que nous avons constitué, confère un caractère de recherche-action à notre travail: les agriculteurs, en échange permanent avec les différents partenaires du réseau (scientifique, gestionnaires des captages, chambres d'agriculture, fédération biologique), peuvent faire évoluer leurs pratiques selon les résultats obtenus sur leurs exploitations, que nous leur communiquons systématiquement. En conséquence, si lors de ces premières années, le réseau ABAC a constitué un observatoire des pratiques réelles (engrais vert, application d'engrais organiques facilement dégradable à l'automne), ces dernières tendent à évoluer au cours des années vers une meilleure efficacité de l'azote (intercultures comportant plusieurs variétés, mélange d'engrais organique et de compost). Ces deux facteurs pourraient avoir conduit à une sous-estimation des pertes environnementales réelles, en système conventionnel comme en système biologique.

Concernant le panel des exploitations étudiées, nous avons pu suivre uniquement des rotations biologiques longues « avec luzerne », car elles possédaient une profondeur de sol suffisante pour permettre l'implantation de bougies poreuses en dessous de la zone racinaire hivernale. Dans les sols semi profonds et superficiels, comme en Essonne (91), les exploitations biologiques sont souvent caractérisées par des rotations plus courtes, que nous

n'avons donc pas pu caractériser, mais pour lesquelles les fuites d'azote pourraient être plus importantes.

L'expérimentation scientifique au sein d'exploitations commerciales, nous a obligés à utiliser des dispositifs légers qui n'entravent pas les travaux agricoles. Notre choix s'est donc porté, pour la mesure de la lixiviation, sur l'usage de bougies poreuses verticales, souvent réputées donner lieu à des écoulements verticaux préférentiels. Nous avons pu valider nos mesures grâce à la mesure parallèle de reliquats d'azote minéral à l'entrée de l'hiver, et montrer que ces fuites verticales étaient négligeables.

Le passage de la mesure des concentrations dans l'eau sous-racinaire à l'estimation des flux lixiviés nécessite de déterminer la lame d'eau infiltrée. Nous avons pour cela développé un modèle réservoir à une couche, qui pourrait être affiné. Il ne prend pas en compte en effet l'action de la végétation hivernale ni les caractéristiques propres à chaque parcelle. La comparaison de nos estimations avec la mesure directe du débit d'un collecteur de drains dans une exploitation bien instrumentée (Chantemerle) a cependant permis de montrer la cohérence des résultats de notre modèle.

Deux types de chambres d'accumulation de GES ont été utilisés au cours de cette thèse. Les mesures réalisées en chambres manuelles ont les mêmes tendances de flux que ceux mesurés en chambres automatiques, cependant, elles ne prennent pas en considération les faibles variations journalières. Cette différence de méthodes peut conduire à sous- ou surestimer les émissions dues à l'apport des fertilisants et donc à sous- ou surestimer le cumul des émissions à l'année.

Les acquis

Sur une exploitation mixte (AB/AC) instrumentée et suivie sur trois ans, nous avons pu montrer, dans des conditions pédoclimatiques identiques, que les moyennes de pertes azotées par lixiviation et par émission de N_2O intégrées sur l'ensemble du cycle de rotation culturale étaient inférieures en agriculture biologique par rapport à ce qu'elles sont en agriculture conventionnelle. Cette conclusion, particulièrement nette lorsque les résultats sont exprimés par rapport à la surface agricole concernée (-25% et -29% pour la lixiviation et les émissions de N_2O respectivement), reste vraie en termes d'émissions rapportées à la production de protéines des deux systèmes (-11% et -15% respectivement).

Les pertes d'azote inférieures en AB sont principalement dues à de plus faibles apports d'engrais exogènes en AB (60 kg N ha⁻¹) qu'en AC (160–200 kg N ha⁻¹), ainsi qu'à une rotation plus longue laissant une large place aux légumineuses. Toutefois des arrières-effets peuvent donner lieu à une lixiviation importante après l'apport d'engrais organiques (fumier, compost) et l'incorporation régulière de légumineuses (luzerne, féverole, lentille).

Ces conclusions tirées de l'examen d'une exploitation mixte sont confortées par celles issues de deux ans de suivi d'une quinzaine d'exploitations du réseau ABAC, couvrant une plus large diversité de conditions pédoclimatiques. Les pertes d'azote par lixiviation des rotations AB sont toutes inférieures ou égales aux pertes issues des exploitations conventionnelles dans

les mêmes conditions pédoclimatiques. Les concentrations sous-racinaires les plus faibles ($< 5 \text{ mg N l}^{-1}$) sont mesurées sous luzerne, ou sous une culture intermédiaire bien implantée. Par contre sous les blés semés en automne après retournement d'une légumineuse, des concentrations beaucoup plus élevées ($> 15 \text{ mg N l}^{-1}$) sont couramment observées.

Si la majeure partie de notre travail s'est focalisé sur l'échelle des exploitations et du système de culture, nos résultats, associés à de précédents travaux, ont aussi permis d'aborder la question de la cascade de l'azote à l'échelle d'un petit bassin versant, tel que celui de l'Orgeval. Nous y avons testé, sur la base de nos résultats, deux scénarii (curatif et préventif) de réduction de la contamination nitrique de l'hydrosystème. Le premier scénario, curatif, consiste en l'implantation d'étangs, sur 1% de la surface du bassin, de manière à éliminer naturellement une partie de la pollution nitrique agricole par dénitrification dans les sédiments. Une réduction de 34 à 47% (selon les années hydrologiques) des flux nitriques transférés vers l'aval du réseau hydrographique peut être obtenue par ce dispositif, mais il engendre un accroissement significatif des émissions de N_2O dans l'atmosphère ($+ 0.6 \text{ kg N ha}^{-1} \text{ an}^{-1}$). En revanche, un scénario « préventif », reposant sur une conversion à l'AB de toutes les surfaces agricoles, permettrait une réduction de 25 à 68% (selon les années) de la contamination nitrique, avec une baisse des émissions de N_2O .

Enfin, à l'échelle des processus, nous avons établi la relation à la température des processus de nitrification et de dénitrification dans les sols et de leurs émissions de N_2O associées. Nous avons ainsi pu montrer que la proportion de N_2O émise lors du processus de dénitrification s'accroît rapidement lorsque la température augmente, ce qui suggère un feed-back positif d'un réchauffement (période de canicule, changement climatique, ...).

Les perspectives

Les nombreuses mesures expérimentales acquises durant ce travail forment un ensemble de données d'une grande richesse dont l'exploitation est loin d'avoir pu être menée de manière exhaustive.

Rapidement, elles pourront permettre de tester un certain nombre d'indicateurs de pertes environnementales d'azote. Ainsi, la méthode du bilan d'azote des sols, intégrée à l'échelle de la rotation pour tenir compte des effets des précédents culturaux, permet le calcul d'un surplus (différence entre le total des apports d'azote au sol et l'exportation d'azote par la récolte) qui pourrait constituer un bon indicateur des pertes potentielles d'azote par lixiviation. Cette méthode a déjà été utilisée à plusieurs échelles (exploitation, bassin versant, pays) au sein de notre laboratoire et sera également appliquée aux exploitations du réseau ABAC. La comparaison de ces surplus avec la mesure directe de la lixiviation devrait permettre d'estimer la fraction du surplus effectivement lixiviée dans différents contextes pédoclimatiques, tant en agriculture biologique que conventionnelle. Les performances agro-environnementales des exploitations du réseau ABAC seront aussi mises en perspective avec un échantillon plus large de systèmes biologiques en grandes cultures et polyculture-élevage étudié en terme de surplus uniquement, dans le bassin de la Seine (54 exploitations et 70 rotations respectivement, Thèse J. Anglade prévue au printemps 2015).

Nos données devraient également permettre de tester divers modèles de calcul des processus de pertes azotées.

Au cours de cette thèse, des mesures de reliquats azotés en automne (Oct-Nov-Déc) et au printemps (Mars-Avril-Mai) ont été réalisés sur l'ensemble des parcelles étudiées. Les reliquats azotés pourraient être un bon outil scientifique de substitution aux bougies poreuses sur le long terme afin de limiter la lixiviation hivernale dans les parcelles agricoles. Toutefois leur interprétation est parfois complexe et nécessite l'utilisation d'un modèle qui permet de prendre en compte les différents processus de minéralisation et de réorganisation qui ont lieu au cours de la période hivernale, comme le modèle LIXIM (dérivé du modèle agronomique STICS) mis au point par l'unité de recherche (UR) AgroImpact (INRA, Laon).

Le modèle STICS est en effet un outil puissant pour simuler l'infiltration de l'eau dans les sols et le prélèvement par les plantes à l'échelle de la parcelle. Les jeux de données que nous avons établis pourraient servir à valider les prévisions de ce modèle, à mettre en évidence les principaux facteurs de pertes et à tester l'effet de changements de pratiques.

Les émissions de N_2O que nous avons mesurées en chambres automatiques l'ont été dans le cadre du réseau Effemair- N_2O , financé par l'ADEME, et dont l'UR AgroImpact assure la coordination. Ce réseau comprend trois autres sites (Estrées-Mons, Auzeville, Versailles) et une vingtaine de parcelles suivant différents itinéraires techniques, tels que des associations de cultures, des sols sous couvert permanent, des systèmes très autonomes en intrants, la culture de miscanthus... De plus, la caractérisation microbiologique des communautés bactériennes du sol est réalisée en parallèle des mesures en chambres automatiques par l'UMR Agroécologie (INRA, Dijon), ainsi que l'évaluation des potentiels de production de N_2O (UMR METIS et UMR Agroécologie). L'ensemble de ces mesures devra permettre de caractériser les facteurs (biologiques et abiotiques) responsables des émissions de N_2O , et là aussi de tester différents modèles (DNDC, DayCent outre STICS), et de proposer des stratégies performantes pour limiter l'empreinte environnementale des systèmes agricoles.

Les mesures de lixiviation du nitrate et d'émissions de N_2O réalisées sur le site de l'Orgeval s'intègrent aussi dans le cadre du projet ANR Escapade, dont notre équipe de recherche fait partie. L'objectif est de mieux cerner les processus liés à la cascade de l'azote à l'échelle fine du paysage (prairie, cultures sans labour, zones ripariennes et forêt), pour une meilleure gestion de l'azote réactif dans les territoires. Cette problématique est particulièrement importante dans les aires d'alimentation de captages d'eau potable (AAC). Certaines des exploitations du réseau ABAC sont localisées sur des AAC et la qualité de l'eau des aquifères y est suivie. Une caractérisation hydrogéologique plus détaillée serait nécessaire pour appréhender l'évolution des contaminations nitriques sur le long terme. Pour ce faire, plusieurs outils scientifiques ont déjà été mis en place sur le bassin de l'Orgeval, tels que des profils géophysiques, des cartes piézométriques et la datation des eaux via les gaz rares.

L'ensemble de ces résultats pourront être valorisés avec la collaboration de différentes équipes de recherche (Mines ParisTech, INRA, IRSTEA, Université Rennes, mais aussi avec l'équipe géophysique de notre laboratoire METIS).

Comme déjà mentionné, le réseau ABAC constitue un lieu d'échange particulièrement riche entre agriculteurs, chercheurs et gestionnaires des ressources en eau, en partie grâce au

comité de pilotage qui s'est déjà réuni trois fois. Les mesures réalisées dans les exploitations agricoles (azote, carbone, granulométrie, teneurs en eau, etc.) sont communiquées annuellement et individuellement aux exploitants sous forme d'un rapport personnalisé, et de manière anonyme lors des comités de pilotage scientifique et réunions informelles ou non, ainsi que lors des présentations à des colloques. Le réseau est maintenu dans son état actuel en cette période de drainage 2014-2015, et devrait l'être encore en 2015-2016 (financement Eau-de-Paris/AESN acquis), afin de mieux appréhender les variations climatiques et les arrières-effets des cultures. De plus, des stations météorologiques ont été installées dans chaque pôle afin d'améliorer les estimations des lame drainantes.

Des tours de plaine avec des groupes d'agriculteurs ainsi que des ateliers participatifs sont prochainement prévus. Ce réseau constitue donc un lieu privilégié pour mettre en place, de manière consensuelle, des mesures visant à réduire les pertes d'azote dans l'environnement, protégeant ainsi la qualité de la ressource en eau. Ces mesures sont tout à fait dans la ligne de celles préconisées à divers niveaux (agriculture, services agricoles, réglementation et recherche) lors d'un séminaire tenu à Lisbonne en juillet 2014 (18th Nitrogen Symposium) (**Annexe 4**).

Ainsi le réseau ABAC pourra contribuer de façon efficace, et si possible dans la durée, à la réflexion menée sur la gouvernance territoriale nécessaire à mettre en œuvre les leviers de la transition vers une agriculture plus durable.

Bibliographie

- Addiscott TM, Whitmore AP, Powlson DS (1991) Farming, fertilizers and the nitrate problem. 170 pp. CAB International
- AESN (2013) Etat des lieux du bassin de la Seine et des cours d'eau côtiers normands, décembre 2013. Adoptée par le comité de bassin délibération n° CB 13-08.
- Aguilera E, Lassaletta L, Gattinger A, Gimeno BS (2013) Managing soil carbon for climate change mitigation and adaptation in Mediterranean cropping systems: A meta-analysis. *Agric Ecosyst Environ* 168:25–36. doi: 10.1016/j.agee.2013.02.003
- Agreste (2013) Recensement agricole 2010 : productions végétales.
- Allard G, David C, Henning JC (2000) L'agriculture biologique face à son développement: les enjeux futurs. Ed Quae 394 pp.
- Allen RG (2000) Using the FAO-56 dual crop coefficient method over an irrigated region as part of an evapotranspiration intercomparison study. *J Hydrol* 229:27–41. doi: 10.1016/S0022-1694(99)00194-8
- Altieri MA (2002) Agroecology: the science of natural resource management for poor farmers in marginal environments. *Agric Ecosyst Environ* 93:1–24. doi: 10.1016/S0167-8809(02)00085-3
- Anglade J, Billen G, Garnier J (subm) New relationships for estimating N₂ fixation in legumes: incidence for N balance of low-input cropping systems in Europe. *Ecol. Applic.*
- Angle JS, Gross CM, Hill RL, McIntosh MS (1993) Soil Nitrate Concentrations under Corn as Affected by Tillage, Manure, and Fertilizer Applications. *J Environ Qual* 22:141. doi: 10.2134/jeq1993.00472425002200010018x
- Arlot M-P (1999) Thèse : Nitrates dans les eaux : drainage acteur, drainage témoin ? Les enseignements d'une approche hydrologique et hydraulique sur (1) l'influence du drainage agricole sur la contamination diffuse des eaux par les nitrates et (2) sur les possibilités d'utiliser les parcelles drainées pour l'analyse de la dite contamination, 586 pp
- Arlot MP and Zimmer D (1990) Drainage agricole et lessivage des nitrates. In : *Nitrates Agriculture Eau*, Ed INRA 263-268.
- Armitage PD, Hawczak A, Blackburn JH (2012) Tyre track pools and puddles – Anthropogenic contributors to aquatic biodiversity. *Limnol - Ecol Manag Inland Waters* 42:254–263. doi: 10.1016/j.limno.2012.07.002
- Arnold JG, Srinivasan R, Muttiah RS, Williams JR (1998) Large area hydrologic modeling and assessment part I: Model development1. Wiley Online Library

- Aronsson H, Torstensson G, Bergstrom L (2007) Leaching and crop uptake of N, P and K from organic and conventional cropping systems on a clay soil. *Soil Use Manag* 23:71–81. doi: 10.1111/j.1475-2743.2006.00067.x
- Askegaard M, Olesen JE, Rasmussen IA, Kristensen K (2011) Nitrate leaching from organic arable crop rotations is mostly determined by autumn field management. *Agric Ecosyst Environ* 142:149–160. doi: 10.1016/j.agee.2011.04.014
- Augeard B, Kao C, Chaumont C, Vauclin M (2005) Mechanisms of surface runoff genesis on a subsurface drained soil affected by surface crusting: A field investigation. *Phys Chem Earth Parts ABC* 30:598–610. doi: 10.1016/j.pce.2005.07.014
- Baggs EM, Watson CA, Rees RM (2000) The fate of nitrogen from incorporated cover crop and green manure residues. *Nutr Cycl Agroecosystems* 56:153–163. doi: 10.1023/A:1009825606341
- Ballif J. L., Muller J. C. (1990) Les bougies poreuses et les lysimètres – Perspectives agricoles n° 144-145, février, mars
- Balser TC, McMahon KD, Bart D, et al. (2006) Bridging the gap between micro - and macro-scale perspectives on the role of microbial communities in global change ecology. *Plant Soil* 289:59–70. doi: 10.1007/s11104-006-9104-5
- Bàrberi P (2002) Weed management in organic agriculture: are we addressing the right issues? *Weed Res* 42:177–193. doi: 10.1046/j.1365-3180.2002.00277.x
- Barles S (2010) Society, Energy and Materials: What are the Contributions of Industrial Ecology, Territorial Ecology and Urban Metabolism to Sustainable Urban Development Issues? *J Environ Plan Manage* 53, 439-455.
- Barles S (2013) Socio-ecological trajectories: the urban dimension. Paris, 18th-20th centuries. Actes du colloque de l'European Society for Ecological Economics and Institutional Dynamics, Lille, 17-21 juin 2013. 17pp.
- Barnard R, Leadley PW, Hungate BA (2005) Global change, nitrification, and denitrification: A review. *Glob Biogeochem Cycles* 19:n/a–n/a. doi: 10.1029/2004GB002282
- Barracough PB (1989) Root growth, macro-nutrient uptake dynamics and soil fertility requirements of a high-yielding winter oilseed rape crop. *Plant Soil* 119:59–70. doi: 10.1007/BF02370269
- Basso B, Ritchie JT (2005) Impact of compost, manure and inorganic fertilizer on nitrate leaching and yield for a 6-year maize–alfalfa rotation in Michigan. *Agric Ecosyst Environ* 108:329–341. doi: 10.1016/j.agee.2005.01.011
- Bateman EJ, Baggs EM (2005) Contributions of nitrification and denitrification to N₂O emissions from soils at different water-filled pore space. *Biol Fertil Soil* 41:379–388. doi: 10.1007/s00374-005-0858-3
- Beaudoin N, Denys D, Muller J, et al. (1992) Influence d'une culture de luzerne sur le lessivage du nitrate dans les sols de Champagne crayeuse. *Fourrages*
- Beaudoin N, Launay M, Sauboua E, et al. (2008) Evaluation of the soil crop model STICS over 8 years against the “on farm” database of Bruyères catchment. *Eur J Agron* 29:46–57. doi: 10.1016/j.eja.2008.03.001
- Beaudoin N, Saad JK, Van Laethem C, et al. (2005) Nitrate leaching in intensive agriculture in Northern France: Effect of farming practices, soils and crop rotations. *Agric Ecosyst Environ* 111:292–310.
- Beek CL van, Brouwer L, Oenema O (2003) The use of farmgate balances and soil surface balances as estimator for nitrogen leaching to surface water. *Nutr Cycl Agroecosystems* 67:233–244. doi: 10.1023/B:FRES.0000003619.50198.55

- Benoit M, Garnier J, Anglade J, Billen G (2014) Nitrate leaching from organic and conventional arable crop farms in the Seine Basin (France). *Nutr Cycl Agroecosystems*. doi: 10.1007/s10705-014-9650-9
- Berg M, Haas G, Kopke U (1999) Konventioneller, integrierter oder organischer Landbau: Fallbeispiel Wasserschutzgebiet am Niederrhein. Im Tagungsband
- Bergstrom LF, Kirchmann H (1999) Leaching of Total Nitrogen from Nitrogen-15-Labeled Poultry Manure and Inorganic Nitrogen Fertilizer. *J Environ Qual* 28:1283. doi: 10.2134/jeq1999.00472425002800040032x
- Bergstrom L, Johansson R (1991) Leaching of Nitrate from Monolith Lysimeters of Different Types of Agricultural Soils. *J Environ Qual* 20:801. doi: 10.2134/jeq1991.00472425002000040015x
- Bessou C, Mary B, Léonard J, et al. (2010) Modelling soil compaction impacts on nitrous oxide emissions in arable fields. *Eur J Soil Sci* 61:348–363. doi: 10.1111/j.1365-2389.2010.01243.x
- Bertuzzi P, Justes E, Le Bas C, Mary B, Souchère V (2012) Effets des cultures intermédiaires sur l'érosion, les propriétés physiques du sol et le bilan carbone (Chapitre 5). Réduire les fuites de nitrate au moyen de cultures intermédiaires.
- Billen G, Barles S, Garnier J, et al. (2009a) The food-print of Paris: long-term reconstruction of the nitrogen flows imported into the city from its rural hinterland. *Reg Environ Change* 9:13–24. doi: 10.1007/s10113-008-0051-y
- Billen G, Garnier J (1999) Nitrogen transfers through the Seine drainage network: a budget based on the application of the Riverstrahler' model. *Hydrobiologia* 410:139–150.
- Billen G, Garnier J, Benoit M, Anglade J (2013) The nitrogen cascade in arable crop areas of the North of France. *Cah Agric* 22:272–281.
- Billen G, Garnier J, Hanset P (1994) Modelling phytoplankton development in whole drainage networks: the RIVERSTRAHLER Model applied to the Seine river system. In: Descy J-P, Reynolds CS, Padisák J (eds) *Phytoplankton Turbid Environ. Rivers Shallow Lakes*. Springer Netherlands, 119–137 pp.
- Billen G, Garnier J, Lassaletta L (2013) Modelling the nitrogen cascade from watershed soils to the sea: from regional to global scales. *Phil. Trans. R. Soc. B* 368, 20130123. <http://dx.doi.org/10.1098/rstb.2013.0123>.
- Billen G, Garnier J, Mouchel JM, Silvestre M (2007a) The Seine system: Introduction to a multidisciplinary approach of the functioning of a regional river system. *Sci Total Environ* 375:1–12.
- Billen G, Garnier J, Nemery J, et al. (2007b) A long-term view of nutrient transfers through the Seine river continuum. *Sci Total Environ* 375:80–97.
- Billen G, Thieu V, Garnier J, Silvestre M (2009b) Modelling the N cascade in regional watersheds: The case study of the Seine, Somme and Scheldt rivers. *Agric Ecosyst Environ* 133:234–246.
- Billy C (2008) Thèse : Transfert et rétention d'azote à l'échelle d'un bassin versant agricole artificiellement drainé. 198 pp.
- Billy C, Billen G, Sebilo M, et al. (2010) Nitrogen isotopic composition of leached nitrate and soil organic matter as an indicator of denitrification in a sloping drained agricultural plot and adjacent uncultivated riparian buffer strips. *Soil Biol Biochem* 42:108–117. doi: 10.1016/j.soilbio.2009.09.026

- Billy C, Birgand F, Sebilo M, et al. (2011) Nitrate dynamics in artificially drained nested watersheds. *Phys Chem Earth Parts ABC* 36:506–514. doi: 10.1016/j.pce.2008.09.007
- Bioteau T, Bordenave P, Laurent F, et al. (2002) Evaluation des risques de pollution diffuse par l'azote d'origine agricole à l'échelle de bassins versants: intérêts d'une approche par modélisation avec SWAT®. *ET Ingénieries* - p. 3- 12.
- Borges AV, Vanderborght J-P, Schiettecatte L-S, et al. (2004) Variability of the gas transfer velocity of CO₂ in a macrotidal estuary (the Scheldt). *Estuaries* 27:593–603. doi: 10.1007/BF02907647
- Boulet (1975) Toposéquences de sol tropicaux en Haute-Volta 2équilibres dynamiques et bioclimats. *Cah ORSTOM (IRD) série Pédologique*, vol XIII, n°1
- Bouwman (1994) Method to estimate direct nitrous oxide emissions from agricultural soils. *RIVM. P.* 1- 28
- Bouwman AF (1996) Direct emission of nitrous oxide from agricultural soils. *Nutr Cycl Agroecosystems* 46:53–70. doi: 10.1007/BF00210224
- Bouwman AF, Boumans LJM, Batjes NH (2002) Emissions of N₂O and NO from fertilized fields: Summary of available measurement data. *Glob Biogeochem Cycles* 16:1058. doi: 10.1029/2001GB001811
- Bowman MS, Clune TS, Sutton BG (2002) A modified ceramic sampler and lysimeter design for improved monitoring of soil leachates. *Water Res* 36:799–804.
- Bramley R, White R (1989) The effect of pH, liming, moisture and temperature on the activity of nitrifiers in a soil under pasture. *Soil Res* 27:711–724.
- Briggs LJ, McCall AG (1904) An artifical root for inducing capillary movement of soil moisture. *Science* 20:566–569. doi: 10.1126/science.20.513.566
- Brion N, Billen G (1998) Une réévaluation de la methode d'incorporation de H¹⁴C₀₃- pour mesurer la nitrification autotrophe et son application pour estimer des biomasses de bactéries nitrifiantes. *Rev Sci Eau* 11:283. doi: 10.7202/705308ar
- Brisson N, Gary C, Justes E, et al. (2003) An overview of the crop model stics. *Eur J Agron* 18:309–332. doi: 10.1016/S1161-0301(02)00110-7
- Bronner C.E, Bartlett AM, Whiteway SL, Lambert DC, Bennett SJ, Rabideau AJ (2013) An assessment of U.S. stream compensatory mitigation policy: necessary changes to protect ecosystem functions and services. *J Am Water Res Ass*, 49: 449-462.
- Bruand A., Duval O., Gaillard H., Darthout R., Jamagne M. (1996) Variabilités des propriétés de rétention en eau des sols : importance de la densité apparente. *Etude Gest Sols* 3 :1, 27-40
- Bruand A, Duval O, Cousin I (2004) Estimation des propriétés de rétention en eau des sols à partir de la base de données SOLHYDRO : Une première proposition combinant le type d'horizon, sa texture et sa densité apparente. *Etude Gest Sols* 11:3, 323–334.
- Burgin AJ, Hamilton SK (2007) Have we overemphasized the role of denitrification in aquatic ecosystems? A review of nitrate removal pathways. *Front Ecol Environ* 5:89–96. doi: 10.1890/1540-9295(2007)5[89:HWOTRO]2.0.CO;2
- Burgin AJ, Lazar JG, Groffman PM, et al. (2013) Balancing nitrogen retention ecosystem services and greenhouse gas disservices at the landscape scale. *Ecol Eng* 56:26–35. doi: 10.1016/j.ecoleng.2012.05.003
- Burns IG (1976) Equations to predict the leaching of nitrate uniformly incorporated to a known depth or uniformly distributed throughout a soil profile. *J Agric Sci* 86:305–313. doi: 10.1017/S0021859600054769

- Burt TP, Pinay G, Matheson FE, et al. (2002) Water table fluctuations in the riparian zone: comparative results from a pan-European experiment. *J Hydrol* 265:129–148. doi: 10.1016/S0022-1694(02)00102-6
- Camargo JA, Alonso Á (2006) Ecological and toxicological effects of inorganic nitrogen pollution in aquatic ecosystems: A global assessment. *Environ Int* 32:831–849. doi: 10.1016/j.envint.2006.05.002
- Carter MR and Gregorich EG (2007) *Soil Sampling and Methods of Analysis*, Ed CRC Press, 1264 pp.
- Catt JA, Howse KR, Christian DG, et al. (1998) Strategies to decrease nitrate leaching in the Brimstone Farm Experiment, Oxfordshire, UK, 1988–1993: the effects of winter cover crops and unfertilised grass leys. *Plant Soil* 203:57–69. doi: 10.1023/A:1004389426718
- Chalk PM, Smith CJ (1983) Chemodenitrification. In: Frenay JR, Simpson JR (eds) *Gaseous Loss Nitrogen Plant-Soil Syst.* Springer Netherlands, 65–89.
- Chapot JY (1990) Estimation sur lysimètres de l'incidence de l'introduction d'un engrais vert (moutarde blanche) dans une rotation blé-maïs sur la lixiviation des nitrates. In : *Nitrates-Agriculture-Eau*, Ed INRA 411–416.
- Chen D, Li Y, Grace P, Mosier AR (2008) N₂O emissions from agricultural lands: a synthesis of simulation approaches. *Plant Soil* 309:169–189. doi: 10.1007/s11104-008-9634-0
- Cheneby D, Nicolardot B, Godden B, Penninckx M (1994) Mineralization of Composted ¹⁵N-Labelled Farmyard Manure During Soil Incubations. *Biol Agric Hortic* 10:255–264. doi: 10.1080/01448765.1994.9754677
- Chester ET, Robson BJ (2013) Anthropogenic refuges for freshwater biodiversity: Their ecological characteristics and management. *Biol Conserv* 166:64–75. doi: 10.1016/j.biocon.2013.06.016
- Chirinda N, Carter MS, Albert KR, et al. (2010) Emissions of nitrous oxide from arable organic and conventional cropping systems on two soil types. *Agric Ecosyst Environ* 136:199–208. doi: 10.1016/j.agee.2009.11.012
- Cicerone RJ (1987) Changes in Stratospheric Ozone. *Science* 237:35–42. doi: 10.1126/science.237.4810.35
- Conrad R, Seiler W, Bunse G (1983) Factors influencing the loss of fertilizer nitrogen into the atmosphere as N₂O. *J Geophys Res Oceans* 88:6709–6718. doi: 10.1029/JC088iC11p06709
- Constantin J, Beaudoin N, Laurent F, et al. (2011) Cumulative effects of catch crops on nitrogen uptake, leaching and net mineralization. *Plant Soil* 341:137–154. doi: 10.1007/s11104-010-0630-9
- Constantin J, Mary B, Laurent F, et al. (2010) Effects of catch crops, no till and reduced nitrogen fertilization on nitrogen leaching and balance in three long-term experiments. *Agric Ecosyst Environ* 135:268–278. doi: 10.1016/j.agee.2009.10.005
- Corre MD, van Kessel C, Pennock DJ (1996) Landscape and Seasonal Patterns of Nitrous Oxide Emissions in a Semiarid Region. *Soil Sci Soc Am J* 60:1806. doi: 10.2136/sssaj1996.03615995006000060028x
- Creasey CL, Dreiss SJ (1988) Porous cup samplers; cleaning procedures and potential samples bias from trace element contamination. *Soil Sci.* Vol 145:2
- Cugier P, Billen G, Guillaud JF, et al. (2005) Modelling the eutrophication of the Seine Bight (France) under historical, present and future riverine nutrient loading. *J Hydrol* 304:381–396. doi: 10.1016/j.jhydrol.2004.07.049

- Čuhel J, Šimek M, Laughlin RJ, et al. (2010) Insights into the Effect of Soil pH on N₂O and N₂ Emissions and Denitrifier Community Size and Activity. *Appl Environ Microbiol* 76:1870–1878. doi: 10.1128/AEM.02484-09
- Dahl TE (2011) Status and trends of wetlands in the conterminous United States 2004 to 2009. US Department of the Interior, US Fish and Wildlife Service, Fisheries and Habitat Conservation
- Dalias P, Anderson JM, Bottner P, Coûteaux M-M (2002) Temperature responses of net nitrogen mineralization and nitrification in conifer forest soils incubated under standard laboratory conditions. *Soil Biol Biochem* 34:691–701. doi: 10.1016/S0038-0717(01)00234-6
- Dalsgaard T, De Brabandere L, Hall POJ (2013) Denitrification in the water column of the central Baltic Sea. *Geochim Cosmochim Acta* 106:247–260. doi: 10.1016/j.gca.2012.12.038
- Dalsgaard T, Thamdrup B, Canfield DE (2005) Anaerobic ammonium oxidation (anammox) in the marine environment. *Res Microbiol* 156:457–464. doi: 10.1016/j.resmic.2005.01.011
- David MB, Wall LG, Royer TV, Tank JL (2006) Denitrification and the nitrogen budget of a reservoir in an agricultural landscape. *Ecol Appl* 16:2177–2190. doi: 10.1890/1051-0761(2006)016[2177:DATNBO]2.0.CO;2
- Davidson EA, David MB, Galloway JN, Goodale CL, Haeuber R, Harrison JA, Howarth RW et al. (2012) Excess Nitrogen in the U.S. Environment: Trends, Risks, and Solutions. *Issues in Ecology*, ESA15: 1-15.
- Davis AS, Hill JD, Chase CA, et al. (2012) Increasing Cropping System Diversity Balances Productivity, Profitability and Environmental Health. *PLoS ONE* 7:e47149. doi: 10.1371/journal.pone.0047149
- Delin S and Engstrom L (2009) Timing of organic fertiliser application to synchronise nitrogen supply with crop demand. *Acta Agric Scand Sect B — Soil Plant Sci* 60:78–88. doi: 10.1080/09064710802631943
- Denys D (1990) Conséquences de l'organisation de l'azote minéral d'un engrais sur la disponibilité pour la plante et sur la lixiviation. In: *Nitrates-Agriculture-Eau*, Ed INRA 189-194.
- De Ponti T, Rijk B, van Ittersum MK (2012) The crop yield gap between organic and conventional agriculture. *Agric Syst* 108:1–9. doi: 10.1016/j.agry.2011.12.004
- Destain J-P, Fonder N, Xanthoulis D, Reuter V (2010) L'isotope stable ¹⁵N et le lysimètre, des outils complémentaires pour l'étude de la lixiviation de l'azote dans les sols agricoles. *Biotechnol Agron Société Environ BASE* 14:91–96.
- De Vries W, Leip A, Reinds GJ, et al. (2011) Comparison of land nitrogen budgets for European agriculture by various modeling approaches. *Environ Pollut* 159:3254–3268. doi: 10.1016/j.envpol.2011.03.038
- Dhondt K, Boeckx P, Hofman G, Cleemput OV (2004) Temporal and spatial patterns of denitrification enzyme activity and nitrous oxide fluxes in three adjacent vegetated riparian buffer zones. *Biol Fertil Soils* 40:243–251. doi: 10.1007/s00374-004-0773-z
- Diaz RJ, Rosenberg R (2008) Spreading Dead Zones and Consequences for Marine Ecosystems. *Science* 321:926–929. doi: 10.1126/science.1156401

- Dobbie KE, Smith KA (2001) The effects of temperature, water-filled pore space and land use on N₂O emissions from an imperfectly drained gleysol. *Eur J Soil Sci* 52:667–673. doi: 10.1046/j.1365-2389.2001.00395.x
- Drury CF, Findlay WI, Gaynor JD, McKenney DJ (1993) Influence of Tillage on Nitrate Loss in Surface Runoff and Tile Drainage. *Soil Sci Soc Am J* 57:797. doi: 10.2136/sssaj1993.03615995005700030028x
- Doré T, Dufumier M, Rivière-Wekstein G (2013) *Agriculture biologique : espoir ou chimère ?* collection Le choc des idées, 128 p
- Eckard RJ, Cullen BR (2011) Impacts of future climate scenarios on nitrous oxide emissions from pasture based dairy systems in south eastern Australia. *Anim Feed Sci Technol* 166-67:736–748. doi: 10.1016/j.anifeedsci.2011.04.052
- Eriksen J, Askegaard M, Kristensen K (1999) Nitrate leaching in an organic dairy/crop rotation as affected by organic manure type, livestock density and crop. *Soil Use Manag* 15:176–182.
- EU (2013) Overview of CAP Reform 2014-2020 - European Commission. Agricultural perspectives brief. N° 5, Decembre 2013. http://ec.europa.eu/agriculture/policy-perspectives/policy-briefs/05_en.pdf
- Evanylo G, Sherony C, Spargo J, et al. (2008) Soil and water environmental effects of fertilizer-, manure-, and compost-based fertility practices in an organic vegetable cropping system. *Agric Ecosyst Environ* 127:50–58. doi: 10.1016/j.agee.2008.02.014
- Feaga JB, Selker JS, Dick RP, Hemphill DD (2010) Long-Term Nitrate Leaching Under Vegetable Production with Cover Crops in the Pacific Northwest. *Soil Sci Soc Am J* 74:186. doi: 10.2136/sssaj2008.0178
- Firestone MK, Davidson EA (1989) Microbiological basis of NO and N₂O production and consumption in soil. *Sci* 7–21.
- Firestone MK, Firestone RB, Tiedje JM (1980) Nitrous Oxide from Soil Denitrification: Factors Controlling Its Biological Production. *Sci* 208:749–751. doi: 10.1126/science.208.4445.749
- Fischer EN, Whalen SC (2005) Rates and controls on denitrification in an agricultural soil fertilized with liquid lagoonal swine waste. *Nutr Cycl Agroecosystems* 71:271–287. doi: 10.1007/s10705-004-6379-x
- Fischer-Kowalski M, Rotmans J (2009) Conceptualizing, observing, and influencing socio-ecological transitions. *Ecol Soc* 14-3.
- Flessa H, Ruser R, Dorsch P, et al. (2002) Integrated evaluation of greenhouse gas emissions (CO₂, CH₄, N₂O) from two farming systems in southern Germany. *Agric Ecosyst Environ* 91:175–189.
- Focht DD, Verstraete W (1977) Biochemical ecology of nitrification and denitrification [Soils]. *Adv Microb Ecol* 1:135-214
- Fowler D, Coyle M, Flechard C, et al. (2001) Advances in micrometeorological methods for the measurement and interpretation of gas and particle nitrogen fluxes. *Plant Soil* 228:117–129. doi: 10.1023/A:1004871511282
- Francis GS, Haynes RJ, Sparling GP, et al. (1992) Nitrogen mineralization, nitrate leaching and crop growth following cultivation of a temporary leguminous pasture in autumn and winter. *Fertil Res* 33:59–70. doi: 10.1007/BF01058010

- Franzluebbers AJ (2002) Water infiltration and soil structure related to organic matter and its stratification with depth. *Soil Tillage Res* 66:197–205. doi: 10.1016/S0167-1987(02)00027-2
- Frolking SE, Mosier AR, Ojima DS, et al. (1998) Comparison of N₂O emissions from soils at three temperate agricultural sites: simulations of year-round measurements by four models. *Nutr Cycl Agroecosystems* 52:77–105. doi: 10.1023/A:1009780109748
- Gaines TP and Gaines ST (1994) Soil texture effect on nitrate leaching in soil percolates. *Commun Soil Sci Plant Anal* 25:2561–2570. doi: 10.1080/00103629409369207
- Galloway JN, Aber JD, Erisman JW, et al. (2003) The nitrogen cascade. *BioScience* 53:341–356.
- Garnier J and Billen G (1993) Ecological interactions in a shallow sand-pit lake (Créteil Lake, France). A modelling approach. In: *Nutrient dynamics and biological structure in shallow freshwater and brackish lakes*. *Hydrobiologia* 275/276, 97–114.
- Garnier J, Mounier E, Laverman A, Billen G (2010) Potential denitrification and nitrous oxide production in the sediments of the Seine River Drainage Network (France). *J Environ Qual* 39:449–459. doi: 10.2134/jeq2009.0299
- Garnier J, Billen G, Cébron A (2007) Modelling nitrogen transformations in the lower Seine river and estuary (France): impact of wastewater release on oxygenation and N₂O emission. *Hydrobiologia* 588:291–302.
- Garnier J, Billen G, Hannon E, et al. (2002) Modelling the Transfer and Retention of Nutrients in the Drainage Network of the Danube River. *Estuar Coast Shelf Sci* 54:285–308. doi: 10.1006/ecss.2000.0648
- Garnier J, Billen G, Sanchez N, Leporcq B (2000) Ecological functioning of the Marne reservoir (upper Seine basin, France). *Regul Rivers Res Manag* 16:51–71. doi: 10.1002/(SICI)1099-1646(200001/02)16:1<51::AID-RRR571>3.0.CO;2-I
- Garnier J, Billen G, Vilain G, et al. (2009) Nitrous oxide (N₂O) in the Seine river and basin: Observations and budgets. *Agric Ecosyst Environ* 133:223–233.
- Garnier P, Néel C, Mary B, Lafolie F (2001) Evaluation of a nitrogen transport and transformation model in a bare soil. *Eur J Soil Sci* 52:253–268. doi: 10.1046/j.1365-2389.2001.00374.x
- Garrido F, Hénault C, Gaillard H, et al. (2002) N₂O and NO emissions by agricultural soils with low hydraulic potentials. *Soil Biol Biochem* 34:559–575. doi: 10.1016/S0038-0717(01)00172-9
- Gaury (1992) Thèse : Systèmes de cultures et teneurs en nitrates des eaux souterraines, 229 pp.
- Grossman J et Udluft P (1991) The extraction of soil water by the suction-cup method : a review. *J Soil Sci*, 42:83–93
- Gelfand I, Yakir D (2008) Influence of nitrite accumulation in association with seasonal patterns and mineralization of soil nitrogen in a semi-arid pine forest. *Soil Biol Biochem* 40:415–424.
- Glavan M, Miličić V, Pintar M (2013) Finding options to improve catchment water quality—Lessons learned from historical land use situations in a Mediterranean catchment in Slovenia. *Ecol Model* 261–262:58–73. doi: 10.1016/j.ecolmodel.2013.04.004
- Glavan M, White SM, Holman IP (2012) Water quality targets and maintenance of valued landscape character – Experience in the Axe catchment, UK. *J Environ Manage* 103:142–153. doi: 10.1016/j.jenvman.2012.03.009

- Godde M and Conrad R (1999) Immediate and adaptational temperature effects on nitric oxide production and nitrous oxide release from nitrification and denitrification in two soils. *Biol Fertil Soils* 30:33–40.
- Grizzetti B, Bouraoui F, Aloe A (2012) Changes of nitrogen and phosphorus loads to European seas. *Glob Change Biol* 18:769–782. doi: 10.1111/j.1365-2486.2011.02576.x
- Grizzetti B, Bouraoui F, de Marsily G, Bidoglio G (2005) A statistical approach to estimate nitrogen sectorial contribution to total load. *Water Sci. Technol.* 51:83–90.
- Groffman PM, Tiedje JM (1991) Relationships between denitrification, CO₂ production and air-filled porosity in soils of different texture and drainage. *Soil Biol Biochem* 23:299–302. doi: 10.1016/0038-0717(91)90067-T
- Grover BL, Lamborn RE (1970) Preparation of Porous Ceramic Cups to be Used for Extraction of Soil Water Having Low Solute Concentrations1. *Soil Sci Soc Am J* 34:706. doi: 10.2136/sssaj1970.03615995003400040048x
- Gruca-Rokosz R, Tomaszek J.A (2007) The effect of abiotic factors on denitrification rates in sediment of Solina Reservoir, Poland. *Environ. Prot. Eng.* 33, 131–40.
- Grundmann GL, Renault P, Rosso L, Bardin R (1995) Differential Effects of Soil Water Content and Temperature on Nitrification and Aeration. *Soil Sci Soc Am J* 59:1342–1349. doi: 10.2136/sssaj1995.03615995005900050021x
- Gutser R, Ebertseder T, Weber A, et al. (2005) Short-term and residual availability of nitrogen after long-term application of organic fertilizers on arable land. *J Plant Nutr Soil Sci* 168:439–446. doi: 10.1002/jpln.200520510
- Guyomard H. (sous la direction de) (2013) Vers des agricultures à hautes performances. Volume 1. Analyse des performances de l'agriculture biologique. INRA, 368 pp.
- Haas G, Berg M, Kopke U (2002a) Nitrate leaching: comparing conventional, integrated and organic agricultural production systems. *Int Assoc Hydrol Sci Publ* 131–136.
- Haas G, Wetterich F (2000) Optimizing agri-environmental program to reduce negative environmental impact in the Allgaeu region using life cycle assessment. *BERICHTE UBER Landwirtschaft* 78:92–105.
- Hansen B, Alroe H, Kristensen E (2001) Approaches to assess the environmental impact of organic farming with particular regard to Denmark. *Agric Ecosyst Environ* 83:11–26. doi: 10.1016/S0167-8809(00)00257-7
- Hansen B, Kristensen ES, Grant R, et al. (2000) Nitrogen leaching from conventional versus organic farming systems—a systems modelling approach. *Eur J Agron* 13:65–82.
- Hansen EA, Harris AR (1975) Validity of Soil-Water Samples Collected with Porous Ceramic Cups1. *Soil Sci Soc Am J* 39:528. doi: 10.2136/sssaj1975.03615995003900030041x
- Haycock NE, Pinay G (1993) Groundwater Nitrate Dynamics in Grass and Poplar Vegetated Riparian Buffer Strips during the Winter. *J Environ Qual* 22:273. doi: 10.2134/jeq1993.00472425002200020007x
- Hefting M, Beltman B, Karssenberg D, et al. (2006) Water quality dynamics and hydrology in nitrate loaded riparian zones in the Netherlands. *Environ Pollut* 139:143–156. doi: 10.1016/j.envpol.2005.04.023
- Hénault C, Bizouard F, Laville P, et al. (2005) Predicting in situ soil N₂O emission using NOE algorithm and soil database. *Glob Change Biol* 11:115–127. doi: 10.1111/j.1365-2486.2004.00879.x

- Hénault C, Devis X, Lucas JL, Germon JC (1998) Influence of different agricultural practices (type of crop, form of N-fertilizer) on soil nitrous oxide emissions. *Biol Fertil Soils* 27:299–306. doi: 10.1007/s003740050437
- Hénault C, Devis X, Page S, et al. (1998) Nitrous oxide emissions under different soil and land management conditions. *Biol Fertil Soils* 26:199–207. doi: 10.1007/s003740050368
- Hénault C, Grossel A, Mary B, et al. (2012) Nitrous Oxide Emission by Agricultural Soils: A Review of Spatial and Temporal Variability for Mitigation. *Pedosphere* 22:426–433. doi: 10.1016/S1002-0160(12)60029-0
- Henine H, Nédélec Y, Augéard B, et al. (2010) Effect of Pipe Pressurization on the Discharge of a Tile Drainage System. *Vadose Zone J* 9:36. doi: 10.2136/vzj2008.0152
- Hennart S, Lambert R, Oger R, Stilmant D (2010) Sampling scheme to quantify nitrogen leaching risk in grazed grassland: which scheme for which accuracy? *Les Presses Agronomiques de Gembloux*, 39–46
- Hennart S, Lambert R, Oger R, Stilmant D (2009) Échantillonnage des prairies pâturées pour quantifier l'azote potentiellement lessivable: quel schéma pour quelle précision. *Biotechnol Agron Soc Env* 14:39–46.
- Henneron L, Bernard L, Hedde M, et al. (2014) Fourteen years of evidence for positive effects of conservation agriculture and organic farming on soil life. *Agron Sustain Dev* 1–13. doi: 10.1007/s13593-014-0215-8
- Hertel O, Skjøth CA, Reis S, et al. (2012) Governing processes for reactive nitrogen compounds in the European atmosphere. *Biogeosciences* 9:4921–4954. doi: 10.5194/bg-9-4921-2012
- Hoffmann M, Johnsson H (1999) A method for assessing generalised nitrogen leaching estimates for agricultural land. *Environ Model Assess* 4:35–44. doi: 10.1023/A:1019087511708
- Hooker KV, Coxon CE, Hackett R, et al. (2008) Evaluation of Cover Crop and Reduced Cultivation for Reducing Nitrate Leaching in Ireland. *J Environ Qual* 37:138. doi: 10.2134/jeq2006.0547
- Howard-Williams C (1985) Cycling and retention of nitrogen and phosphorus in wetlands: a theoretical and applied perspective. *Freshw Biol* 15:391–431.
- Howarth R, Chan F, Conley DJ, et al. (2011) Coupled biogeochemical cycles: eutrophication and hypoxia in temperate estuaries and coastal marine ecosystems. *Front Ecol Environ* 9:18–26. doi: 10.1890/100008
- Hutchinson GL, Livingston GP (1993) Use of Chamber Systems to Measure Trace Gas Fluxes. *ASA Spec. Publ.*
- IPCC (2006) Chapter 11: N₂O emissions from managed soils, and CO₂ emissions from lime and urea application. Volume 4 : Agriculture, Forestry and Other Land Use. Guidelines for National Greenhouse Gas Inventories. Intergovernmental Panel on Climate Change, IGES, Japan.
- Isermann K (1994) Agriculture's share in the emission of trace gases affecting the climate and some cause-oriented proposals for sufficiently reducing this share. *Environ Pollut* 83:95–111. doi: 10.1016/0269-7491(94)90027-2
- Jamagne M, Betremieux R, Begon JC, Mori A (1977) Quelques donnees sur la variabilite dans le milieu naturel de la reserve en eau des sols. *Bull Tech Inf Minist Agric Paris*

- James C, Fisher J, Russell V, et al. (2005) Nitrate availability and hydrophyte species richness in shallow lakes. *Freshw Biol* 50:1049–1063. doi: 10.1111/j.1365-2427.2005.01375.x
- Jemison JM, Jabro JD, Fox RH (1994) Evaluation of LEACHM: II. Simulation of Nitrate Leaching from Nitrogen-Fertilized and Manured Corn. *Agron J* 86:852. doi: 10.2134/agronj1994.00021962008600050019x
- Jensen ME (1968) Water consumption by agricultural plants (Chapter 1). *Water Deficits and Plant growth*
- Jeuffroy MH, Baranger E, Carrouée B, et al. (2013) Nitrous oxide emissions from crop rotations including wheat, oilseed rape and dry peas. *Biogeosciences* 10:1787–1797. doi: 10.5194/bg-10-1787-2013
- Johnson JM-F, Franzluebbers AJ, Weyers SL, Reicosky DC (2007) Agricultural opportunities to mitigate greenhouse gas emissions. *Environ Pollut* 150:107–124. doi: 10.1016/j.envpol.2007.06.030
- Jones CA, Kiniry JR, Dyke PT (1986) CERES-Maize: a simulation model of maize growth and development.
- Jones MN (1984) Nitrate reduction by shaking with cadmium: Alternative to cadmium columns. *Water Res* 18:643–646. doi: 10.1016/0043-1354(84)90215-X
- Jones RD, Hood MA (1980) Effects of temperature, pH, salinity, and inorganic nitrogen on the rate of ammonium oxidation by nitrifiers isolated from wetland environments. *Microb Ecol* 6:339–347. doi: 10.1007/BF02010496
- Jones SK, Rees RM, Skiba UM, Ball BC (2007) Influence of organic and mineral N fertiliser on N₂O fluxes from a temperate grassland. *Agric Ecosyst Environ* 121:74–83. doi: 10.1016/j.agee.2006.12.006
- Justes E, Beaudoin N, Bertuzzi P, et al. (2012) Réduire les fuites de nitrate au moyen de cultures intermédiaires : conséquences sur les bilans d'eau et d'azote, autres services écosystémiques.
- Justes E, Mary B, Nicolardot B (1999) Comparing the effectiveness of radish cover crop, oilseed rape volunteers and oilseed rape residues incorporation for reducing nitrate leaching. *Nutr Cycl Agroecosystems* 55:207–220. doi: 10.1023/A:1009870401779
- Justes E., Thiébeau P., Cattin G., Larbre D., Nicolardot B (2001) Libération d'azote après retournement de luzerne. *Perspectives Agricoles*. 264:22-28
- Kankanen H, Kangas A, Mela T (2008) Timing incorporation of different green manure crops to minimize the risk of nitrogen leaching. *Agric Food Sci in Finland* 7:553–567.
- Katerji N, Perrier A (1985) Détermination de la résistance globale d'un couvert végétal à la diffusion de vapeur d'eau et de ses différentes composantes. Approche théorique et vérification expérimentale sur une culture de luzerne. *Agric For Meteorol* 34:105–120. doi: 10.1016/0168-1923(85)90012-7
- Keeney DR, Fillery IR, Marx GP (1979) Effect of Temperature on the Gaseous Nitrogen Products of Denitrification in a Silt Loam Soil. *Soil Sci Soc Am J* 43:1124–1128. doi: 10.2136/sssaj1979.03615995004300060012x
- Khalil K, Mary B, Renault P (2004) Nitrous oxide production by nitrification and denitrification in soil aggregates as affected by O₂ concentration. *Soil Biol Biochem* 36:687–699. doi: 10.1016/j.soilbio.2004.01.004

- Kim J-H, Guo X, Park H-S (2008) Comparison study of the effects of temperature and free ammonia concentration on nitrification and nitrite accumulation. *Process Biochem* 43:154–160.
- Kirchmann H, Bergstrom L (2001) Do organic farming practices reduce nitrate leaching? *Commun Soil Sci Plant Anal* 32:997–1028. doi: 10.1081/CSS-100104101
- Kladivko EJ, Frankenberger JR, Jaynes DB, et al. (2004) Nitrate Leaching to Subsurface Drains as Affected by Drain Spacing and Changes in Crop Production System. *J Environ Qual* 33:1803. doi: 10.2134/jeq2004.1803
- Knowles R (1982) Denitrification. *Microbiol Rev* 46:43–70.
- Kool DM, Dolfing J, Wrage N, Van Groenigen JW (2011) Nitrifier denitrification as a distinct and significant source of nitrous oxide from soil. *Soil Biol Biochem* 43:174–178.
- Korsaeth A (2008) Relations between nitrogen leaching and food productivity in organic and conventional cropping systems in a long-term field study. *Agric Ecosyst Environ* 127:177–188. doi: 10.1016/j.agee.2008.03.014
- Korsaeth A, Eltun R (2000) Nitrogen mass balances in conventional, integrated and ecological cropping systems and the relationship between balance calculations and nitrogen runoff in an 8-year field experiment in Norway. *Agric Ecosyst Environ* 79:199–214. doi: 10.1016/S0167-8809(00)00129-8
- Kristensen SP, Mathiasen J, Lassen J, et al. (1994) A Comparison of the Leachable Inorganic Nitrogen Content in Organic and Conventional Farming Systems. *Acta Agric Scand Sect B - Soil Plant Sci* 44:19–27. doi: 10.1080/09064719409411253
- Lafolie F, Neel C (1997) Migration du nitrate dans la zone non saturée et impacts sur la qualité des eaux souterraines. *Colloq. INRA* No 83
- Lamine C and Bellon S (2009) *Transitions vers l'agriculture biologique*. Ed Quae 318 pp.
- Lancelot C, Thieu V, Polard A, et al. (2011) Cost assessment and ecological effectiveness of nutrient reduction options for mitigating *Phaeocystis* colony blooms in the Southern North Sea: An integrated modeling approach. *Sci Total Environ* 409:2179–2191. doi: 10.1016/j.scitotenv.2011.02.023
- Laverman AM, Garnier JA, Mounier EM, Roose-Amsaleg CL (2010) Nitrous oxide production kinetics during nitrate reduction in river sediments. *Water Res* 44:1753–1764. doi: 10.1016/j.watres.2009.11.050
- Laville P, Jambert C, Cellier P, Delmas R (1999) Nitrous oxide fluxes from a fertilised maize crop using micrometeorological and chamber methods. *Agric For Meteorol* 96:19–38. doi: 10.1016/S0168-1923(99)00054-4
- Leclerc B (2001) *Guide des matières organiques Tome 1*. Ed ITAB 238 pp.
- Ledoux E, Gomez E, Monget JM, et al. (2007) Agriculture and groundwater nitrate contamination in the Seine basin. The STICS–MODCOU modelling chain. *Sci Total Environ* 375:33–47. doi: 10.1016/j.scitotenv.2006.12.002
- Lehman JT, Botkin DB, Likens GE (1975) The assumptions and rationales of a computer model of phytoplankton population dynamics. *Limnol Ocean* 20:343–364.
- Leip A, Marchi G, Koeble R, et al. (2008) Linking an economic model for European agriculture with a mechanistic model to estimate nitrogen and carbon losses from arable soils in Europe. *Biogeosciences* 5:73–94.
- Lemaire G and Nicolardot B (1997) *Maîtrise de l'azote dans les agrosystèmes*. Ed Quae 333 pp.

- Leridon H and de Marsily G (2011) Démographie, climat et alimentation mondiale. Rapport sur la Science et la Technologie N°32. Académie des Sciences. Ed EDP Sci 313 pp. ISBN : 978-2-7598-05581-5.
- Lesschen JP, van den Berg M, Westhoek HJ, et al. (2011) Greenhouse gas emission profiles of European livestock sectors. *Anim Feed Sci Technol* 166–167:16–28. doi: 10.1016/j.anifeedsci.2011.04.058
- Le Viol I, Chiron F, Julliard R, Kerbiriou C (2012) More amphibians than expected in highway stormwater ponds. *Ecol Eng* 47:146–154. doi: 10.1016/j.ecoleng.2012.06.031
- Livingston GP, Hutchinson GL (1995) Biogenic Trace Gases: Measuring Emissions from Soil and Water. John Wiley & Sons 409 pp.
- Lord EI, Shepherd MA (1993) Developments in the use of porous ceramic cups for measuring nitrate leaching. *J Soil Sci* 44:435–449. doi: 10.1111/j.1365-2389.1993.tb00466.x
- Lotter DW (2003) Organic Agriculture. *J Sustain Agric* 21:59–128. doi: 10.1300/J064v21n04_06
- Loubet B, Laville P, Lehuger S, et al. (2011) Carbon, nitrogen and Greenhouse gases budgets over a four years crop rotation in northern France. *Plant Soil* 343:109–137. doi: 10.1007/s11104-011-0751-9
- Luo Y, Wan S, Hui D, Wallace LL (2001) Acclimatization of soil respiration to warming in a tall grass prairie. *Nature* 413:622–625. doi: 10.1038/35098065
- Maag M, Vinther FP (1996) Nitrous oxide emission by nitrification and denitrification in different soil types and at different soil moisture contents and temperatures. *Appl Soil Ecol* 4:5–14.
- Macdonald AJ, Poulton PR, Powlson DS, Jenkinson DS (1997) Effects of season, soil type and cropping on recoveries, residues and losses of ¹⁵N-labelled fertilizer applied to arable crops in spring. *J Agric Sci* 129:125–154.
- Macdonald AJ, Powlson DS, Poulton PR, Jenkinson DS (1989) Unused fertiliser nitrogen in arable soils—its contribution to nitrate leaching. *J Sci Food Agric* 46:407–419. doi: 10.1002/jsfa.2740460404
- Machet JM and Mary B (1990) Effet de différentes successions culturales sur les risques des pertes de nitrate en région de grande culture. In: *Nitrates-Agriculture-Eau*, Ed INRA 395-404.
- MacMillan T, Benton TG (2014) Agriculture: Engage farmers in research. *Nature* 509:25–27. doi: 10.1038/509025a
- Maître V (1991) Protocole d'extraction, de conservation et de filtration des eaux libres du sol. *Sci Sol* 29:71–76.
- Maître V, Bourrie G, Curmi P (1991) Contamination of collected soil water samples by the dissolution of the mineral constituents of porous P.T.F.E. cups. *Soil Sci* 152:4
- Mary B, Beaudoin N, Justes E, Machet JM (1999) Calculation of nitrogen mineralization and leaching in fallow soil using a simple dynamic model. *Eur J Soil Sci* 50:549–566. doi: 10.1046/j.1365-2389.1999.00264.x
- Medina LC, Sartain JB, Obreza TA, et al. (2014) Evaluation of a Soil Incubation Method to Characterize Nitrogen Release Patterns of Slow- and Controlled-Release Fertilizers. *J Aoac Int* 97:643–660. doi: 10.5740/jaoacint.13-065
- Mégnyen C (1979) Hydrogéologie du centre du Bassin de Paris: contribution à l'étude de quelques aquifères principaux. Principaux résultats scientifiques et techniques du Service géologique national, Paris, 122 pp.

- Meng L, Ding W, Cai Z (2005) Long-term application of organic manure and nitrogen fertilizer on N₂O emissions, soil quality and crop production in a sandy loam soil. *Soil Biol Biochem* 37:2037–2045. doi: 10.1016/j.soilbio.2005.03.007
- Meynard J-M, Dore T, Habib R (2001) L'évaluation et la conception de systèmes de culture pour une agriculture durable. *Comptes Rendus Académie Agric Fr* 87:223–236.
- Meynard J-M, Doré T, Lucas P (2003) Agronomic approach: cropping systems and plant diseases. *C R Biol* 326:37–46. doi: 10.1016/S1631-0691(03)00006-4
- Misselbrook TH, Van Der Weerden TJ, Pain BF, et al. (2000) Ammonia emission factors for UK agriculture. *Atmos Environ* 34:871–880. doi: 10.1016/S1352-2310(99)00350-7
- Moller K, Stinner W, Leithold G (2008) Growth, composition, biological N₂ fixation and nutrient uptake of a leguminous cover crop mixture and the effect of their removal on field nitrogen balances and nitrate leaching risk. *Nutr Cycl Agroecosystems* 82:233–249. doi: 10.1007/s10705-008-9182-2
- Mondelaers K, Aertsens J, Van Huylenbroeck G (2009) A meta-analysis of the differences in environmental impacts between organic and conventional farming. *Br Food J* 111:1098–1119.
- Morizet J, Robelin M, Baucher G, et al. (1984) Résultats de 18 années d'observations lysimétriques sous climat limagnais. II. Etude des relations entre l'eau et la production végétale. *Agronomie* 4:407–416.
- Mosier A, Kroeze C, Nevison C, et al. (1998) Closing the global N₂O budget: nitrous oxide emissions through the agricultural nitrogen cycle. *Nutr Cycl Agroecosystems* 52:225–248. doi: 10.1023/A:1009740530221
- Muller JC, Ledain C (1992) Epandage des eaux résiduaires des industries alimentaires et agricoles et valorisation agronomique en Champagne-Ardenne: évolution des idées et solutions techniques. *Ind Aliment Agric* 109:531–536.
- Muller S, Pereira P a. A, Martin P (1993) Effect of different levels of mineral nitrogen on nodulation and N₂ fixation of two cultivars of common bean (*Phaseolus vulgaris* L.). *Plant Soil* 152:139–143. doi: 10.1007/BF00016343
- Myrold DD, Tiedje JM (1985) Establishment of denitrification capacity in soil: Effects of carbon, nitrate and moisture. *Soil Biol Biochem* 17:819–822. doi: 10.1016/0038-0717(85)90140-3
- Nguyen ML, Haynes RJ, Goh KM (1995) Nutrient budgets and status in three pairs of conventional and alternative mixed cropping farms in Canterbury, New Zealand. *Agric Ecosyst Environ* 52:149–162.
- Nicola L, Schott C, Mignolet C (2011) Dynamique de changement des pratiques agricoles dans le bassin versant de l'Orgeval et création de la base de données APOCA (Agricultural Practices of the Orgeval Catchment Area). Rapport PIREN-Seine 48pp
- Nicollaud B, Darthout R, Duval O (1995) Etude de l'enracinement du blé tendre d'hiver et du maïs dans les sols argilo-limoneux de Petite Beauce. *Etude et gestion des sols*, 2, 3.
- Nieder R, Kersebaum KC, Richter J (1995) Significance of nitrate leaching and long term N immobilization after deepening the plough layers for the N regime of arable soils in N.W. Germany. *Plant Soil* 173:167–175. doi: 10.1007/BF00155528
- Oenema O, Kros H, de Vries W (2003) Approaches and uncertainties in nutrient budgets: implications for nutrient management and environmental policies. *Eur J Agron* 20:3–16. doi: 10.1016/S1161-0301(03)00067-4

- Olesen JE, Petersen BM, Berntsen J, et al. (2002) Comparison of methods for simulating effects of nitrogen on green area index and dry matter growth in winter wheat. *Field Crops Res* 74:131–149. doi: 10.1016/S0378-4290(01)00204-0
- Olesen J, Schelde K, Weiske A, et al. (2006) Modelling greenhouse gas emissions from European conventional and organic dairy farms. *Agric Ecosyst Environ* 112:207–220. doi: 10.1016/j.agee.2005.08.022
- Oorts K, Merckx R, Gréhan E, et al. (2007) Determinants of annual fluxes of CO₂ and N₂O in long-term no-tillage and conventional tillage systems in northern France. *Soil Tillage Res* 95:133–148.
- Pacini C, Wossink A, Giesen G, et al. (2003) Evaluation of sustainability of organic, integrated and conventional farming systems: a farm and field-scale analysis. *Agric Ecosyst Environ* 95:273–288. doi: 10.1016/S0167-8809(02)00091-9
- Pappa VA, Rees RM, Walker RL, et al. (2011) Nitrous oxide emissions and nitrate leaching in an arable rotation resulting from the presence of an intercrop. *Agric Ecosyst Environ* 141:153–161. doi: 10.1016/j.agee.2011.02.025
- Parnaudeau V, Condom N, Oliver R, et al. (2008) Vinasse organic matter quality and mineralization potential, as influenced by raw material, fermentation and concentration processes. *Bioresour Technol* 99:1553–1562. doi: 10.1016/j.biortech.2007.04.012
- Passy P, Garnier J, Billen G, et al. (2012) Restoration of ponds in rural landscapes: Modelling the effect on nitrate contamination of surface water (the Seine River Basin, France). *Sci Total Environ* 430:280–290. doi: 10.1016/j.scitotenv.2012.04.035
- Passy P, Gypens N, Billen G, et al. (2013) A model reconstruction of riverine nutrient fluxes and eutrophication in the Belgian Coastal Zone since 1984. *J Mar Syst* 128:106–122. doi: 10.1016/j.jmarsys.2013.05.005
- Pattey E, Edwards GC, Desjardins RL, et al. (2007) Tools for quantifying N₂O emissions from agroecosystems. *Agric For Meteorol* 142:103–119. doi: 10.1016/j.agrformet.2006.05.013
- Pelosi C, Barot S, Capowicz Y, et al. (2014) Pesticides and earthworms. A review. *Agron Sustain Dev* 34:199–228. doi: 10.1007/s13593-013-0151-z
- Pelosi C, Bertrand M, Roger-Estrade J (2009) Earthworm community in conventional, organic and direct seeding with living mulch cropping systems. *Agron Sustain Dev* 29:287–295. doi: 10.1051/agro/2008069
- Pendall E, Bridgham S, Hanson PJ, et al. (2004) Below-ground process responses to elevated CO₂ and temperature: a discussion of observations, measurement methods, and models. *New Phytol* 162:311–322. doi: 10.1111/j.1469-8137.2004.01053.x
- Perrier A, Katerji N, Gosse G, Itier B (1980) Etude “In Situ” de l’évapotranspiration réelle d’une culture de ble. *Agric Meteorol* 21:295–311. doi: 10.1016/0002-1571(80)90073-4
- Petersen SO (1999) Nitrous Oxide Emissions from Manure and Inorganic Fertilizers Applied to Spring Barley. *J Environ Qual* 28:1610. doi: 10.2134/jeq1999.00472425002800050027x
- Petersen S, Regina K, Pollinger A, et al. (2006) Nitrous oxide emissions from organic and conventional crop rotations in five European countries. *Agric Ecosyst Environ* 112:200–206. doi: 10.1016/j.agee.2005.08.021
- Petit C (2013) Thèse : Transitions des exploitations agricoles vers l’agriculture biologique dans un territoire : approche par les interactions entre systèmes techniques et de

- commercialisation. Application aux aires d'alimentation de captages en Ile-de-France, 412 pp.
- Pinay G, Roques L, Fabre A (1993) Spatial and Temporal Patterns of Denitrification in a Riparian Forest. *J Appl Ecol* 30:581–591. doi: 10.2307/2404238
- Poss R, Noble AD, Dunin FX, Reyenga W (1995) Evaluation of ceramic cup samplers to measure nitrate leaching in the field. *Eur J Soil Sci* 46:667–674.
- Rassam DW, Pagendam DE, Hunter HM (2008) Conceptualisation and application of models for groundwater–surface water interactions and nitrate attenuation potential in riparian zones. *Environ Model Softw* 23:859–875. doi: 10.1016/j.envsoft.2007.11.003
- Ravishankara AR, Daniel JS, Portmann RW (2009) Nitrous Oxide (N₂O): The Dominant Ozone-Depleting Substance Emitted in the 21st Century. *Science* 326:123–125. doi: 10.1126/science.1176985
- Recous S, Machet JM, Mary B (1988) The fate of labelled ¹⁵N urea and ammonium nitrate applied to a winter wheat crop. *Plant Soil* 112:215–224. doi: 10.1007/BF02139998
- Reichman R, Rolston DE (2002) Design and performance of a dynamic gas flux chamber. *J Environ Qual* 31:1774–1781.
- Riley W., Matson P (2000) NLOSS: a mechanistic model of denitrified N₂O and N₂ evolution from soil. *Soil Sci* 165:237–249
- Rinnofner T, Friedel JK, de Kruijff R, et al. (2008) Effect of catch crops on N dynamics and following crops in organic farming. *Agron Sustain Dev* 28:551–558. doi: 10.1051/agro:2008028
- Rodrigo A, Recous S, Neel C, Mary B (1997) Modelling temperature and moisture effects on C–N transformations in soils: comparison of nine models. *Ecol Model* 102:325–339. doi: 10.1016/S0304-3800(97)00067-7
- Rolland B, Oury F-X, Bouchard C, Loyce C (2006) Vers une évolution de la création variétale pour répondre aux besoins de l'agriculture durable? L'exemple du blé tendre. *Doss L'environnement L'INRA* 30:79–90.
- Romero E, Garnier J, Lassaletta L, et al. (2013) Large-scale patterns of river inputs in southwestern Europe: seasonal and interannual variations and potential eutrophication effects at the coastal zone. *Biogeochemistry* 113:481–505. doi: 10.1007/s10533-012-9778-0
- RotAB (2011) Rotations pratiquées en grandes cultures biologiques en France : état des lieux par région. ITAB 65 pp.
- Ruelland D, Billen G, Brunstein D, Garnier J (2007) SENEQUE: A multi-scaling GIS interface to the Riverstrahler model of the biogeochemical functioning of river systems. *Sci Total Environ* 375:257–273. doi: 10.1016/j.scitotenv.2006.12.014
- Ruggiero A, Céréghino R, Figuerola J, et al. (2008) Farm ponds make a contribution to the biodiversity of aquatic insects in a French agricultural landscape. *C R Biol* 331:298–308. doi: 10.1016/j.crvi.2008.01.009
- Rustad L, Campbell J, Marion G, et al. (2001) A meta-analysis of the response of soil respiration, net nitrogen mineralization, and aboveground plant growth to experimental ecosystem warming. *Oecologia* 126:543–562. doi: 10.1007/s004420000544
- Saad OALO, Conrad R (1993) Temperature dependence of nitrification, denitrification, and turnover of nitric oxide in different soils. *Biol Fertil Soils* 15:21–27.

- Saggar S, Jha N, Deslippe J, Bolan NS et al. (2012) Denitrification and $\text{N}_2\text{O}:\text{N}_2$ production in temperate grasslands: Processes, measurements, modelling and mitigating negative impacts, *Sci. Total Environ.* 465, 173-195. <http://dx.doi.org/10.1016/j.scitotenv.2012.11.050>.
- Saleh-Lakha S, Shannon KE, Henderson SL, et al. (2009) Effect of pH and temperature on denitrification gene expression and activity in *Pseudomonas mandelii*. *Appl Environ Microbiol* 75:3903–3911.
- Sanz-Cobena A, Sánchez-Martín L, García-Torres L, Vallejo A (2012) Gaseous emissions of N_2O and NO and NO_3^- leaching from urea applied with urease and nitrification inhibitors to a maize (*Zea mays*) crop. *Agric Ecosyst Environ* 149:64–73. doi: 10.1016/j.agee.2011.12.016
- Sapkota TB, Askegaard M, Lægdsmand M, Olesen JE (2012) Effects of catch crop type and root depth on nitrogen leaching and yield of spring barley. *Field Crops Res* 125:129–138. doi: 10.1016/j.fcr.2011.09.009
- Schaufler G, Kitzler B, Schindlbacher A, et al. (2010) Greenhouse gas emissions from European soils under different land use: effects of soil moisture and temperature. *Eur J Soil Sci* 61:683–696. doi: 10.1111/1.1365-2389.2010.01277.x
- Scherer HW (1993) Dynamics and availability of the non-exchangeable $\text{NH}_4\text{-N}$ — a review. *Eur J Agron* 2:149–160. doi: 10.1016/S1161-0301(14)80124-X
- Schindlbacher A, Zechmeister-Boltenstern S, Butterbach-Bahl K (2004) Effects of soil moisture and temperature on NO , NO_2 , and N_2O emissions from European forest soils. *J Geophys Res Atmospheres* 109:D17302. doi: 10.1029/2004JD004590
- Schott C., Mignolet C., Benoît M. (2011) Agriculture du bassin de la Seine - PIREN-Seine fascicule
- Scoones I, Toulmin C (1998) Soil nutrient balances: what use for policy? *Agric Ecosyst Environ* 71:255–267. doi: 10.1016/S0167-8809(98)00145-5
- Sebillotte M (1974) Agronomie et agriculture. Essai d'analyse des tâches de l'agronome. *Cah L'ORSTOM* 24:3–25.
- Sebillotte M and Meynard JM (1990) Systèmes de culture, systèmes d'élevage et pollutions azotées. Nitrates – Agriculture – Eau : International Simposium : Paris la Défense
- Sebilo M, Billen G, Grably M, Mariotti A (2003) Isotopic composition of nitrate-nitrogen as a marker of riparian and benthic denitrification at the scale of the whole Seine River system. *Biogeochemistry* 63:35–51. doi: 10.1023/A:1023362923881
- Sebilo M, Mayer B, Nicolardot B, et al. (2013) Long-term fate of nitrate fertilizer in agricultural soils. *Proc Natl Acad Sci* 201305372. doi: 10.1073/pnas.1305372110
- Sharpley AN, Chapra SC, Wedepohl R, et al. (1994) Managing Agricultural Phosphorus for Protection of Surface Waters: Issues and Options. *J Environ Qual* 23:437. doi: 10.2134/jeq1994.00472425002300030006x
- Skinner C, Gattinger A, Muller A, et al. (2014) Greenhouse gas fluxes from agricultural soils under organic and non-organic management—A global meta-analysis. *Sci Total Environ* 468:553–563.
- Slawyk G, MacIsaac JJ (1972) Comparison of two automated ammonium methods in a region of coastal upwelling. *Deep Sea Res Oceanogr Abstr* 19:521–524. doi: 10.1016/0011-7471(72)90019-8
- Smith KA, Thomson PE, Clayton H, et al. (1998) Effects of temperature, water content and nitrogen fertilisation on emissions of nitrous oxide by soils. *Atmos Environ* 32:3301–3309.

- Smith MS and Zimmerman K (1981) Nitrous Oxide Production by Nondenitrifying Soil Nitrate Reducers¹. *Soil Sci Soc Am J* 45:865. doi: 10.2136/sssaj1981.03615995004500050008x
- Solberg SØ (1995) Influence of Crops and Cultivation Management on the Nitrogen Leaching Potential on Ecological Farms in South East Norway. *Biol Agric Hortic* 11:115–121. doi: 10.1080/01448765.1995.9754698
- Sommer SG, Petersen SO, Møller HB (2004) Algorithms for calculating methane and nitrous oxide emissions from manure management. *Nutr Cycl Agroecosystems* 69:143–154. doi: 10.1023/B:FRES.0000029678.25083.fa
- Stalenga J, Kawalec A (2008) Emission of greenhouse gases and soil organic matter balance in different farming systems. *Int AGROPHYSICS* 22:287–290.
- Stanford G, Dzienia S, Pol V AR (1975) Effect of Temperature on Denitrification Rate in Soils. *Soil Sci Soc Am J* 39:867–870. doi: 10.2136/sssaj1975.03615995003900050024x
- Stehfest E, Bouwman L (2006) N₂O and NO emission from agricultural fields and soils under natural vegetation: summarizing available measurement data and modeling of global annual emissions. *Nutr Cycl Agroecosystems* 74:207–228. doi: 10.1007/s10705-006-9000-7
- Stenberg M, Aronsson H, Lindén B, et al. (1999) Soil mineral nitrogen and nitrate leaching losses in soil tillage systems combined with a catch crop. *Soil Tillage Res* 50:115–125. doi: 10.1016/S0167-1987(98)00197-4
- Sterckeman T and Ciesielski H (1991) L'azote et le soufre dans le sol. Ed Frontières 229 pp.
- Stolze M, Piorr A, Haring A, Dabbert S (2000) Organic farming in Europe: economics and policy. *Economics and Policy* 6:1-143
- Stopes C, Lord E i., Philipps L, Woodward L (2002) Nitrate leaching from organic farms and conventional farms following best practice. *Soil Use Manag* 18:256–263. doi: 10.1111/j.1475-2743.2002.tb00267.x
- Sullivan PG, Parrish DJ, Luna JM (1991) Cover crop contributions to N supply and water conservation in corn production. *Am J Altern Agric* 6:106–113. doi: 10.1017/S0889189300003982
- Sundkvist A, Jansson A, Larsson P (2001) Strengths and limitations of localizing food production as a sustainability-building strategy — an analysis of bread production on the island of Gotland, Sweden. *Ecol Econ* 37:217–227. doi: 10.1016/S0921-8009(00)00277-9
- Sutton MA, Howard CM, Erisman JW, et al. (2011) The European Nitrogen Assessment: Sources, Effects and Policy Perspectives. Cambridge University Press 665 pp.
- Swinnen J and Squicciarini P (2012) Mixed messages on prices and food security, *Science* 335: 405-406.
- Syvasalo E, Regina K, Turtola E, et al. (2006) Fluxes of nitrous oxide and methane, and nitrogen leaching from organically and conventionally cultivated sandy soil in western Finland. *Agric Ecosyst Environ* 113:342–348. doi: 10.1016/j.agee.2005.10.013
- Tallec G, Garnier J, Gousailles M (2006) Nitrogen removal in a wastewater treatment plant through biofilters: nitrous oxide emissions during nitrification and denitrification. *Bioprocess Biosyst Eng* 29:323–333. doi: 10.1007/s00449-006-0081-0
- Thieu V, Billen G, Garnier J (2009) Nutrient transfer in three contrasting NW European watersheds: the Seine, Somme, and Scheldt Rivers. A comparative application of the Seneque/Riverstrahler model. *Water Res* 43:1740–1754.

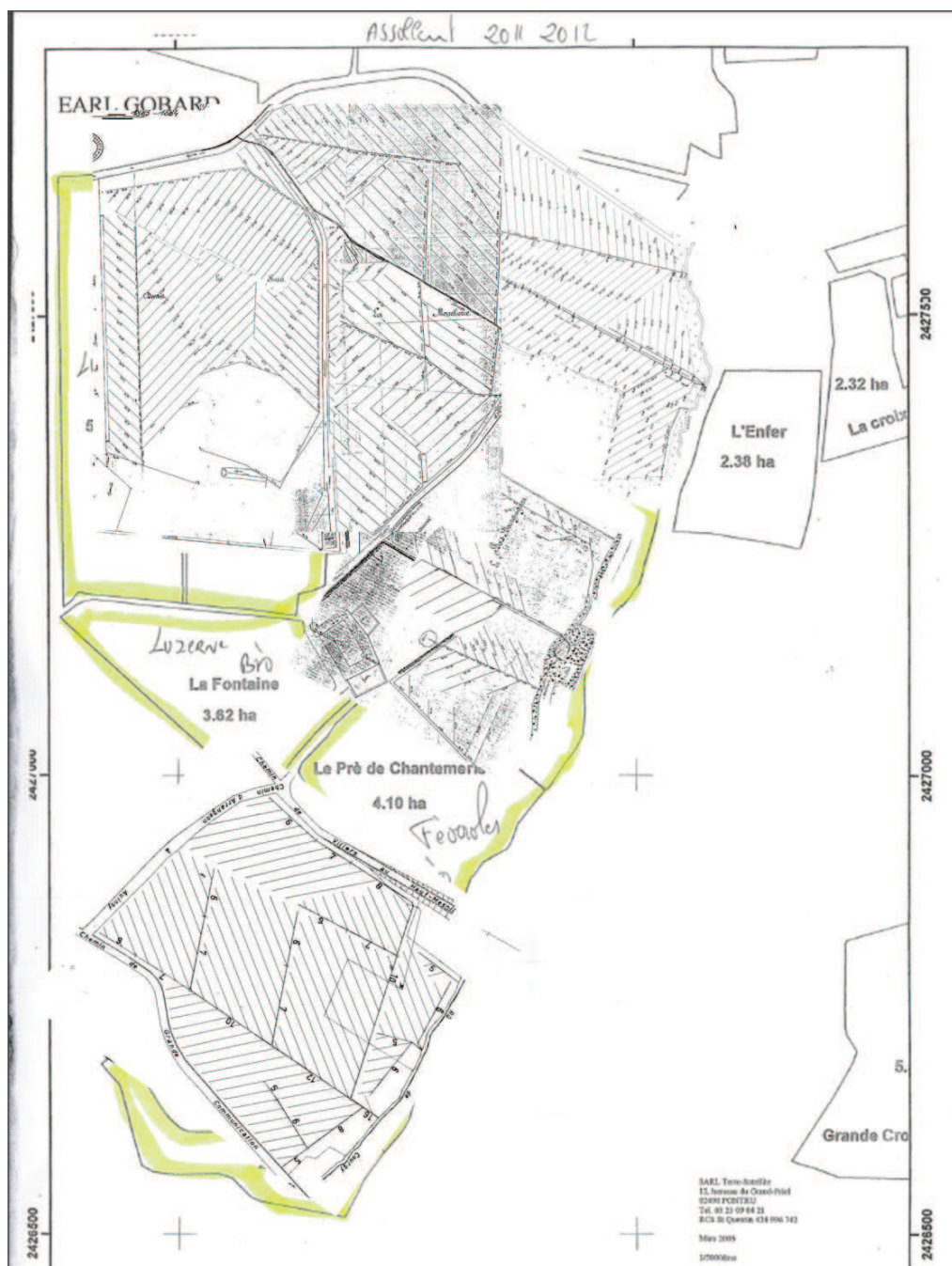
- Thieu V, Billen G, Garnier J, Benoît M (2011) Nitrogen cycling in a hypothetical scenario of generalised organic agriculture in the Seine, Somme and Scheldt watersheds. *Reg Environ Change* 11:359–370.
- Thorup-Kristensen K, Magid J, Jensen LS (2003) Catch crops and green manures as biological tools in nitrogen management in temperate zones. *Adv. Agron. Academic Press* 227–302.
- Thouvenot-Korppoo M, Billen G, Garnier J (2009) Modelling benthic denitrification processes over a whole drainage network. *J Hydrol* 379:239–250. doi: 10.1016/j.jhydrol.2009.10.005
- Tiemeyer B, Kahle P, Lennartz B (2006) Nutrient losses from artificially drained catchments in North-Eastern Germany at different scales. *Agric Water Manag* 85:47–57. doi: 10.1016/j.agwat.2006.03.016
- Tomaszek JA, Czerwieniec E (2000) In situ chamber denitrification measurements in reservoir sediments: an example from southeast Poland. *Ecol Eng* 16:61–71. doi: 10.1016/S0925-8574(00)00090-2
- Tomer MD, Meek DW, Jaynes DB, Hatfield JL (2003) Evaluation of Nitrate Nitrogen Fluxes from a Tile-Drained Watershed in Central Iowa. *J Environ Qual* 32:642. doi: 10.2134/jeq2003.6420
- Torstensson G, Aronsson H, Bergstrom L (2006) Nutrient use efficiencies and leaching of organic and conventional cropping systems in Sweden. *Agron J* 98:603–615. doi: 10.1234/agronj2005.0224
- Tournebize J, Arlot MP, Billy C, Birgand F, Gillet JP, Dutertre A (2008) Quantification et maîtrise des flux en nitrates : de la parcelle drainée au bassin versant. *Ingénieries Eau-Agriculture-Territoire*, n° spécial, 25 pp.
- Tournebize J, Clement R, Guerin A, Ansart P, Chaumont C, Morau S, Henine H (2011) Y a-t-il infiltration vertical sous drainage agricole ? Conséquence pour la recharge de la nappe de Brie. *Rapport Piren-Seine*.
- Tuomisto HL, Helenius J (2008) Comparison of energy and greenhouse gas balances of biogas with other transport biofuel options based on domestic agricultural biomass in Finland. *Agric Food Sci* 17:240–251. doi: 10.2137/145960608786118857
- Tuomisto HL, Hodge ID, Riordan P, Macdonald DW (2012) Does organic farming reduce environmental impacts? – A meta-analysis of European research. *J Environ Manage* 112:309–320. doi: 10.1016/j.jenvman.2012.08.018
- UWWTD (Urban wastewater treatment Directive), 1991. 91/271/CEE du Conseil, du 21mai 1991, relative au traitement des eaux urbaines résiduaires, JO L 135 du 30.5.1991, 40–52.
- Vachaud G, Dancette C, Sonko S, Thony L (1978) Field Determination of Hydraulic Characteristics of an Unsaturated Soil - Application on 2 Typical Soils of Republic of Senegal with an Example. *Ann Agron* 29:1–36.
- Van Es HM, Sogbedji JM, Schindelbeck RR (2006) Effect of Manure Application Timing, Crop, and Soil Type on Nitrate Leaching. *J Environ Qual* 35:670. doi: 10.2134/jeq2005.0143
- Veldkamp E, Keller M (1997) Nitrogen oxide emissions from a banana plantation in the humid tropics. *J Geophys Res Atmospheres* 102:15889–15898. doi: 10.1029/97JD00767

- Vermue A (2014) Thèse : Contribution à l'évaluation multicritère des systèmes de cultures en protection intégrée contre les adventices : cas des émissions de protoxyde d'azote, 118 pp.
- Viennot P, Ledoux E, Ducharne A, Devos A, Lejeune O (2011) Hydro-géologie du bassin de la Seine – PIREN Seine fascicule, 56 pp.
- Vilain G, Garnier J, Decuq C, Lugnot M (2014) Nitrous oxide production from soil experiments: denitrification prevails over nitrification. *Nutr Cycl Agroecosystems* 98:169–186. doi: 10.1007/s10705-014-9604-2
- Vilain G, Garnier J, Passy P, et al. (2012a) Budget of N₂O emissions at the watershed scale: role of land cover and topography (the Orgeval basin, France). *Biogeosciences* 9:1085–1097. doi: 10.5194/bg-9-1085-2012
- Vilain G, Garnier J, Roose-Amsaleg C, Laville P (2012b) Potential of denitrification and nitrous oxide production from agricultural soil profiles (Seine Basin, France). *Nutr Cycl Agroecosystems* 92:35–50. doi: 10.1007/s10705-011-9470-0
- Vilain G, Garnier J, Tallec G, Cellier P (2010) Effect of slope position and land use on nitrous oxide (N₂O) emissions (Seine Basin, France). *Agric For Meteorol* 150:1192–1202.
- Vilain G, Garnier J, Tallec G, Tournebize J (2012c) Indirect N₂O emissions from shallow groundwater in an agricultural catchment (Seine Basin, France). *Biogeochemistry* 111:253–271. doi: 10.1007/s10533-011-9642-7
- Volk M, Liersch S, Schmidt G (2009) Towards the implementation of the European Water Framework Directive? Lessons learned from water quality simulations in an agricultural watershed. *Land use policy* 26:580–588. doi: 10.1016/j.landusepol.2008.08.005
- Wagner G (1962) Use of porous cermaic cups to sample soil water within the profile. *Soil Sci* 94:379–386
- Wang K, Zheng X, Pihlatie M, et al. (2013) Comparison between static chamber and tunable diode laser-based eddy covariance techniques for measuring nitrous oxide fluxes from a cotton field. *Agric For Meteorol* 171–172:9–19. doi: 10.1016/j.agrformet.2012.11.009
- Wanninkhof R (1992) Relationship between wind speed and gas exchange over the ocean. *J Geophys Res Oceans* 97:7373–7382. doi: 10.1029/92JC00188
- Ward MH, Kok TM, Levallois P, Brender J, Gulis G, Nolan BT et al. (2005) Workgroup report: drinking-water nitrate and health— recent findings and research needs. *Environ Health Persp*. 113:1607–1614
- WFD (Water Framework Directive) (2000) OJ L 327/1, 22.12:1–72.
- Webster C, Shepherd M, Goulding K, Lord E (1993) Comparisons of Methods for Measuring the Leaching of Mineral Nitrogen from Arable Land. *J Soil Sci* 44:49–62.
- Weihermuller L, Siemens J, Deurer M, et al. (2007) In Situ Soil Water Extraction: A Review. *J Environ Qual* 36:1735. doi: 10.2134/jeq2007.0218
- Weiss RF, Price BA (1980) Nitrous oxide solubility in water and seawater. *Mar Chem* 8:347–359. doi: 10.1016/0304-4203(80)90024-9
- Welti N, Bondar-Kunze E, Singer G, et al. (2012) Large-scale controls on potential respiration and denitrification in riverine floodplains. *Ecol Eng* 42:73–84. doi: 10.1016/j.ecoleng.2012.02.005
- Whitehead PG, Wilson EJ, Butterfield D (1998) A semi-distributed Integrated Nitrogen model for multiple source assessment in Catchments (INCA): Part I — model structure

- and process equations. *Sci Total Environ* 210–211:547–558. doi: 10.1016/S0048-9697(98)00037-0
- Wosten JHM, Pachepsky YA, Rawls WJ (2001) Pedotransfer functions: bridging the gap between available basic soil data and missing soil hydraulic characteristics. *J Hydrol* 251:123–150. doi: 10.1016/S0022-1694(01)00464-4
- Wrage N, Velthof GL, van Beusichem ML, Oenema O (2001) Role of nitrifier denitrification in the production of nitrous oxide. *Soil Biol Biochem* 33:1723–1732. doi: 10.1016/S0038-0717(01)00096-7

Annexes

Annexe 1. Plan de drainage de l'exploitation de Chantemerle (Bassin d'Orgeval)



Annexe 2. Description des analyses de sol réalisées par le LAS, INRA-Arras

Description du SOL-0302 : Granulométrie 5 fractions sans décarbonatation, unité g/kg.

On détermine, selon la norme **NF X 31-107**, les proportions des classes de particules suivantes :

- Argiles : < 2 μm
- Limons fins : 2 μm à 20 μm
- Limons grossiers : 20 μm à 50 μm
- Sables fins : 0,050 mm à 0,200 mm
- Sables grossiers 0,200 mm à 2,00 mm

La détermination des fractions les plus fines (< 50 μm) s'effectue au moyen de 3 prélèvements successifs (**à la pipette dite de Robinson**) dans une suspension de sol en cours de **sédimentation**. La fraction des sables fins est séparée par passage sur tamis de 50 μm et sous courant d'eau de la suspension après prélèvements des fractions fines.

Prélèvements et tamisage sont réalisés après **destruction de la matière organique** par l'eau oxygénée (H_2O_2) sur une prise d'essai d'environ 10 g. La dispersion finale est réalisée par un court passage aux ultrasons après addition de dispersant $[(\text{NaPO}_3)_6 + \text{Na}_2\text{CO}_3]$ et après avoir au préalable séparé les sables grossiers (> 0,200 mm) par tamisage.

Les pesées après évaporation et séchage des fractions prélevées à la pipette permettent de déterminer les proportions des différentes classes granulométriques.

Les résultats sont exprimés par rapport à la phase minérale (somme des 5 fractions = 1000).

Description du SOL-0404 : Azote total par combustion sèche, unité g/kg.

La teneur en azote (**organique et minéral**) de l'échantillon est déterminée en le chauffant à environ 1000°C en présence d'oxygène.

Les produits de combustion ou décomposition sont réduits à l'état d'azote moléculaire (N_2). Les quantités de N_2 formées sont quantifiées, après séparation chromatographique, au moyen d'un catharomètre.

La prise d'essai est d'environ 50 mg d'échantillon broyé à 250 μm .

Le protocole suit la norme **NF ISO 13878**

Description du SOL-0403 : Carbone organique par combustion sèche, unité g/kg.

La méthode repose sur la **transformation en dioxyde de carbone (CO₂) de la totalité du carbone** présent dans l'échantillon. La réaction s'effectue en portant ce dernier à environ 1000°C en présence d'oxygène. Après séparation chromatographique, la quantité de gaz carbonique formée est quantifiée au moyen d'un catharomètre (conductibilité thermique).

Quand l'échantillon contient des quantités non négligeables de carbone minéral sous forme de carbonates, une correction ou un prétraitement de l'échantillon est nécessaire. Lorsque la détermination du calcaire total a été demandée sur l'échantillon, on va utiliser la teneur mesurée pour une éventuelle correction ($C_{\text{minéral}} = 0,12 \times CaCO_3$).

Dans le cas contraire, un prétraitement sera systématiquement appliqué. Par prétraitement on entend toute procédure utile à la quantification de la teneur en carbonates ou leur élimination avant analyse si nécessaire.

La prise d'essai est d'environ 50 mg d'échantillon broyé à 250 µm.

Le protocole suit la norme **NF ISO 10694**

Annexe 3. Caractéristiques physico-chimiques et hydrologiques des exploitations biologiques (a) et conventionnelles (b) équipées en bougies poreuses (réseau ABAC 2013-14). BD : Bulk Density, VWC : Volumetric Water Content, SOC : Soil Organic Carbon, N tot : N total, C/N et WHC_{fc} : Water Holding Capacity at field capacity.

(a)

Farm	depths	clay	silt	sand	BD	VWC	SOC	N tot	C/N	WHC _{fc}
	cm	g 100 g ⁻¹	g 100 g ⁻¹	g 100 g ⁻¹		mm	g 100 g ⁻¹	g 100 g ⁻¹		mm
OF1	0-30	22	67	11	1.50	75	1.11	0.11	9.8	176
	30-60	24	61	14	1.55	50	0.52	0.05	8.9	
	60-90	27	61	12	1.55	50	0.42	0.04	8.7	
OF2	0-30	21	74	5	1.50	75	1.13	0.11	9.9	209
	30-60	24	72	4	1.55	67	0.46	0.05	8.8	
	60-90	27	69	4	1.55	67	0.34	0.04	8.5	
OF3	0-30	18	66	16	1.50	75	0.94	0.09	10.1	176
	30-60	25	63	12	1.55	50	0.54	0.06	9.2	
	60-90	30	55	15	1.55	50	0.41	0.05	8.2	
OF4	0-30	24	67	9	1.50	75	1.88	0.17	11.0	192
	30-60	25	67	8	1.55	67	1.42	0.12	11.0	
	60-90	25	63	12	1.55	50	0.89	0.08	11.1	
OF5	0-30	24	67	9	1.50	75	1.29	0.12	10.3	195
	30-60	28	65	7	1.55	67	0.65	0.06	9.8	
	60-90	36	57	7	1.65	53	0.43	0.05	8.7	
OF6	0-30	18	57	26	1.50	67	1.02	0.08	11.5	167
	30-60	21	56	23	1.55	50	0.58	0.05	11.8	
	60-90	26	51	23	1.55	50	0.41	0.03	11.1	
OF7	0-30	18	73	8	1.50	75	0.86	0.08	9.9	209
	30-60	24	70	7	1.55	67	0.52	0.05	8.9	
	60-90	27	67	6	1.55	67	0.44	0.05	8.7	
OF8	0-30	32	59	9	1.65	53	1.45	0.13	10.7	157
	30-60	31	56	13	1.65	53	0.79	0.07	10.7	
	60-90	28	48	24	1.55	51	0.47	0.03	12.5	

(b)

Farm	depths	clay	silt	sand	BD	VWC	SOC	N tot	C/N	WHC _{fc} (%)
	cm	g 100 g ⁻¹	g 100 g ⁻¹	g 100 g ⁻¹		mm	g 100 g ⁻¹	g 100 g ⁻¹		mm
CF1	0-30	17	72	11	1.30	71	1.11	0.11	10.2	222
	30-60	16	72	12	1.55	84	0.63	0.06	9.4	
	60-90	20	68	12	1.55	67	0.67	0.07	9.1	
CF2	0-30	21	65	14	1.50	75	1.16	0.11	10.1	192
	30-60	23	66	11	1.55	67	0.75	0.07	9.6	
	60-90	24	65	11	1.55	50	0.54	0.05	9.1	
CF3	0-30	22	74	5	1.50	75	1.11	0.11	9.7	195
	30-60	27	69	4	1.55	67	0.50	0.05	9.0	
	60-90	26	71	4	1.65	53	0.32	0.03	8.7	
CF4	0-30	20	68	12	1.50	67	0.95	0.09	9.9	167
	30-60	22	65	13	1.55	50	0.62	0.06	9.3	
	60-90	23	63	14	1.55	50	0.43	0.05	8.7	
CF5	0-30	21	65	14	1.50	75	1.06	0.09	11.0	176
	30-60	25	60	15	1.55	50	0.37	0.04	8.6	
	60-90	28	54	18	1.55	50	0.32	0.04	8.4	
CF6	0-30	25	65	10	1.50	75	1.37	0.12	10.8	201
	30-60	26	66	9	1.50	75	0.70	0.07	9.8	
	60-90	37	53	10	1.55	51	0.46	0.06	8.3	
CF7	0-30	21	72	7	1.50	75	0.96	0.08	10.7	209
	30-60	29	66	6	1.55	67	0.53	0.05	9.3	
	60-90	29	67	5	1.55	67	0.30	0.03	8.0	
CF8	0-30	20	58	22	1.50	67	0.95	0.08	11.1	168
	30-60	26	55	19	1.55	50	0.46	0.04	10.8	
	60-90	36	44	20	1.55	51	0.28	0.03	9.1	
CF9	0-30	24	63	13	1.50	67	1.89	0.15	12.0	167
	30-60	24	62	14	1.55	50	0.97	0.08	12.6	
	60-90	27	61	12	1.55	50	0.55	0.04	12.4	
CF10	0-30	28	66	6	1.50	75	1.34	0.14	9.8	209
	30-60	28	67	5	1.55	67	0.74	0.08	9.0	
	60-90	31	65	4	1.55	67	0.44	0.05	8.4	

Annexe 4. Liste des mesures d'actions afin de réduire les pertes d'azote dans les agrosystèmes, élaborée au cours d'un atelier du colloque international *18th Nitrogen Symposium* (Lisbon, 2014).

Actions pour réduire les pertes en azote	Acteurs	politique	agriculteur	services agricoles	science
1. Avoir une bonne maîtrise de l'azote					
Avoir des alimentations équilibrées et précises pour les animaux			x		
Avoir un bon dosage des apports (engrais organiques et minéraux)			x		
Avoir des recommandations de fertilisation pour les engrais organiques et amendements		x		x	x
Avoir une bonne gestion de l'irrigation pour minimiser les pertes			x	x	
Avoir une bonne gestion du fumier (stockage, collecte)		x	x		
ex. un taux de stockage du fumier en fonction de la fertilisation, superficie de l'exploitation					
2. Recherche & Innovation					
Mettre en place des nouvelles formes de fertilisation			x	x	x
ex. mélanger des sources d'azote (organique et inorganique)					
Rechercher les relations N, P (limite des composts)				x	x
Réorganiser les différents systèmes de productions			x	x	x
ex. introduire des légumineuses en guise de compensation					
ex. avoir une complémentarité entre les cultures et le bétail					
ex. promouvoir les cultures intermédiaires					
Mettre en place des outils scientifique à la portée des agriculteurs					x
Mettre en place des outils rapide et économique pour déterminer l'efficience de l'azote d'un système			x		x
Avoir accès à la technologie dans les petites fermes			x	x	
Des outils dynamiques pour la fertilisation (sol, climat, culture)				x	x
Penser le système avec une gestion globale (pesticides, nutriments, ...)			x	x	x
ex. lier la génétique des plantes/animaux à l'utilisation de l'azote					
Mettre en place des politiques territoriales d'organisation et d'interactions entre les productions		x	x	x	x
ex. limiter les déconnexions entre la production végétale et animale					
Avoir une vision spatialisée et à long-terme des mesures et des outils de conseil			x	x	x
ex. mesurer l'efficience de l'azote sur une rotation, plutôt que sur une culture					
ex. considérer l'impact de l'utilisation de l'azote sur un territoire (ex. bassin versant)					
Impliquer les agriculteurs dans les programmes de recherches, les lieux de décisions et de conseil		x	x	x	x
3. Réglementation					
Avoir une réglementation appropriée pour l'application des engrais organiques		x			
ex. digestat de méthaniseur					
Encourager les analyses de sols pour une répartition de l'azote équilibrée		x	x	x	x
Introduire des incitations économiques pour les bonnes pratiques		x			x
ou mettre en place des taxes sur les engrais		x			